FINAL REPORT

Title: Long-term effects of restoration fire and thinning on soil fungi, fine root biomass, and litter depth

JFSP PROJECT ID: 12-1-01-20

January 2017

Jane E. Smith

USDA Forest Service, PNW Research Station

Daniel L. Luoma

Oregon State University, Dept. of Forest Ecosystems & Society

Benjamin T. N. Hart

Oregon State University, Dept. of Forest Ecosystems & Society

Table of Contents

Table of Tables

Table of Figures

Acknowledgements

We thank Jason Lyman and the Wallowa-Whitman National Forest for providing accommodations at the Billy Meadows Guard Station. Thanks to Lucas Longway and Tammy Jackson for help with collecting field samples, Elizabeth Bowman and Joseph Cagle for laboratory assistance, and Ariel Muldoon for statistical advice. Thank you to Drs. Kermit Cromack, Jr. and Jeff Hatten for providing insight about soils nutrient results. We also acknowledge Dr. Jim McIver, Karen Bennet, and Robin Darbyshire for their support and interest in this research project.

Acronyms & Abbreviations

BLAST	Basic Local Alignment Search Tool
°C	Degrees Celsius
C	Carbon
cm	Centimeter
CTAB	Cetyltrimethylammonium bromide
EMF	Ectomycorrhizal fungi
FFS	Fire & Fire Surrogate
ha	Hectare
ITS	Internal Transcribed Spacer
km	Kilometer
m	Meter
MRPP	Multiple Response Permutation Procedure
N	Nitrogen
NMS	Non-metric multidimensional scaling
Oa	O-horizon duff
Oe	O-horizon duff
Oi	O-horizon litter
OTU	Operational Taxonomic Unit
P	Phosphorus
PCR	Polymerase chain reaction

Long-term effects of restoration fire and thinning on soil fungi, fine root biomass, and litter depth

Abstract

To increase ecosystem resiliency, and achieve the desired future condition of stands with large tree retention and low fuel loads, federal agencies have actively implemented a large number of fuel reduction and forest restoration projects in low-elevation dry conifer forests throughout the western United States. A noteworthy example was the investigation of alternative methods of fuels reduction on the Hungry Bob field site in the Blue Mountains of northeastern Oregon. The multidisciplinary study later became the premier site for the nationwide Fire and Fire Surrogate study. At the Hungry Bob field site, treatments implemented in 1998 and 2000 included mechanical thinning of forested areas, prescribed fire, a combination of thinning followed by fire, and an untreated control. Soil sampling for this study occurred in 2014 and included four replications of each treatment for a total of 16 experiemental units. Study objectives were to evaluate the long-term impact of forest restoration practices on soil biochemistry and the mycorrhizal fungi associated with ponderosa pine (*Pinus ponderosa*). Additionally, the importance of litter depth as a measure of soil recovery and treatment impact was assessed. Bray-P, pH, and total carbon and nitrogen differed among treatments. Soil nutrient differences may have been driven by the thinning treatments and the resultant deposition of residual slash following harvesting or the consumption of slash by prescribed fire. Soil bulk density did not significantly differ among treatments. Previous research indicated that mycorrhizal species richness, live root biomass, and litter levels were reduced significantly in the short-term by prescribed fire treatments compared to the non-burned treatments. After more than a decade of recovery, mycorrhizal fungi in dry inland forests dominated by ponderosa pine returned to levels similar to the untreated controls. Similar litter depths across treatments suggest that litter depth stabilized over time in these forests. The results of this study demonstrate the resiliency of these forests to disturbances associated with restoration treatments, providing managers increased flexibility if maintaining abundant and persistent fungal communities for healthy soils is an objective.

Objectives

Re-measurement of prior studies – Effects of prescribed fire on vegetation, fuels, and soil. The objective of this study was to assess the ectomycorrhiza fungal (EMF) community in the dry-climate of eastern Oregon where fuels reduction treatments were implemented at the Hungry Bob study site over 15 years ago. Specifically, did the impact of mechanical thinning, prescribed burning, or a combined effect of the two compared to untreated forest stands have a transient or long-term effect?

1) **Quantify the effect of restoration thinning and prescribed fire on the ectomycorrhiza fungal (EMF) community and live root biomass**

2) **Assess the importance of litter depth as a measure of soil recovery and treatment impact**

The EMF community is defined as the richness, spatial frequency (number of treatment units and number of soil cores in which a species is found), and abundance (mycorrhizal root biomass - dry weight) of EMF species colonizing tree roots. Also investigated were differences among treatments in soil total C and N, Bray-P, pH, and soil bulk density as well as the recovery of litter which contributes to nutrient cycling near the soil surface. Although the early effects (one year post burning) of the restoration treatments with fire showed a significant reduction of EMF species richness, live root biomass, and litter levels (Smith et al. 2005), we hypothesized that after a decade these values would be similar among the prescribed fire and non-burned treatments. We hypothesized that the EMF community compositions would differ between the prescribed fire and non-burned treatments, reflecting a shift towards a dominance of fire-adapted EMF species in the prescribed burn treatments. We also hypothesized that a greater amount of fine root biomass would be found in the upper 5cm compared to the lower 5cm soil core.

Background

Increased emphasis on restoring and maintaining healthy landscapes along with reduction in limitations to fire management (FLAME Act 2009) and public education of the ecological benefits of fire, has heightened interest about the potential for prescribed fire and manual removal of woody materials from forested areas to decrease fire severity. In the Blue Mountains of eastern Oregon, at the original site of the nationwide Fire and Fire Surrogate study (FFS) network (http://www.fs.fed.us/ffs/sites.htm), fuels reduction with low-intensity prescribed fire and mechanical thinning/removal were studied at the operations

Figure 1 Prescribed fire (2000) after thinning (1998) at the Hungry Bob Site.

scale (Fig. 1). A limitation of previous research was that many studies were conducted at scales relatively easy to measure, but not applicable beyond a single site (McIver et al. 2000). Research at the first FFS project area, known as Hungry Bob, incorporated research considerations including wildlife, insects, economics, forest pathology, vegetation, fuels, soils, and soil microbial communities (Youngblood et al. 2006; McIver et al. 2013).

The objective of the FFS study was to determine the effects of fuels reduction treatments to transition fires from those of high severity, and often stand-replacing, to those of a low severity. Fire severity is a secondary metric related to fire intensity. Fire intensity relates to the energy released by fire (temperature, flames height, duration of heat pulse), while fire severity relates to the consequences of the fires intensity (amount of vegetation killed, effects on soil strength/properties, damage to roots) (Keeley 2009). High severity fires may burn at soil surface temperatures exceeding 300°C (Smith et al. 2016) and cause partial to total vegetation mortality aboveground and complete or near complete loss of belowground soil microbes in the top 15cm of the soil profile (Rundel 1983; Hebel et al. 2009; Reazin et al. 2016; Smith et al. 2017). In contrast, low severity fires typically experience temperatures below 100°C at the surface and remove mostly smaller shrubs and small diameter trees, leaving larger trees and soil microbes below the top 5cm intact (Agee 1973; Cowan et al. 2016).

Soil microbes tend to congregate in the upper soil profile where nutrient concentrations are highest (Oliver et al. 2015), decrease in frequency with increasing depth (Anderson et al. 2014), and respond quickly to ecosystem disturbances such as fire (Smith et al. 2004, 2005; Barker et al. 2013; Kageyama et al. 2013; Reazin et al. 2016; Smith et al. 2017). However, their long-term response to restoration treatment disturbances, exacerbated by over a century of fire suppression, is largely unknown. Forests in which low-intensity fire has been excluded have diminished release of nutrients bound in accumulated surface leaf litter, duff, and woody debris (Monleon and Cromack 1996; Monleon et al. 1997). This pool of nutrients, localized in the upper soil layer, creates a chemical gradient that presumably is followed by the belowground microbial community (Hart et al. 2005a). This accumulated duff and organic layer also contributes to increased soil heating in the event of a fire (Ryan, 2002). In a low-severity fire, flames typically have a low duration in any one place, translating into temperatures that barely penetrate the soil, leaving the microbial community intact. Deep organic layers at the soil surface provide fuel that increase the duration of time heat radiates into the soil (Busse et al. 2013; Smith et al. 2016), and have the potential to heat soil to temperatures lethal to soil microbes, which can remain altered after more than a decade (Klopatek et al., 1990).

Soil microbes are directly responsible in many ways for the survival of forest trees. Mycorrhizal fungi exist in a unique symbiotic relationship with most plant families, including the Pinaceae, where they create a sub-category of the symbiosis known as *ectomycorrhizae*. In this relationship, the symbiotic fungal partner expands the functional root network of the tree by growing over an increased surface area, making it possible to access water and nutrients that non-colonized tree roots would be unable to access on their own. In many forested systems, the primary limiting nutrients to tree growth are N and P. Mineralization (mobilization) of these nutrients is often facilitated by other soil microbes in the interstitial space between soil particles (Binkley and Fisher 2012) and are then absorbed and transported via the mycelial network to the fungal symbionts host tree. In exchange for nutrients, the host tree provides carbon (C) to the ectomycorrhizal fungi (EMF) in the form of sucrose, which is then converted to glycogen by the fungi (Smith and Read 2010). Additional services of EMF to the host tree include physical and chemical protection from antagonistic/pathogenic fungi (Smith and Read 2010), and connection and transfer of C among other host trees through *common mycorrhizal networks* (i.e. shared mycelial connections among trees) (Molina and Trappe 1982; Molina et al. 1992; Simard et al. 2015).

There is a large pool of research on mycorrhizal fungi response to experimental changes to their environment, but few studies investigate beyond a few growing seasons. Indeed, there is a paucity of long-term research on mycorrhizal responses to fire and fire-mitigation (Bastias et al. 2006; Dooley and Treseder 2012; Holden et al. 2013; Oliver et al. 2015; Overby et al. 2015). It could be expected that after forestry operations to reduce fire risk (mechanical thinning and prescribed fire), adverse effects on fungal populations could occur. Smith et al. (2005) determined this outcome, concluding that following restoration activities, fungal abundance and diversity were significantly decreased two years following thinning and one year following subsequent burning treatments. Some evidence suggests that long-term short rotation burning can alter the belowground EMF to a depth of up to 10cm (Bastias et al. 2006; Hart et al. 2005b). What is less well known is the long-term response of mycorrhizal fungi to restoration practices and if initial damage to the fungal communities creates long-term consequences.

Materials and Methods

Study Area

The Hungry Bob study site (www.fs.fed.us/ffs/docs/hb/pubs.html) is located on the Wallowa Valley Ranger District (Wallowa-Whitman NF) within the 12,000 ha "Waipiti Ecosystem Restoration Project" (Matzka 2003) and located between the Crow Creek and Davis Creek drainages (45° 38' N, 117° 13' W), about 45km north of Enterprise, OR (Fig. 2). Soils found at the study site are generally derived from ancient Columbia River basalts, which form steep topographies interspersed with numerous plateaus, draws, and ridges. In addition, the soil has received ash from pre-historic eruptions of ancient Mount Mazama and other volcanos in the Cascade Mountains to the west (Powers and Wilcox, 1964). Soil depth varies from deep to shallow depending on aspect and elevation,

Figure 2 Blue Mountains in the Wallowa Whitman NF near the Hungry Bob Site.

and soil profiles include Typic Vitrixerands from the Olot series, Vitrandic Argixerolls from the Melhorn and Larabee series, Lithic Ultic Haploxerolls from the Fivebit series, and Lithic Haploxerolls in the Bocker Series (Youngblood et al. 2006). In this area, soils generally support plant communities dominated by ponderosa pine and sometimes mixed with Douglas-fir (*Psuedotsuga menziesii* (Mirb.) Franco), grand fir, and some lodgepole pine (*Pinus contorta* Douglas ex Loudon var. *latifolia* Engelm. ex S. Watson). Elevation at the study area ranges from 1040m to 1480m (Youngblood et al. 2008). Annual temperatures average 7.8°C. Annual precipitation averages 50cm, along with an average of 66cm occurring in the form of snow. The two distinct periods of precipitation occur annually, with snow occurring in November to February and rain occurring in March (Youngblood et al. 2006).

Previous to the treatments, the forests surrounding the Hungry Bob were impacted by over a century of fire-suppression as well as timber harvesting and grazing pressure. Timber harvest early in the 20th century resulted in removal of most of the larger trees, and forests in the area are generally in the stem-exclusion phase (O'Hara et al. 1996). Trees found at the study site are predominately second-growth (harvesting occurred as recently as 1986) and exhibit the following characteristics: tree diameters average 33.5cm dbh, and stand basal areas averaging around 26 m^2 ha^{-1}, average SDI of 206, and post-harvest tree density of 26 ha^{-1}, with some residual large diameter trees that are 100-200 years old (Matzka 2003). The understory consists of the ponderosa pine/snowberry and Douglas-fir/snowberry plant associations (Johnson and Clausnitzer 1992), with vegetation dominated by snowberry (*Symporicarpos albus* (L.) S.F. Blake), white spirea (*Spirea betulifolia* Pall.), pinegrass (*Calamagrostis rubescens* Buckley), Idaho fescue (*Festuca idahoensis* Elmer) and bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh). The forest floor is covered in a layer of primarily ponderosa pine needles that averages 3cm in depth.

Experimental Design

The study was designed as an operations-scale experimental group of harvest units approximately 9400 ha (50 km^2) in size for the purpose of evaluating the economic and environmental impacts of restoration timber harvesting and prescribed underburning (Youngblood et al. 2008). From a set of 44 potential stands, 37 were determined to be the most suitable for experimental units in regard to topography and stand structure (Youngblood et. al. 2006). Of these, 16 units were randomly selected for operational experimentation. Experimental units varied from 10-20 ha in size. Four replications of each of the four treatment types were randomly assigned to units for a completely randomized study design.

The treatments were:

Control- Untreated or no action
Thinning- single entry from below
Prescribed Fire- single underburning of forest floor
Thin + Burn- thinning from below followed by prescribed underburning of forest floor

Thinning was conducted in 1998 via mechanical harvesting systems. Burning was scheduled for 1999 but due to unfavorable weather conditions was delayed until the fall of 2000. Restoration treatments were designed to create stands of trees that would adhere to/exceed the "80/80 Rule", where if a head fire were to pass through treated stands, 80% of the trees would survive under 80th percentile weather conditions (Weatherspoon & McIver 2000). For thinning treatments, silvicultural prescriptions for the stands prioritized establishment of irregular gaps and clumps of fire-resistant ponderosa pine into the dominant and co-dominant size classes. Trees were marked for harvesting by focusing on smaller diameter trees leaving behind trees to establish dominant and co-dominant classes. Treatments were designed to reduce total basal area from ~25 to ~16 m^2ha^{-1} (Youngblood et al. 2006). Felling, limbing, and bucking of trees was accomplished with tracked, single-grip harvesters; yarding was ground based with a wheeled forwarder that used logging residuals left on site to minimize soil compaction.


Figure 3 Thin + Burn treatment unit (Site 6B).

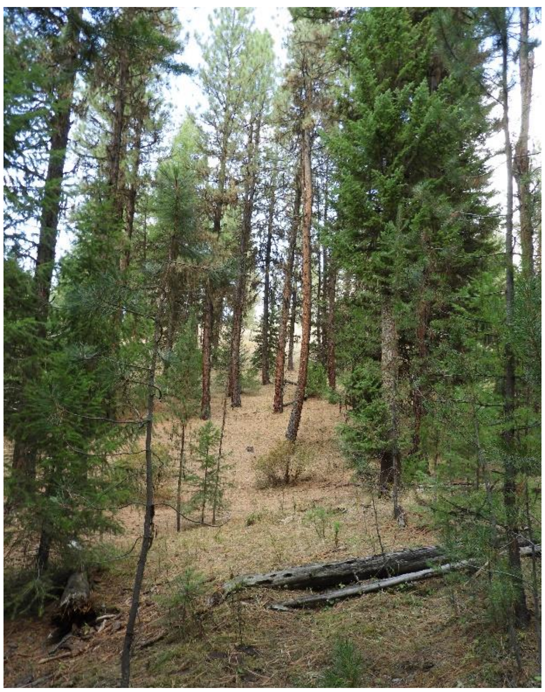
Figure 4 Control unit (Site 18).

Burn treatments had individual burn plans developed to address site specific characteristics of fuels and terrain (Youngblood et al. 2006). Burning included both backing and strip head fires depending on which would accomplish low flame heights. Ignition was from hand-carried drip torches, beginning in the early afternoons. Thin + Burn treatments followed protocols described for thinning (above) in 1998 and burning (above) in 2000, with the added caveat that burning procedures had to be modified in order to account for full combustion of logging residuals. Units randomly assigned to the no-treatment class were later determined to be dissimilar enough to the treated units (in regard to site characteristics and disturbance history) that they should not be considered "true" controls (Youngblood et al. 2006). Therefore, the use of "control" in referring to the no-treatment units is in reference to the original experimental design.

Plot establishment and re-establishment

Prior to any treatment, plot networks were established for later sampling by creating grids of points all 50m apart using a staff compass and no less than 50m from stand edge to address edge-related issues. At time of establishment, points were recorded with GPS units and marked with either wooden stakes in control and thin-only units (to prevent damage to harvest equipment and reduce costs) or metal posts in burn-only and thin + burn units (to relocate plots immediately following treatment). At the time of re-sampling for the current study, some plots were not readily relocated and were re-established within each stand. Using previous GPS coordinates and aerial photo map overlays, plot networks were surveyed until plots from original sampling were found, marked with flagging, and new GPS points recorded.

Soil Sampling

Within each experimental unit, three plots had been randomly selected at the time of the original sampling in June 1998 (Smith et al. 2005) and were re-sampled for the current study. Within each 0.2ha circular plot, a randomly pre-determined ponderosa pine tree over 20cm DBH was used for sampling. In cases where that tree had died or could not be located, the nearest ponderosa pine to plot was used. Each tree was followed from the base due east until the edge of the canopy (dripline) was reached. At dripline, a 20x20cm square was cut through the O-horizon, litter (Oi) was removed but the duff (Oe) was left intact. Litter depth was measured to the nearest 0.5cm. Two separate soil cores were taken at each plot for analysis of soil chemistry and ectomycorrhizas, respectively. A slide-hammer with a 5cm x 15cm coring head lined with three 5cm sleeves was used to extract soil (AMS Inc., American Falls, Idaho) (Fig. 2). Each core was separated into 0-5cm and 5-10cm depth classes; the lower 10-15 cm depth was not collected. Soil was placed into sealed plastic bags and stored on ice until brought to the lab for cold storage at 0°C.

Figure 5 Ben Hart with slide hammer soil corer.

Soil Chemistry Analysis

Soil chemical analysis was performed by the Central Analytical Laboratory (CAL) at Oregon State University. Prior to analysis, soils were dried at 105°C for 24 hours, sieved to 2mm, homogenized in a coffee grinder, and 40g samples were packaged in coin envelopes. Samples were analyzed for total C and total N using a dry combustion process utilizing an Elementar vario MACRO Cube. Bray-phosphorus (Bray-P) was determined using sodium bicarbonate extraction following protocols by Olsen et al. (1982) and modified by Horneck et al. (1989). pH was determined using a 1:2 soil to water ratio on an ATI Orion PerpHecT LogR Meter Model 350 following Horneck et al. (1989).

Fine Root Processing of Mycorrhizas

After being removed from the freezer, each sample was first soaked for no more than 12 hours at 4°C to loosen soil particles. Samples were then rinsed with tap water through a 2mm sieve to remove soil. Fine roots were isolated and kept in petri dishes for examination via stereomicroscopy. Roots were examined with a stereomicroscope (Zeiss Stemi SV6, Jena, Germany), at 40x or higher. Fine roots were examined for presence of mycorrhizal colonization and were classified by morphotype following criteria as described in the Colour Atlas of Ectomycorrhizae (Agerer 1997). Group morphotypes were kept in individual PCR tubes and stored in 2x cetyltrimethylammonium bromide (CTAB) at 0°C until molecular analysis.

Molecular Analysis

From each group morphotype, one root tip was selected for DNA extraction and amplification. Each root tip was pulverized with a micro-pestle and then DNA was extracted using a Sigma Extract-n-Amp kit™ (Sigma Aldrich, St. Louis, Missouri). The extracted DNA was then amplified via PCR of the internal transcribed spacer region (ITS) using primers ITS 1F and ITS 4 (White et al. 1990) and according to methods described by Cowan et al. (2016).

Purified PCR samples were shipped to the University of Kentucky Advanced Genetics Technology Center (UK-AGTC) for sequencing by Sanger Reaction (see Cowan et al. 2016). DNA sequences were compiled and analyzed using Geneious® v6.1.7. Operational taxonomic units (OTUs) were defined as those sequences having ≥ 98% similarity to each other. Sequences were analyzed for taxonomic placement using the blastn suite of the Basic Local Alignment Search Tool (BLAST) from the National Center for Biotechnology Information (NCBI) database (Altschul et al. 1997). The morphotyped root tips were given names to the lowest taxonomic level possible based on BLAST results and the following criteria: OTUs with ≥ 98% similarity over at least 500 base pairs were assigned to the species or species group level; at the genus level, OTUs generally had 96% to 97% similarity, similarities in the 90% to 95% range tended to group by family, and OTUs with similarities in the 80% to 94% range were assigned to an order if the top 5 matches were consistent at that level.

Statistical Analysis

Univariate soil nutrient data, species count data, and species abundance data were analyzed in RStudio v.0.99.491. Analysis of statistical assumptions yielded data that were all normally distributed except for total C and total N which were naturally log-transformed for statistical analysis and then back-transformed for ratio inference of treatment comparisons. An extension of a two-way analysis of variance was used that included a repeated measures (autoregressive 1) linear mixed model except for litter data that did not have a repeated measures element. Variation from sample units and depth were accounted for in the models. Mean soil nutrient differences and median ratios of differences for log-transformed data and the subsequent confidence intervals were estimated from the model.

Non-parametric multivariate comparisons of species abundance and soil nutrient data were conducted using PC-Ord™ software version 6.2 (McCune and Mefford 2009). Non-metric multidimensional scaling (NMS) ordinations were used to provide graphical representations of variation in community structure among treatments while accounting for soil nutrient data. Species with only 1 occurrence were removed to increase the chance of detecting trends. The analysis used Sørenson (Bray-Curtis) distance measures, run in auto-pilot, penalizing for ties, in "slow and thorough" mode, used 250 runs of real data and 249 runs of randomized data, to produce an NMS output with a stable 2-axis solution that had real and random stress values of 14.52 and 19.8, respectively, and passed the Monte Carlo significance test ($p=0.012$). Multi-response Permutation Procedure (MRPP) was used to test if there was a difference among communities based on treatment type. MRPP analysis provides a p-value for a test of the hypothesis of no difference between groups and an A statistic that represents the effect size of random chance, with $A=0$ meaning that significant results are no more or less due to chance, and $A=1$ means that all sample units within each group are identical.

Species accumulation curves were generated using the rarefaction sample-based estimator of EstimateS (Cowell, 2013). Data from three soil cores from each treatment replication (4) were used to generate the curves. The ectomycorrhizal fungus species richness curves were extrapolated in the model for samples 13–36.

Results and Discussion

Project Overview

In discussion of the results, it is important to note that the sampling methods for this study targeted the ectomycorrhizal community response. Our finding of similar fine root biomass in the upper and lower 5cm cores did not support the hypothesis that greater mycorrhizal abundance would be found in the upper core. However, microbial abundance and colonization is often highest near the soil surface (Oliver et al. 2015) and decreases with depth (Anderson et al., 2014). Knicker (2007) found that mycorrhizas tend to occur mainly in the top 2.5cm. Our methods included exclusion of the loose litter layer (Oi) but *included* the early and late litter de-compositional layers (Oe and Oa respectively). The inclusion of organic matter into the upper soil cores may help explain why values for C, N, and Bray P are higher and bulk density is lower in the upper soil cores than would be expected if only mineral soil had been collected.

Soil Physical Properties

Soil bulk density did not significantly differ among treatments ($F_{3,12}$ = 1.93, p =0.17, Table 1), even though it is known that prescribed fire can increase soil bulk density (Certini 2005). Bulk density could also be related to treatment placement on landforms within the study area. Although the treatments were assigned to sample units by complete randomization, the outcome by chance was that units with a burning component were often along ridges where soils were shallow and rocky; Thinning units typically had a lower slope position and deeper soils (Morici 2017). Harvesting equipment in Thinning units operated on trails covered with harvested tree limbs and tops, thus adding low-density organic material to the soil.

Litter accumulation (Oi) and subsequent "duff" depth (Oe and Oa) did not differ among treatments ($F_{3,12}$ = 0.36, p = 0.78, Table 1), providing strong evidence that productivity of these sites has recovered well since treatments. Litter accumulation may be correlated with densities of mycorrhizal root formation (Malajczuk and Hingston 1981) and the recovery of litter depth could contribute to our finding of no difference among treatments in mycorrhizal root biomass and EM species richness in this study (Erland and Taylor 2002). Smith et al. (2005) determined that one year post burning and two years post thinning at this study site, mycorrhizal root abundance as well as duff depth was significantly lower in the Dual treatment. Whereas now, litter depth and fine root biomass have recovered to levels that are similar to those found in the control.

Table 1 Effects of management treatments and soil depth on live root biomass, ectomycorrhizal species richness, soil properties, and nutrient content. Means are listed with standard errors in parentheses. Within a row, bold indicates a difference among treatments or between depths, means with a common lowercase letter are not different at $\alpha \leq 0.05$. NA = not applicable.

Response variable	Treatment				Depth	
	Control	Thin	Burn	Dual	Upper (0-5cm)	Lower (5-10cm)
Root biomass (g)	0.06 (0.03)a	0.05 (0.01)a	0.05 (0.01)a	0.07 (0.01)a	0.05 (0.01)a	0.06 (0.01)a
Species richness	3.13 (0.79)a	3.75 (0.92)a	2.75 (0.92)a	4.34 (0.98)a	4.06 (0.57)a	2.93 (0.67)a
C (%)	5.59 (1.03)a,b	**8.96 (2.19)a**	**5.07 (0.91)b**	**4.24 (0.69)b**	**8.49 (1.10)a**	**3.44 (0.24)b**
N (%)	0.29 (0.05)a,b	**0.39 (0.07)b**	0.28 (0.04)a,b	**0.24 (0.04)a**	**0.40 (0.04)a**	**0.19 (0.01)b**
C:N	18.95 (0.87)a,b	**21.57 (1.37)a**	**17.53 (1.11)b**	**17.89 (0.54)b**	**20.36 (0.84)a**	**17.61 (0.60)b**
Bray-P (mg kg^{-1})	**40.14 (2.66)a**	**64.52 (3.45)b**	**38.14 (5.34)a**	**46.90 (6.73)a**	**54.77 (3.79)a**	**40.08 (3.81)b**
pH	5.84 (0.05)a,b	**5.57 (0.07)a**	5.76 (0.07)a,b	**6.05 (0.05)**b	5.80 (0.08)a	5.81 (0.05)a
Bulk density (g cm	0.18 (0.06)a	0.15 (0.05)a	0.19 (0.05)a	0.19 (0.13)a	**0.15 (0.05)a**	**0.21 (0.03)b**
Litter depth (cm)	3.31 (1.4)a	3.24 (0.43)a	2.92 (0.95)a	2.67 (0.93)a	NA	NA

Soil Chemical Properties

In analysis of the long-term effects of fuels treatments within the study site, it was determined that although none of the treatments differed in soil biogeochemical and physical responses from the control (with the exception of greater plant available P in the Thinning treatment), treatments occasionally differed significantly in these responses from each other (Tables 1 and 2). The effects of the Thinning treatment created soil nutrient conditions that differed from those with a burning component, and differences with the Dual treatment in particular, were pronounced. This finding is likely attributable to the effects of the post-treatment woody harvesting residual debris and how it was managed. In the Thinning treatment, harvesting residuals were left on site to decompose. In the Dual treatment, harvesting residuals were incorporated into the subsequent burning. This management of the harvesting residuals may have resulted in elevated microbial activity from decomposition in the Thinning treatment and due to increased intensity of combustion of residuals, reduced microbial activity due to less organic matter in the Dual treatment. Moreover, annual temperature and precipitation regimes of the eastern Blue Mountains permit for mobilization/leaching of nutrients for a relatively short period each year, which could in part explain why trends created during initial treatment may still be detectable in this study over a decade post-treatment.

There were significant differences in total soil C among the treatments ($F_{3,12} = 4.02$, $p = 0.03$, Tables 1 and 2) and between depths ($F_{1,12} = 136.02$, $p > 0.01$, Tables 1 and 2) but no evidence of an interaction between treatment type and depth ($F_{3,12} = 2.71$, $p = 0.09$). Among treatments, median soil C content was estimated to be 1.75 times higher in the Thinning treatment compared to the Burn treatment ($p = 0.02$, DF = 12, CI = 1.21 to 2.56) and 2.07 times higher in the Thinning treatment compared to the Dual treatment ($p \leq 0.01$, DF = 12, CI = 1.41 to 3.02). Higher total soil C in the thinning treatment aligns with findings by Chatterjee et al. (2008) and Yanai et al. (2003) who hypothesized that long-term increased soil C following harvesting based restoration treatment was likely due to incorporation of forest residuals into the soil layer. Other mechanisms proposed include increased leaf/root turnover from the forb/shrub community (Campbell et al. 2009) and interactions of soil order and time (Nave et al. 2010).

Although there was no statistical difference among treatments of median soil N content ($F_{3,12} = 2.61$, $p = 0.1$), there was strong evidence that average N differed between soil depths ($F_{1,12} = 98.67$, $p < 0.01$) (Table 1) and no evidence of an interaction between treatment type and depth ($F_{3,12} = 0.91$, $p = 0.46$). Median N was estimated to be 1.99 times greater in the upper soil layer than the lower ($p = 0.009$, DF = 12, CI = 1.55 to 2.55). Nitrogen mobilization can increase as more labile sources of N are released by decomposition of slash in concert with labile C availability that promotes microbial activity (Grady and Hart 2006; Monleon and Cromack 1996). Johnson and Curtis (2001) report increases of up to 18% of soil C and N could be expected when tree harvesting residuals are left on site. Our detection of increases in N of up to 30% in the Thinning treatment is likely attributable to the incorporation of organic material found within our sampling methodology. Combustion of woody material initially leads to a pulse of N within the first few years of burning (Covington and Sackett 1992; Wan et al. 2001;

Schoch and Binkley 1986) but this effect diminishes within a few years (Binkley and Fisher 2012). Due to lack of further treatment since initial burning and little residual organic matter on the floor of burned units, it is probable that uptake by plants in burned units have left overall N levels lower than the Control and Thinning treatments.

Soil chemical processes are often linked to pH which may in part explain our findings for plant-available P. For pH there was evidence of a treatment level effect ($F_{1,12}$ = 5.92, p = 0.01) but no difference between depths ($F_{1,12}$ = 0.06, p = 0.81) (Tables 1 and 2), and no evidence of an interaction between treatment type and depth ($F_{3,12}$ = 1.42, p = 0.28). pH was estimated to be 0.45 units lower in the Thinning treatment compared to the Dual treatment (p = 0.01, DF = 12, CI = -0.45 to -0.23). For plant available P there was strong evidence of a treatment level effect ($F_{3,12}$ = 4.97, p = 0.02) as well as a depth effect ($F_{1,12}$ = 42.57, p < 0.01) (Tables 1 and 2) and no evidence of an interaction between treatment type and depth ($F_{3,12}$ = 1.71, p = 0.22). Mean Bray P was estimated to be 24.38 mg/kg soil greater in the Thinning treatment compared to the Control treatment (p = 0.01, DF = 12, CI = 9.68 to 39.08 mg/kg soil), 26.38 mg/kg soil greater in the Thinning treatment compared to the Burn treatment ($p \leq$ 0.01, DF = 12, CI = 12.82 to 39.94 mg/kg soil), and 17.62 Mg/kg soil greater in the Thinning treatment compared to the Dual treatment (p = 0.03, DF = 12, CI = 4.05 to 31.18). Mean Bray P content was estimated to be 14.68 mg/kg soil greater in the upper soil layer than the lower (p = 0.06, DF = 12, CI = 6.66 to 22.72 mg/kg soil).

In this study, plant-available P was the only response variable where the Thinning treatment was significantly different from the Control as well as Burning and Dual treatments. Ash production from combustion of organic material is known to increase soil pH (Neary et al. 1999; Arocena and Opio 2003, Moghaddas and Stevens 2007) due to the higher concentrations of hydroxides in ash and by the process of calcium, magnesium, and potassium displacing hydrogen and aluminum ions in soil. In areas such as the Blue Mountains where precipitation is low, this effect would last for several years. Values for pH were significantly different between the Dual and Thinning treatments (0.45 units lower in the Thinning). In a review of the Fire Fire-Surrogate Network by Stevens et al. (2012), it was determined that even at the network scale, higher pH also occurred in the Dual treatment. Deposition of harvest residuals that were then burned provided the organic material needed to raise pH in the Dual treatment. Conversely, unburned harvest residuals left in the Thinning treatment likely contributed to acidification of the soil (Binkley and Fisher 2012).

Bray-P can be elevated in response to fire. Temperatures exceeding 400°C can mobilize P into orthophosphates that are available for plant utilization (Binkley and Fisher 2012). However, such extreme temperatures generally are not found in prescribed fires where temperatures rarely exceed 120°C (Cowan et al. 2016) unless induced to higher levels by practitioners (Smith et al. 2016). In a 20 year interval study of controlled burns in SW Arizona, Wright and Hart (1997) found no differences in mineral P compared to controls. This is consistent with other authors (Binkley et al. 1992; Kaye et al. 2005; Saa et al. 1993) and the study presented here.

Given that the Control treatment units in this study were atypical compared to other pre-treatment areas in this study, one of the considerations addressed was if there were any existing differences in soil chemical make-up at the time of treatment. An exploratory ANOVA of pre- and post-treatment soil chemistry data revealed that prior to treatment, the Control was not significantly different from the other treatments. However, similar to the current study, measurements indicated a significant difference of soil C in the Thinning treatment compared to the Burn and Dual treatments. This finding indicates there may have been some inherent difference in stand structure or nutrient deposition from the canopy that was driving some of the post-treatment trends seen.

Table 2 Associated overall ANOVA *p*-values and Fisher's PLSD *p*-values for comparisons of differences among treatment means of those response variables showing significant differences in Table 1 (n = 4). Between-treatment comparison *p*-values of $\alpha \leq 0.05$ are shown in bold.

Comparison	*p*-value
C (p = 0.003)	
Control vs Thin	**0.02**
Control vs Burn	0.51
Control vs Dual	0.11
Thin vs Burn	**0.004**
Thin vs Dual	**0.0004**
Burn vs Dual	0.33
N (p = 0.02)	
Control vs Thin	0.08
Control vs Burn	0.89
Control vs Dual	0.15
Thin vs Burn	0.06
Thin vs Dual	**0.003**
Burn vs Dual	0.19
C:N ratio (p = 0.02)	
Control vs Thin	0.06
Control vs Burn	0.20
Control vs Dual	0.41
Thin vs Burn	**0.003**
Thin vs Dual	**0.01**
Burn vs Dual	0.63
Bray-P (p = 0.0005)	
Control vs Thin	**0.0003**
Control vs Burn	0.73
Control vs Dual	0.26
Thin vs Burn	**0.0001**
Thin vs Dual	**0.006**
Burn vs Dual	0.15
pH (p < 0.0001)	
Control vs Thin	**0.004**
Control vs Burn	0.37
Control vs Dual	**0.03**
Thin vs Burn	**0.03**
Thin vs Dual	**<0.0001**
Burn vs Dual	**0.003**

EMF Communities

Outcomes from the forest restoration treatments studied indicate that the ectomycorrhizal symbionts of ponderosa pine responded positively during a period of recovery without further disturbance. Mean species richness per soil core did not differ among treatments ($F_{3,12}$ = 0.41, p = 0.25) nor was there an effect of depth ($F_{1,12}$ = 2.53, p = 0.86) (Table 1). However, species accumulation curves for each treatment (n=4) showed that the density of ectomycorrhizal fungus species across the thinning treatment units was about 35% lower as compared to the other treatments when estimated for 36 modeled samples (Fig. 6). The lower rate of species accumulation detected across the thinning treatment suggests that over-all species richness for that treatment is lower than that of the other treatments (Fig. 6). The thinning treatment seems to support fewer less-common species.

Species richness and community composition are sensitive to N levels in the soil (Kluber et al. 2011). Luoma and Eberhart (2010) found that a one-time application of N (as urea) decreased feeder-root density by 62% and decreased ectomycorrhiza type richness by 33%. Those reductions in ectomycorrhizae closely mirrored the results of Nilsson and Wallander (2003) who found that EM fungi produced about 50% less mycelium in the soil after N fertilization (as ammonium sulfate) for ten years. Kårén and Nylund (1997) found a 49% reduction in EM root biomass associated with N fertilization (as ammonium sulfate) for six years prior to sampling. Conversely, Jonsson et al. (2000) found that moderate N additions (as ammonium nitrate) for four years had little effect on EM root tip density.

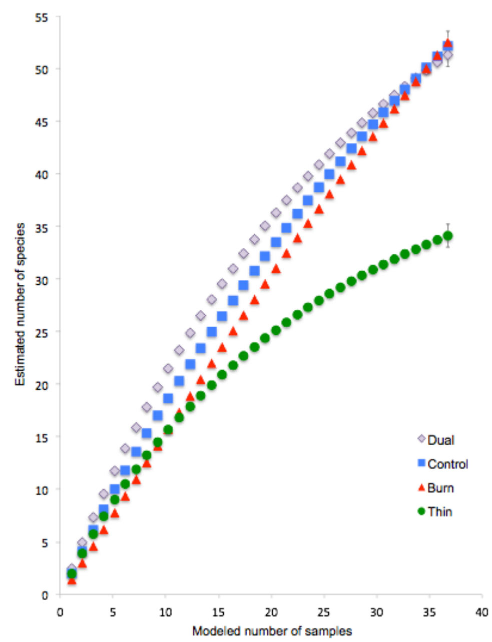

Figure 6 Species accumulation curves for each treatment (*n*=4) generated using the rarefaction sample-based estimator of EstimateS show that the density of ectomycorrhizal fungus species across the thinning treatment units is about 35% lower as compared to the other treatments when estimated for 36 modeled samples. The species richness curves are based on extrapolation for samples 13–36. The standard error intervals (vertical bars) are shown for the values estimated for 36 modeled samples.

Arnebrant and Soderstrom (1992) found that a one-time fertilization with 600 kg N ha^{-1}, 13 years prior to study, produced no difference in the total number of ectomycorrhizal root tips. However, yearly application of 30-80 kg N ha^{-1} over a period of 15 years, was associated with 20% fewer mycorrhizal roots. Other aspects of fertilization effects on EM were examined by Nilsson and Wallander (2003). They found that the addition of phosphorus ameliorated negative effects of N addition alone on mycelia growth of EM fungi.

For most soil chemistry and EMF community variables, the largest differences in responses were between the Thinning and Dual treatments. Although ash deposition has been thought to facilitate recovery of microorganisms in relation to increased pH (Knicker 2007), it could also be that nutrient depleted soils are driving the EMF response. Trees in soils that are rich in N and P have been found to support less abundant EMF (Nilsson et al. 2005; Toljander et al. 2006), presumably because less carbohydrates need to be allocated to root growth for absorption.

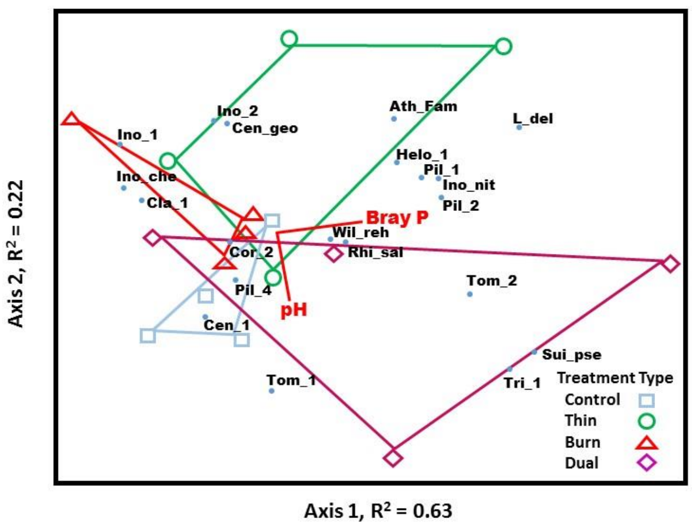

Figure 7 Graphical display of Non-metric Multidimensional Scaling (NMS) ordination. NMS produced according to relative abundance of EMF taxa (n=60 taxa in 16 treatment units and six environmental variables). Color coded lines represent the region in which treatment units are located. R^2 proportion of variance (using the Sørenson distance matrix) among treatment units explained by the axes.

Multi-Response Permutation Process (MRPP) results yielded no evidence of a difference in EMF composition among treatments (T-stat = -1.66, p = 0.06); test results also had a very low effect size (A = 0.04). Indicator Species Analysis showed that *Cenococcum* 1 was correlated with the Control treatment (p-value = 0.02, Indicator Value = 75.0).

The NMS ordination, based on sample units in species space, provided a stable two-axis solution (Fig. 7). The axes accounted for a total of 85.4% of the variation among sample units based on the species abundance, with axis 1 and 2 accounting for 63.1% and 22.2% of variation, respectively. Sample unit points that are closer to each other share a higher proportion of species similarity. Graphically grouping the sample units by treatment in the ordination space demonstrated moderate overlap (Fig. 7). None of the treatment types were completely isolated from the others. A phosphorus gradient was correlated with the Thinning and Dual treatments, aligning well with axis 1. Axis 2 showed alignment with pH, and was correlated more with the Control treatment than other treatments overall.

For live root biomass, there was no evidence of a treatment level effect ($F_{3,12} = 0.66$, $p = 0.59$) or an effect of depth ($F_{1,12} = 1.06$, $p = 0.32$) (Table 1) and no evidence of an interaction between treatment type and depth ($F_{3,12} = 0.34$, $p = 0.79$). Numerous studies point to EMF reductions after disturbance in the short-term (Smith et al. 2004; Smith et al. 2005; Barker et al. 2013) as well as the long-term (Klopatek et al. 1990). The historic fire return interval in the region where this study was conducted is estimated to be every 15-20 years (Heyerdahl et al. 2001). It could be expected that the EMF communities that had developed prior to fire suppression would have been adapted to recovering within this time period. Other studies support this idea, showing community similarity to controls in as little as 6 years after burning in the southeastern United States (Oliver et al. 2015), or recovered communities within 12 years in severely burned boreal systems (Treseder et al. 2004). Covington et al. (1997) estimated 10-20 years before "(metrics studied) stabilize around some long-term mean". Other authors suggested similar timeframes: two decades (Holden et al. 2013), twelve or more years (Fritze et al. 1993), or at a minimum a decade (Oliver et al. 2015) to fifteen years (Treseder at al. 2004) before soil microbial populations recover to pre-disturbance levels. All of these findings support 15 years as being an adequate period of time for microbial recovery, as was found in this study.

Negative effects from heat disturbance on microbial communities have been shown to be transient and community recovery has been reported (Hart et al. 2005a; Overby et al. 2015). Cowan et al. (2016) suggest that patchiness of burn intensities coupled with rapid hyphal penetration into areas where EMF were extirpated due to fire could be one mechanism for recolonization of areas previously burned. Latent spore banks within the soil have also been shown to facilitate EMF community recovery after fire (Bruns et al. 1995). The strip-head type ignition pattern used in the burning treatments produces a mosaic of patchiness, leaving areas of low to high intensity burning as well as unburned areas (Youngblood et al. 2008).

The vertical partitioning of mycorrhizal fungi was one of the questions of interest in this study, and evidence for EMF to preferentially distribute within the soil profile has been demonstrated by several authors (Beiler et al. 2012; Dickie et al. 2002; Rosling et al. 2003; Scattolin et al. 2008; Tedersoo et al. 2003). Statistically the upper partition of the cores was not greater in EMF than the lower, so this finding may warrent further investigation with more intensive sampling. Whether or not this could be attributed to litter recovery and the lack of nutrient cycling in this water-limited system that limits decomposition has yet to be investigated.

A total of 174 purified PCR samples submitted for sequencing yielded 117 taxa among treatments and depths, with 49 taxa (42%) in the upper soil depth and 68 taxa (58%) in the lower depth. Of these, 60 sequences were successfully assigned Family level designation or lower. Only 18 taxa occurred in two or more treatment units (Fig. 8), 45 (38%) taxa were detected from only 1 plot, indicating that species were widely scattered across the study site. This pattern of a few dominant and a large number of infrequent taxa was also indicated in Smith et al. (2005). Among the 18 most prevalent EMF taxa in this study, half reproduce by forming aboveground fleshy fruiting bodies (e.g. mushroom, club or cup fungi). The other form truffle-like fruiting bodies, are cryptic in various substrates, or are non-fruiting. The latter species are members of the genera *Cenococcum*, *Piloderma*, *Rhizopogon*, *or Tomentella*. These fungi, despite not having airborne spores, are well represented in studies of both short- and long-term responses to disturbance (Visser 1995, Dahlberg et al. 2001, Horton and Bruns 2001, Smith et al. 2004, 2005). It is possible that colonization by these typical early seral species can then persist across many seral stages, especially if the ecosystem is subject to regular disturbance such as fire.

Seventeen species, each with 1.7% or more of the total EMF biomass, accounted for 79% of the total EMF biomass (Fig. 9). More than half of the most frequent species (11/18) were also the most abundant (by root biomass). However, some abundant species were infrequently detected including the most abundant species, *Suillus lakei*, collected from a single core and exemplifying the patchy distribution of EMF. The presence of this species indicates that Douglas-fir roots were detected in this soil core since *Suillus lakei* only associates with Douglas-for. Dominance of a few ectomycorrhizal taxa within species-rich assemblages is common in pine forest systems (Gehring et al. 1998; Grogan et al. 2000; Stendell et al. 1999; Smith et al. 2004; 2005). EMF composition did not differ among treatments (MRPP: T-stat = -1.66, p = 0.06, A = 0.04). Ponderosa pine appears to associate with similar EMF assemblages across a fairly broad ecological range, as many of the more prominent species reported in Trappe et al. (2012) from southern Oregon were also found in this study.

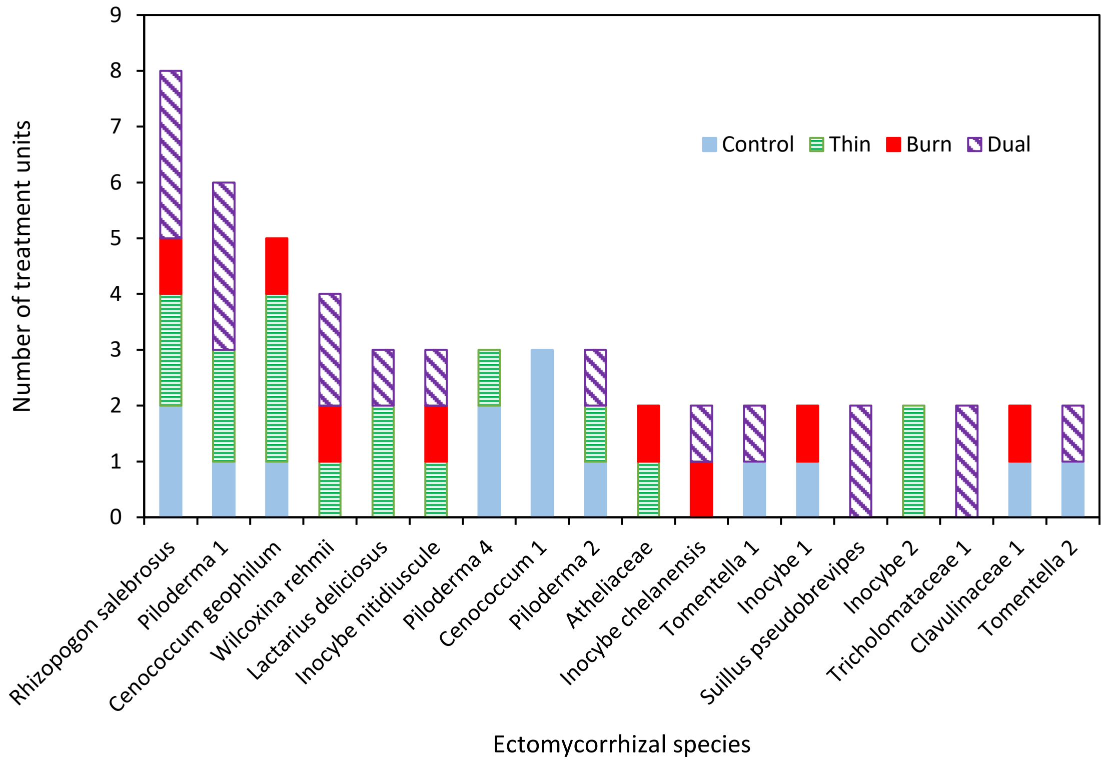

Figure 8 Species occurrence by treatment for the 18 most frequent EMF species (species occurring in two or more treatment units).

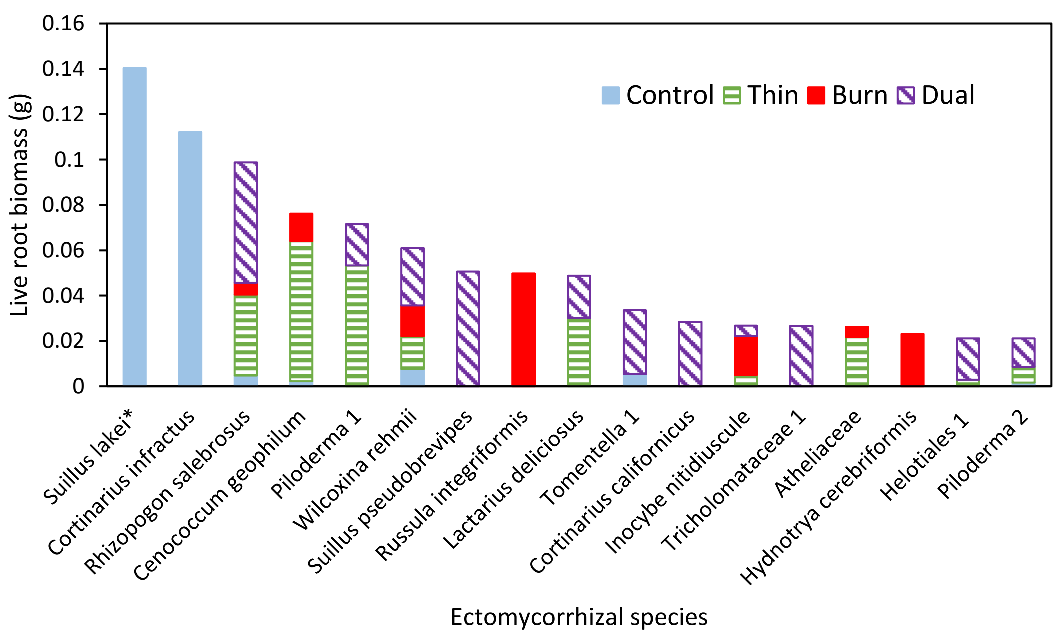

Figure 9 Root biomass by treatment for the 17 most abundant EMF species. Seventeen species, each with ≥0.02g or ≥1.7% of total biomass, accounted for 79% of the total biomass. *Known associated with *Pseudotsuga menziesii*.

Conclusions and Implications for Management

The results of this study suggest that effects of fuel reducing restoration efforts on the variables measured are largely short-term, and that recovery of soil biological, chemical, and physical attributes to levels similar to the control occurred within less than 15 years. The only treatment with significant differences from the control was the Thinning treatment, but of particular management interest is that the Thinning treatment in some cases differed from the Burn treatment and consistently differed from the Dual treatment. A 15-year recovery period is well supported as sufficient time for these soil responses to thinning and burning (Binkley et al. 1992; McKee 1982; Hart et al. 2005b; Busse et al. 2009). In a meta-analysis of the Fire Fire-Surrogate study as a nationwide network, Boerner et al. (2009) found that regardless of treatments across multiple landscapes and environments, the impacts to soil physical and chemical attributes were both “modest in magnitude and transient in duration”. At a site similar to the Hungry Bob, also located in the Blue Mountains of eastern Oregon, Hatten et al. (2008) found no differences in soil C, N, pH or plant-available nutrients 7 years after spring or fall prescribed burning compared to the control.

One of the strengths of the nationwide Fire and Fire Surrogate study was that the primary goal was to assess the effectiveness of fuels reduction treatments at what is known as “the operational scale” (Youngblood 2006). Treatments were implemented on a size scale that reflected typical management considerations and not at the smaller sizes more common in experimental research, often due to costs of implementation. Because even prescribed fires have inherent patchiness associated with their burning patterns, fire managers were instructed to ensure that sampling areas were burned in order to capture the effect of treatment. In the short-term (one year post-burn), both burning treatments significantly reduced EMF species richness, live root biomass, and litter depth (Smith et al. 2005). The general recovery of EMF communities to levels not differing from the control is encouraging with regard to the long-term effects of fuel reduction treatments.

Carbon, N, and Bray P levels were highest in the Thinning treatment and EMF response was not significantly different in the Dual treatment where nutrient levels were lowest. This EMF response may have a relationship to the demand for the services EMF provide in terms of nutrient acquisition. With regard to a changing mosaic of diverse habitats, if a management objective is to maintain high diversity and abundance of ectomycorrhizal fungus species on the landscape, then incorporating a burning component to a restoration management strategy would be preferred to thinning alone.

Unfortunately, even though a relatively low-cost strategy, prescribed burning alone is a resource sink and does not provide revenue. Thinning alone does accomplish the objective of lowering stand densities and modifying stand composition to accelerate development of late seral characteristics (Youngblood 2006) but it does not remove downed wood on the forest floor and as our study has shown, does not promote higher mycorrhizal presence compared to the Dual treatment. The Dual treatment, which removes organic material from the forest floor by combustion during prescribed burning also recoups expenses from revenue generated during the thinning process.

Burning operations conducted in the fall in eastern Oregon were shown to diminish surface layer EMF more than burning conducted during the spring (Smith et al. 2004). Since it is widely recognized that the process of restoring fire suppressed stands in the western United States will require multiple entries into stands (Hessburg et al. 2005; Spies et al. 2006; Youngblood et al. 2006; McIver et al. 2013), it is important to minimize negative impacts during preliminary entries to ensure ecosystem recovery.

Management focusing on mitigation of risks associated with high severity fire as it is relevant to individual stand conditions is needed. The use of thinning activities exclusively is useful for reducing forest crown connectivity and lowering stand densities while elevating lower canopy height by selecting for larger trees with more desirable crown structure. However, thinning alone promotes continued fine root development near the soil surface. Prescribed fire removes fuel from the forest floor and can remove smaller diameter trees by heat girdling, but cannot accomplish goals of broader tree removal from overly dense stands without particularly hot fires that may damage or remove desirable trees as well. Combining the two treatment types with a preparatory thin followed by prescribed burning captures the benefits of both. The results of this study support the use of thinning activities followed by prescribed burning as it relates to positively impacted long-term mycorrhizal responses. If managers seek to reduce the severity of future wildfires in the shortest and most economical manner, the use of prescriptions similar to those used in the Dual treatment may be the optimal option precluding stand specific conditions that would merit an alternative approach.

The results of this study indicate that EMF populations are able to re-establish in areas where they had previously declined as a result of initial thinning and burning restoration activities in the Blue Mountains of eastern Oregon (Smith et al. 2005). This finding supports a growing body of evidence that EMF communities are resilient to low-severity disturbance (Hart et al. 2005b; Overby et al. 2015; Cowan et al. 2016), and expands it to include the activities of mechanical thinning and prescribed burning. This research is valuable to land managers as it provides evidence that aggressive restoration treatments seemingly have little long-term effect on EMF and certain soil parameters and this new information may improve confidence in various options for future management. There is an increased need for accelerated restoration over large landscapes to reduce the risk of stand-replacing wildfire. The species richness and abundance of EMF were equally high in the Dual treatment as compared to the others, suggesting that lower soil nutrients found in association with regular fire return ecosystems may increase the importance of maintaining EMF in these forest types. Thinning alone resulted in soil conditions with higher nutrient values, while thinning followed by prescribed burning yielded lower-nutrient soil.

If managers are faced with resistance to prescribed burning, thinning alone can be an effective way of reducing stand basal area densities and promoting a discontinuous canopy. However, in order to return ecological processes to the fire-suppressed forests of the mountain west, inclusion of a burning component to restoration treatments is essential. Ultimately, objectives of land managers in the future will include creating stands of trees that are resistant to the effects of drought and higher fire-risk resulting from climate change. Reduction of competitive stress by reduction of stand basal area will be a primary approach of restoration efforts. Multiple entries to remove live trees will likely be required.

The findings of this study are limited to fire-suppressed low-elevation stands of ponderosa pine in the Blue Mountains. Further research to evaluate the outcomes presented here in other fire-suppressed forest ecosystems and across elevations would greatly widen the scope of knowledge about the effects of fuel treatments on EMF. The ability to broaden application of these insights would benefit land managers who are faced with an uncertain future of fire, but with a certain call to action.

Literature Cited

Agee, J.K. 1973. Prescribed fire effects on physical and hydrologic properties of mixed-conifer forest floor and soil. University of California Resources Center, Davis, CA, Report 143.

Agerer, R. 1997. Colour atlas of ectomycorrhizae. Einhorn-Verlag Eduard Dietenberger GmbH.

Altschul, S.F., Madden, T.L., Schäffer, A.A., Zhang, J., Zhang, Z., Miller, W., Lipman, D.J. 1997. Gapped BLAST and PSI-BLAST: a new generation of protein database search programs. *Nucleic Acids Research*, *25*(17), 3389–3402.

Anderson, I.C., Genney, D.R., Alexander, I.J. 2014. Fine-scale diversity and distribution of ectomycorrhizal fungal mycelium in a Scots pine forest. *New Phytologist*, *201*(4), 1423–1430.

Arnebrant, K., Soderstrom, B. 1992. Effects of different fertilizer treatments on ectomycorrhizal colonization potential in two Scots pine forests in Sweden. *Forest Ecology and Management, 53*, 77–89.

Arocena, J.M., Opio, C. 2003. Prescribed fire-induced changes in properties of sub-boreal forest soils. *Geoderma*, *113*(1), 1–16.

Barker, J.S., Simard, S.W., Jones, M.D., Durall, D.M. 2013. Ectomycorrhizal fungal community assembly on regenerating Douglas-fir after wildfire and clearcut harvesting. *Oecologia*, *172*(4), 1179–1189.

Bastias, B.A., Zhihong, X., Cairney, J.W.G. 2006. Influence of long-term repeated prescribed burning on mycelial communities of ectomycorrhizal fungi. *New Phytologist*, 172.1: 149–158.

Beiler, K.J., Simard, S.W., LeMay, V., Durall, D.M. 2012. Vertical partitioning between sister species of *Rhizopogon* fungi on mesic and xeric sites in an interior Douglas-fir forest. *Molecular Ecology*, *21*(24), 6163–6174.

Binkley, D., Richter, D., David, M.B., Caldwell, B. 1992. "Soil chemistry in a loblolly/longleaf pine forest with interval burning." *Ecological Applications*, 2.2: 157–164.

Binkley, D., Fisher, R. 2012. *Ecology and Management of Forest Soils*. John Wiley & Sons.

Boerner, R.E.J., Huang, J., Hart, S.C. 2009. Impacts of Fire and Fire Surrogate treatments on forest soil properties: a meta-analytical approach. *Ecological Applications*, 19(2), 338–358

Bruns, T.D., Baar, J., Grogan, P., Horton, T.R., Kretzer, A.M., Redecker, D., Taylor, D.L. 2005. Community dynamics of ectomycorrhizal fungi following the vision fire. In: U.S. Department of the Interior, National Park Service. Vision Fire: Lessons Learned from the October 1995 Fire. *Point Reyes National Seashore*, pp. 33–40.

Busse, M.D., Cochran, P.H., Hopkins, W.E., Johnson, W.H., Riegel, G.M., Fiddler, G.O., Shestak, C.J. 2009. Developing resilient ponderosa pine forests with mechanical thinning and prescribed fire in central Oregon's pumice region. *Canadian Journal of Forest Research*, *39*(6), 1171–1185.

Busse, M.D., Shestak, C.J., Hubbert, K.R. 2013. Soil heating during burning of forest slash piles and wood piles. *International Journal of Wildland Fire*, 22(6): 786–796.

Campbell, J., Alberti, G., Martin, J., Law, B.E. 2009. Carbon dynamics of a ponderosa pine plantation following a thinning treatment in the northern Sierra Nevada. *Forest Ecology and Management*, *257*(2), 453–463.

Certini, G. 2005. Effects of fire on properties of forest soils: a review. *Oecologia*, *143*(1), 1–10.

Chatterjee, A., Vance, G.F., Pendall, E., Stahl, P.D. 2008. Timber harvesting alters soil carbon mineralization and microbial community structure in coniferous forests. *Soil Biology and Biochemistry*, *40*(7), 1901–1907.

Covington, W.W., Sackett, S.S. 1992. Soil mineral nitrogen changes following prescribed burning in ponderosa pine. *Forest Ecology and Management*, *54*(1), 175–191.

Covington, W.W., Fule, P.Z., Moore, M.M., Hart, S.C., Kolb, T.E., Mast, J.N., Wagner, M.R. 1997. Restoring ecosystem health in ponderosa pine forests of the Southwest. *Journal of Forestry*, *95*(4), 23.

Cowan, A.D., Smith, J.E., Fitzgerald, S.A. 2016. Recovering lost ground: effects of soil burn intensity on nutrients and ectomycorrhizal communities of ponderosa pine seedlings. *Forest Ecology and Management*, 378, 160–172.

Dahlberg, A., Schimmel, J., Taylor, A.F.S., Johannesson, H. 2001. Post-fire legacy of ectomycorrhizal fungal communities in the Swedish boreal forest in relation to fire severity and logging intensity. *Biological Conservation*, 100, 151–161.

Dickie, I.A., Xu, B., Koide, R.T. 2002. Vertical niche differentiation of ectomycorrhizal hyphae in soil as shown by T-RFLP analysis. *New Phytologist*, *156*(3), 527–535.

Dooley, S.R., Treseder, K.K. 2012. The effect of fire on microbial biomass: a meta-analysis of field studies. *Biogeochemistry*, *109*(1–3), 49–61.

Erland, S., Taylor, A.F.S. 2002. Diversity of fungal ectomycorrhizal communities in relation to the abiotic environment. In: Mycorrhizal Ecology (Editors: M. van der Heijden and I. Sanders). Ecological Studies vol. 157. Springer pp. 163–199.

Fritze, H., Pennanen, T., Pietikäinen, J. 1993. Recovery of soil microbial biomass and activity from prescribed burning. *Canadian Journal of Forest Research*, *23*(7), 1286–1290.

Gehring, C.A., Theimer, T.C., Whitham, T.G., Keim, P. 1998. Ectomycorrhizal fungal community structure of pinyon pines growing in two environmental extremes. *Ecology*, *79*(5), 1562–1572.

Grady, K.C., Hart, S.C. 2006. Influences of thinning, prescribed burning, and wildfire on soil processes and properties in southwestern ponderosa pine forests: a retrospective study. *Forest Ecology and Management*, 234.1: 123–135.

Grogan, P., Baar, J., Bruns, T.D. 2000. Below-ground ectomycorrhizal community structure in a recently burned bishop pine forest. *Journal of Ecology*, *88*(6), 1051–1062.

Hart, S.C., Classen, A.T., Wright, R.J. 2005b. Long-term interval burning alters fine root and mycorrhizal dynamics in a ponderosa pine forest. *Journal of Applied Ecology*, *42*(4), 752–761.

Hart, S.C., DeLuca, T.H., Newman, G.S., MacKenzie, M.D., Boyle, S.I. 2005a. Post-fire vegetative dynamics as drivers of microbial community structure and function in forest soils. *Forest Ecology and Management*, *220*(1), 166–184.

Hatten, J.A., Zabowski, D., Ogden, A., Thies, W. 2008. Soil organic matter in a ponderosa pine forest with varying seasons and intervals of prescribed burn. *Forest Ecology and Management*, *255*(7), 2555–2565.

Hebel, C.L., Smith, J.E., Cromack Jr., K. 2009. Invasive plant species and soil microbial response to wildfire burn severity in the Cascade Range of Oregon. *Applied Soil Ecology*, *42*(2), 150–159.

Hessburg, P.F., Agee, J.K., Franklin, J.F. 2005. Dry forests and wildland fires of the inland Northwest USA: contrasting the landscape ecology of the pre-settlement and modern eras. *Forest Ecology and Management*, *211*(1), 117–139.

Heyerdahl, E.K., Brubaker, L.B., Agee, J.K. 2001. Spatial controls of historical fire regimes: a multiscale example from the interior west, USA. *Ecology*, *82*(3), 660–678.

Holden, S.R., Gutierrez, A., Treseder, K.K. 2013. Changes in soil fungal communities, extracellular enzyme activities, and litter decomposition across a fire chronosequence in Alaskan boreal forests. *Ecosystems*, *16*(1), 34–46.

Horneck, D.A., Hart, J.M., Topper, K., Koepsell, B. 1989. *Methods of soil analysis used in the soil testing laboratory at Oregon State University*. [Corvallis, Or.]: Agricultural Experiment Station, Oregon State University.

Horton, T.R., Bruns, T.D. 2001. The molecular revolution in ectomycorrhizal ecology: peeking into the black-box. *Molecular Ecology*, 10, 1855–1871.

Johnson, C.G., Clausnitzer, R.R. 1992. Plant associations of the Blue and Ochoco Mountains. Technical Publication R6-ERW-TP-036-92. Portland, OR: US Department of Agriculture, Forest Service. Pacific Northwest Region, Wallowa-Whitman National Forest.

Johnson, D.W., Curtis, P.S. 2001. Effects of forest management on soil C and N storage: meta-analysis. *Forest Ecology and Management*, *140*(2), 227–238.

Jonsson, L., Dahlberg, A., Tor-Erik B. 2000. Spatiotemporal distribution of an ectomycorrhizal community in an oligotrophic Swedish *Picea abies* forest subjected to experimental nitrogen addition: above- and below-ground views. *Forest Ecology and Management, 132*, 143–156.

Kageyama, S.A., Posavatz, N.R., Jones, S.S., Waterstripe, K.E., Bottomley, P.J., Cromack, K., Myrold, D.D. 2013. Effects of disturbance scale on soil microbial communities in the Western Cascades of Oregon. *Plant and Soil*, *372*(1-2), 459–471.

Kårén O., Nylund, J-E. 1997. Effects of ammonium sulphate on the community structure and biomass of ectomycorrhizal fungi in a Norway spruce stand in southwestern Sweden. *Canadian Journal of Botany*, 75: 1628–1642.

Kaye, J.P., Hart, S.C., Fulé, P.Z., Covington, W.W., Moore, M.M., Kaye, M.W. 2005. Initial carbon, nitrogen, and phosphorus fluxes following ponderosa pine restoration treatments. *Ecological Applications*, *15*(5), 1581–1593.

Keeley, J.E. 2009. Fire intensity, fire severity and burn severity: a brief review and suggested usage. *International Journal of Wildland Fire,* 18.1: 116–126.

Klopatek, C.C., DeBano, L.F., Klopatek, J.M. 1990. Impact of fire on the microbial processes in pinyon-juniper woodlands: management implications. *Effects of fire management of southwestern natural resources.* USDA Forest Service, Fort Collins, Colorado, USA, 197–205.

Kluber, L.A., Smith, J.E., Myrold, D.D. 2011. Distinctive fungal and bacterial communites are associated with mats formed by ectomycorrhizal fungi. *Soil Biology & Biochemistry*, 43, 1042–1050.

Knicker, H. 2007. How does fire affect the nature and stability of soil organic nitrogen and carbon? A review. *Biogeochemistry,* 85.1: 91–118.

Luoma, D.L., Eberhart, J.L. 2010. Third-year response of ectomycorrhizae to soil nutritional amendments across a gradient of SNC disease. Pp 82-92 *in*: Mulvey, R., Shaw, D. (eds.). Swiss Needle Cast Cooperative Annual Report 2010, College of Forestry, Oregon State University, Corvallis. 106p.

Malajczuk, M., Hingston, F.J. 1981. Ectomycorrhizae associated with Jarrah. *Australian Journal of Botany,* 29.4: 453–462.

Matzka, P.J. 2003. *Thinning with prescribed fire and timber harvesting mechanization for fuels reduction and forest restoration.* Doctoral dissertation. College of Forestry, Oregon State University.

McCune, B., Mefford, M.J. 2009. PC-ord. *Multivariate analysis of ecological data, version,* 6.0. MjM Software, Gleneden Beach, OR, USA

McKee, W.H. 1982. Changes in soil fertility following prescribed burning on coastal plain pine sites. *Southern Forest Experiment Station Asheville N.C., 234*, 1–23.

McIver, J.D., Stephens, S.L., Agee, J.K., Barbour, J., Boerner, R.E., Edminster, C.B., Haase, S. 2013. Ecological effects of alternative fuel-reduction treatments: highlights of the National Fire and Fire Surrogate study (FFS). *International Journal of Wildland Fire*, *22*(1), 63–82.

McIver, J., Youngblood, A., Niwa, C., Smith, J., Ottmar, R., Matzka, P. 2000. Alternative fuel reduction methods in Blue Mountain dry forests: An introduction to the Hungry Bob project. In *Proceedings of Joint Fire Science Conference* (pp. 15–17).

Moghaddas, E.E., Stephens, S.L. 2007. Thinning, burning, and thin-burn fuel treatment effects on soil properties in a Sierra Nevada mixed-conifer forest. *Forest Ecology and Management*, *250*(3), 156-166.

Molina, R., Trappe, J.M. 1982. Lack of mycorrhizal specificity by the ericaceous hosts *Arbutus menziesii* and *Arctostaphylous ova-ursi*. *New Phytologist*, *90*(3), 495–509.

Molina, R., Massicotte, H. Trappe, J.M. 1992. Specificity phenomena in mycorrhizal symbioses: community-ecological consequences and practical implications. In: *Mycorrhizal functioning: an integrative plant-fungal process*, 357–423.

Monleon, V.J., Cromack, K. 1996. Long-term effects of prescribed underburning on litter decomposition and nutrient release in ponderosa pine stands in central Oregon. *Forest Ecology and Management*, *81*(1-3), 143–152.

Monleon, V.J., Cromack, Jr, K., Landsberg, J. D. 1997. Short-and long-term effects of prescribed underburning on nitrogen availability in ponderosa pine stands in central Oregon. *Canadian Journal of Forest Research*, *27*(3), 369–378.

Morici, K. 2017. Fuel treatment longevity in the Blue Mountains of Oregon. Master's Thesis, 65p, College of Forestry, Oregon State University.

Nave, L.E., Vance, E.D., Swanston, C.W., Curtis, P.S. 2010. Harvest impacts on soil carbon storage in temperate forests. *Forest Ecology and Management*, *259*(5), 857–866.

Neary, D.G., Klopatek, C.C., DeBano, L.F., Ffolliott, P.F. 1999. Fire effects on belowground sustainability: a review and synthesis. *Forest Ecology and Management*, *122*(1), 51–71.

Nilsson, L.O. Wallander, H. 2003. Production of external mycelium by ectomycorrhizal fungi in a Norway spruce forest was reduced in response to nitrogen fertilization. *New Phytologist*, 158(2), pp.409–416.

Nilsson, L.O., Giesler, R., Bååth, E., Wallander, H. 2005. Growth and biomass of mycorrhizal mycelia in coniferous forests along short natural nutrient gradients. *New Phytologist*, *165*(2), 613–622.

O'Hara, K.L., Latham, P.A., Hessburg, P., Smith, B.G. 1996. A structural classification for inland northwest forest vegetation. *Western Journal of Applied Forestry*, 11(3): 97–102.

Oliver, A.K., Callaham, M.A., Jumpponen, A. 2015. Soil fungal communities respond compositionally to recurring frequent prescribed burning in a managed southeastern US forest ecosystem. *Forest Ecology and Management*, *345*, 1–9.

Olsen, S.R., Sommers, L.E., Page, A.L. 1982. Methods of soil analysis. Part 2. *Chemical and microbiological properties of Phosphorus. ASA Monograph*, (9), 403–430.

Overby, S.T., Owen, S.M., Hart, S.C., Neary, D.G., Johnson, N.C. 2015. Soil microbial community resilience with tree thinning in a 40-year-old experimental ponderosa pine forest. *Applied Soil Ecology*, *93*, 1–10.

Powers, H.A., Wilcox, R.E., 1964. Volcanic ash from Mount Mazama (Crater Lake) and from Glacier Peak. Science 144, 1334–1336.

Reazin, C., Morris, S., Smith, J.E., Cowan, A.D., Jumpponen, A. 2016. Fires of differing intensities rapidly select distinct soil fungal communities in a Northwest US ponderosa pine forest ecosystem. *Forest Ecology and Management*, *377*, 118–127.

Rosling, A., Landeweert, R., Lindahl, B.D., Larsson, K.H., Kuyper, T.W., Taylor, A.F.S., Finlay, R.D. 2003. Vertical distribution of ectomycorrhizal fungal taxa in a podzol soil profile. *New Phytologist*, *159*(3), 775–783.

Rundel, P.W. 1983. Impact of fire on nutrient cycles in Mediterranean-type ecosystems with reference to chaparral. *Mediterranean-type ecosystems*. Springer Berlin Heidelberg, 192–207.

Ryan, K.C. 2002. Dynamic interactions between forest structure and fire behavior in boreal ecosystems. *Silva Fennica* 36(1): 13–39.

Saa, A., Trasar-Cepeda, M.C., Gil-Sotres, F., Carballas, T. 1993. Changes in soil phosphorus and acid phosphatase activity immediately following forest fires. *Soil Biology and Biochemistry*, *25*(9), 1223–1230.

Scattolin, L., Montecchio, L., Mosca, E., Agerer, R. 2008. Vertical distribution of the ectomycorrhizal community in the top soil of Norway spruce stands. *European Journal of Forest Research*, *127*(5), 347-357.

Schoch, P., Binkley, D. 1986. Prescribed burning increased nitrogen availability in a mature loblolly pine stand. *Forest Ecology and Management*, *14*(1), 13–22.

Simard, S., Asay, A., Beiler, K., Bingham, M., Deslippe, J., He, X., Teste, F. 2015. Resource transfer between plants through ectomycorrhizal fungal networks. In *Mycorrhizal Networks* (pp. 133–176). Springer Netherlands.

Smith, J.E., McKay, D., Niwa, C.G., Thies, W.G., Brenner, G., Spatafora, J.W. 2004. Short-term effects of seasonal prescribed burning on the ectomycorrhizal fungal community and fine root biomass in ponderosa pine stands in the Blue Mountains of Oregon. *Canadian Journal of Forest Research*, *34*(12), 2477–2491.

Smith, J.E., McKay, D., Brenner, G., McIver, J., Spatafora, J.W. 2005. Early impacts of forest restoration treatments on the ectomycorrhizal fungal community and fine root biomass in a mixed conifer forest. *Journal of Applied Ecology*, *42*(3), 526–535.

Smith, J.E., Cowan, A.D., Fitzgerald, S.A. 2016. Soil heating during the complete combustion of mega-logs and broadcast burning in central Oregon USA pumice soils. *International Journal of Wildland Fire*, *25*(11), 1202–1207.

Smith, J.E., Kluber, L.A., Jennings, T.N., McKay, D., Brenner, G., Sulzman, E.W. 2017. Does the presence of large down wood at the time of a forest fire impact soil recovery? *Forest Ecology and Management*, *391*, 52–62.

Smith, S.E., Read, D.J. (2010). *Mycorrhizal symbiosis*. 3rd Edition. Academic press.

Spies, T.A., Hemstrom, M.A., Youngblood, A., Hummel, S. 2006. Conserving old-growth forest diversity in disturbance-prone landscapes. *Conservation Biology*, *20*(2), 351–362.

Stendell, E.R., Horton, T.R., Bruns, T.D. 1999. Early effects of prescribed fire on the structure of the ectomycorrhizal fungus community in a Sierra Nevada ponderosa pine forest. *Mycological Research*, *103*(10), 1353–1359.

Tedersoo, L., Kõljalg, U., Hallenberg, N., Larsson, K.H. 2003. Fine scale distribution of ectomycorrhizal fungi and roots across substrate layers including coarse woody debris in a mixed forest. *New Phytologist*, *159*(1), 153–165.

Toljander, J.F., Eberhardt, U., Toljander, Y.K., Paul, L.R., & Taylor, A.F. 2006. Species composition of an ectomycorrhizal fungal community along a local nutrient gradient in a boreal forest. *New Phytologist*, *170*(4), 873–884.

Trappe, M.J., Cromack Jr, K., Caldwell, B.A., Griffiths, R.P., Trappe, J.M. 2012. Diversity of mat-forming fungi in relation to soil properties, disturbance, and forest ecotype at Crater lake National Park, Oregon, USA. *Diversity, 4,* 196-223.

Treseder, K.K., Mack, M.C., Cross, A. 2004. Relationships among fires, fungi, and soil dynamics in Alaskan boreal forests. *Ecological Applications*, *14*(6), 1826–1838.

Visser, S. 1995. Ectomycorrhizal fungal succession in jack pine stands following wildfire. *New Phytologist*, *129*(3), 389–401.

Wan, S., Hui, D., Luo, Y. 2001. Fire effects on nitrogen pools and dynamics in terrestrial ecosystems: A meta-analysis. *Ecological Applications*, *11*(5), 1349–1365.

Weatherspoon, C.P., McIver, J. 2000. A national study of the consequences of fire and fire surrogate treatments. *USDA Forest Service Pacific Southwest Research Station, Redding, California, USA.*

Wright, R.J., Hart, S.C. 1997. Nitrogen and phosphorus status in a ponderosa pine forest after 20 years of interval burning. *Ecoscience*, *4*(4), 526–533.

Yanai, R.D., Currie, W.S., Goodale, C.L. 2003. Soil carbon dynamics after forest harvest: an ecosystem paradigm reconsidered. *Ecosystems*, *6*(3), 197-212.

Youngblood, A., Metlen, K.L., Coe, K. 2006. Changes in stand structure and composition after restoration treatments in low elevation dry forests of northeastern Oregon. *Forest Ecology and Management*, *234*(1), 143–163.

Youngblood, A., Wright, C.S., Ottmar, R.D., McIver, J.D. 2008. Changes in fuelbed characteristics and resulting fire potentials after fuel reduction treatments in dry forests of the Blue Mountains, northeastern Oregon. *Forest Ecology and Management*, *255*(8), 3151–3169.

Appendix A: Contact Information for Key Project Personnel

Jane E. Smith, Principal Investigator (PI)
Pacific Northwest Research Station
Corvallis Forestry Sciences Laboratory
3200 SW Jefferson Way
Corvallis, OR 97331
Jsmith01@fs.fed.us
541-750-7387

Daniel L. Luoma, Co-principal Investigator (COI)
Oregon State University
Dept. of Forest Ecosystems and Society
321 Richardson Hall
Corvallis, OR 97331
Daniel.luoma@oregonstate.edu
541-737-2244

Benjamin T. N. Hart, Graduate Research Assistant (GRA)
Oregon State University
Dept. of Forest Ecosystems and Society
321 Richardson Hall
Corvallis, OR 97331
Forestry.fungi@gmail.com

Appendix B: Deliverables

1. **Articles**

   Hart, B.T.N., Smith, J.E., Luoma, D.L., Hatten, J.A. Recovery of ectomycorrhizal fungus communities fifteen years after fuels reduction treatments in ponderosa pine forests of the Blue Mountains, Oregon. Submitted to Forest Ecology & Management, 12/21/2017

2. **Technical Reports**

3. **Text books or book chapters**

4. **Graduate Thesis**

   Hart BTN. Fuel treatments of ponderosa pine (*Pinus ponderosa*) in the Blue Mountains of eastern Oregon: A Mycorrhiza Perspective. April 20, 2017. MS Thesis. Department of Forest Ecosystems and Society, Oregon State University, Corvallis, Oregon.

5. **Conference or symposium proceedings scientifically recognized and referenced**

   Smith JE. 2013. Forestry events and issues shape mycological research in the Pacific Northwestern United States. In: Proceedings of the International Symposium on Forest Mushrooms. August 6, 2013. Korea Forest Research Institute, Seoul, Korea. pp. 103-124.

6. **Conference or symposium abstracts**

   Hart BT, Smith JE, Luoma DL. Fire in the Future, Lessons from the Past: Forest Fire Fuel Reduction Treatment Impacts on Ponderosa Pine Mycorrhizal Fungi Diversity. Mycological Society of America Meeting, August 8-12, 2016, Berkeley, California.

   Hart B, Smith JE, Luoma D. Forest fire fuel reduction treatment impacts on ponderosa pine mycorrhizal fungi diversity. Northwest Scientific Association 87th Annual Meeting, March 23-26, 2016, Central Oregon Community College, Bend, Oregon.

   Hart B, Smith JE, Luoma D. Fire in the Future, Lessons from the Past: Forest Fire Fuel Reduction Treatment Impacts on Ponderosa Pine Mycorrhizal Fungi Diversity. 6th International Fire Ecology & Management Congress, November 16-20, 2015, San Antonio, Texas.

   Hart BT, Smith JE, Luoma DL. Fire in the future, lessons from the past: Mycorrhizal perspectives on Federal forest fire management practices. Strategies to preserve and restore mycorrhizas for sustainable forestry, 8th International Conference on Mycorrhiza, August 3-7, 2015, Flagstaff, Arizona.

Hart B, Smith JE. Forest fire fuel reduction treatments and the impacts on fungi in the Blue Mountains of eastern Oregon: A mycorrhizal perspective. Western Forestry Graduate Research Symposium, April 21-22, 2014, Richardson Hall, OSU, Corvallis, Oregon.

Hart B, Smith JE. Forest fire fuel reduction treatments and the impacts on fungi in the Blue Mountains of eastern Oregon: A mycorrhizal perspective. Central Oregon Fire Science Symposium. April 7-11, 2014. Bend, Oregon.

Smith JE. Forestry events and issues shape mycological research in the Pacific Northwestern United States. In: Proceedings of the International Symposium on Forest Mushrooms. August 6, 2013. Korea Forest Research Institute, Seoul, Korea.

7. Posters

Hart BT, Smith JE, Luoma DL. Fire in the future, lessons from the past: Mycorrhizal perspectives on Federal forest fire management practices. Strategies to preserve and restore mycorrhizas for sustainable forestry, 8th International Conference on Mycorrhiza, August 3-7, 2015, Flagstaff, Arizona.

Hart B, Smith JE. Forest fire fuel reduction treatments and the impacts on fungi in the Blue Mountains of eastern Oregon: A mycorrhizal perspective. Western Forestry Graduate Research Symposium, April 21-22, 2014, Richardson Hall, OSU, Corvallis, Oregon.

Hart B, Smith JE. Forest fire fuel reduction treatments and the impacts on fungi in the Blue Mountains of eastern Oregon: A mycorrhizal perspective. Central Oregon Fire Science Symposium. April 7-11, 2014. Bend, Oregon.

8. Workshop materials and outcome reports

9. Field demonstration/tour summaries

10. Website development

11. Presentations/webinars/other outreach/science delivery materials

Two Oregon State University undergraduates and two graduate students, and one volunteer gained job experience and learned about restoration of Blue Mountain, OR forests as a result of participation in the project.

Exchange of information related to fire ecology in dry forest ecosystems occurred on numerous occasions with international scholars from the University of Valladolid, Spain: Dr. Maria Hernandez (2015-2016), Dr. Pablo Martin Pinto (2016-2017).

Smith JE, Hart BTN, Luoma D.L. Webinar planned for March 2018. Long term effects of fuels reductions treatments on soil nutrients and ectomycorrhizal communities of ponderosa pine in the Blue Mountains of eastern Oregon. Corvallis, Oregon.

Hart BTN. Fuel treatments of ponderosa pine (Pinus ponderosa) in the Blue Mountains of eastern Oregon: A Mycorrhiza Perspective. April 20, 2017. Thesis presentation. Richardson Hall, Oregon State University, Corvallis, Oregon.

Smith JE. Forest Fires and Fungi. April 3, 2017. TAP Talk at Block 15 Brewery, Corvallis, Oregon.

Smith JE, Reazin C, Morris S, Cowan AD, Jumpponen A. Fires of differing intensities rapidly select soil fungal communities in a Northwest US ponderosa pine forest ecosystem. October 3, 2016. Departmental Seminar, University of Valladolid, Palencia, Spain.

Hart BT, Smith JE, Luoma DL. Fire in the Future, Lessons from the Past: Forest Fire Fuel Reduction Treatment Impacts on Ponderosa Pine Mycorrhizal Fungi Diversity. Mycological Society of America Meeting, August 8-12, 2016, Berkeley, CA.

Hart B, Smith JE, Luoma D. Forest fire fuel reduction treatment impacts on ponderosa pine mycorrhizal fungi diversity. Northwest Scientific Association 87th Annual Meeting, March 23-26, 2016, Central Oregon Community College, Bend, Oregon.

Hart B, Smith JE, Luoma D. Fire in the Future, Lessons from the Past: Forest Fire Fuel Reduction Treatment Impacts on Ponderosa Pine Mycorrhizal Fungi Diversity. 6th International Fire Ecology & Management Congress, November 16-20, 2015, San Antonio, Texas.

Smith JE. Forestry events and issues shape mycological research in the Pacific Northwestern United States. International Symposium on Forest Mushrooms, August 6, 2013. Seoul, Korea.

Appendix C: Metadata

Data collected for this report includes soil biogeochemical data, litter depth data, and ectomycorrhizal species richness and biomass data. All data is stored as .xlsx and .docx, and can be accessed from the US Forest Service Research Data Archive.