LONG-TERM VEGETATION RESPONSE FOLLOWING POST-FIRE STRAW MULCHING

A Thesis

Presented in Partial Fulfillment of the Requirements for the

Degree of Master of Science

with a

Major in Natural Resources

in the

College of Graduate Studies

University of Idaho

by

Jonathan David Bontrager

Major Professor: Penelope Morgan, Ph.D.

Committee Members: Andrew Hudak Ph.D., Peter Robichaud, Ph.D., Eva Strand, Ph.D.

Department Administrator: Randy Brooks, Ph.D.

May 2017

AUTHORIZATION TO SUBMIT THESIS

This thesis of Jonathan David Bontrager, submitted for the degree of Master of Science with a major in Natural Resources and titled "Long-Term Vegetation Response Following Post-Fire Straw Mulching," has been reviewed in final form. Permission, as indicated by the signatures and dates below, is now granted to submit final copies to the College of Graduate Studies for approval.

Major Professor____Date_______________
Penelope Morgan, Ph.D.

Committee
Members ____Date_______________
Andrew Hudak, Ph.D.

____Date_______________
Peter Robichaud, Ph.D.

____Date_______________
Eva Strand, Ph.D.

Department
Administrator ____Date_______________
Randy Brooks, Ph.D.

ABSTRACT

Although mulching is one of the most common post-fire treatments to reduce soil erosion potential, little is known about the long-term effects (over 10 years) on vegetation response. We measured understory plant species diversity and abundance, tree seedling density by species, as well as a remotely-sensed change detection algorithm called LandTrendr, to assess differences measured on mulched and unmulched paired plots on six large forest fires. On mulched plots, tree seedlings grew taller faster, especially on north-facing aspects, and there was 2% more graminoid cover in the vegetation component. While mulch did not affect density in tree seedling density ponderosa pine was favored on north-facing slopes, while Douglas-fir had higher tree seedling density on south-facing slopes. There are many concerns about using straw mulch, our study suggests the long-term effects are subtle and do not suggest a change in vegetation trajectories.

Keywords: BAER, LandTrendr, mulch, post-fire recovery

ACKNOWLEDGEMENTS

This project is a collaborative effort of the University of Idaho's Department of Forest, Rangeland, and Fire Sciences and the USDA Forest Service, Rocky Mountain Research Station.

I thank committee members Penny Morgan, Andy Hudak, Pete Robichaud, and Eva Strand for your guidance, patience, and support. I thank the Rocky Mountain Research Station employees, especially Ben Bright and John Byrne

A special thanks to the many people who helped with fieldwork: M Schmidt, K Schmidt, C Minerich, B Brown, B Newingham, D Truitt, J Byrne, C Littlefield, L Vantieghem, E Berman, J Johnson, and D Hammond. We would also thank the Pike-San Isabel, Okanogan-Wenatchee, Malheur, Idaho Panhandle, and Boise National Forests for providing mulch locations and logistical support, as well as S Larson, formerly Rocky Mountain Research Station, and S Gunderson, formerly of Revegetation Services, Inc. for mulching data. Funding was provided by Joint Fire Science Program (Project # 14-1-02-27) through Joint Venture Agreement between the USDA Forest Service Rocky Mountain Research Station and the University of Idaho (Agreement # 14-JV-11221633-112).

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LONG-TERM VEGETATION RESPONSE FOLLOWING POST-FIRE STRAW MULCHING

Jonathan D. Bontrager[1], Penelope Morgan[1], Andrew T. Hudak[2], Peter R. Robichaud[2]

[1]University of Idaho, Department of Forest, Rangeland, and Fire Sciences, 875 Perimeter Drive MS 1133, Moscow, Idaho 83844, USA

[2]Rocky Mountain Research Station, USDA Forest Service, 1221 South Main Street, Moscow, Idaho 83843, USA

Abstract

As high severity fires continue to increase so does the need to mitigate post-fire effects on downstream values at risk. Although mulching is one of the most common post-fire treatments to reduce soil erosion potential, little is known about the long-term effects (over 10 years) on vegetation response. We assessed the differences in plant understory diversity abundance, fractional cover, tree seedling density, and tree seedling height growth between mulched and unmulched areas on six large forest fires. We chose six fires in the Interior West, spanning two forest types that were mulched between nine and 13 years ago. We measured understory plant species diversity and abundance, tree seedling density by species, as well as a remotely-sensed change detection algorithm called LandTrendr, to assess differences measured on mulched and unmulched paired plots on six large forest fires. Mulch did not influence understory plant diversity, species richness, or fractional cover. On mulched plots, tree seedlings grew taller faster, especially on north-facing aspects, and there was 2% more graminoid cover in the vegetation component. While mulch did not affect density in tree seedling density ponderosa pine was favored on north-facing slopes, while Douglas-fir had higher tree seedling density on south-facing slopes. Ponderosa pine was favored on north-facing slopes, while Douglas-fir had higher density on south-facing slopes. Our study helps to understand long-term vegetation recovery after a severe wildfire, especially how altering immediate post-fire conditions can have lasting effects across the landscape. Managers will be able to weigh the long term implications of mulching against the short-term reductions in soil erosion potential. While there are many concerns about vegetation suppression and exotic species introduction from using straw mulch, our study suggests the long-term effects are relatively subtle and do not suggest a change in vegetation trajectories in the first ten years post-fire.

Keywords: BAER, LandTrendr, mulch, post-fire recovery

1. Introduction

Although mulching is one of the most common post-fire treatments to reduce soil erosion potential, little is known about the long-term effects (over 10 years) on vegetation response. Large wildfires have been increasing in both size and frequency in recent decades (Dennison et al. 2014, Westerling et al. 2006, Westerling 2016). Mulching and other post-fire rehabilitation techniques are often used to stabilize soil, protect values, and promote vegetation recovery in areas burned with high severity by providing immediate ground cover (Robichaud and Ashmun 2013, Bautista et al. 2009, Beyers 2004). By providing a physical barrier, mulching increases soil moisture, decreases soil temperature, and alters nutrient availability (Berryman et al. 2014). These factors may influence on recovery of plant communities (Morgan et al. 2014, Facelli and Picket 1991). Mulch has both positive and negative effects on plant species diversity, total cover, and tree regeneration (Dodson and Peterson 2010). These vegetation differences in the first few years after a disturbance could have lasting effects on an ecosystem (Morgan et al. 2014, 2015). Even though mulching is one of the most common post-fire treatment methods, we know little about the long-term effects (over 10 years) on vegetation trajectories (Morgan et al. 2014).

Agricultural straw mulch is used post-fire to reduce soil erosion potential on steep slopes where values are at risk. After a wildfire, the loss of vegetation biomass can greatly reduce the stability of soil (Robichaud 2005, Wagenbrenner et al. 2006). In addition, high severity fires also affect soil properties which in turn can affect the potential for soil erosion, loss of nutrients, and water repellency (Neary et al. 1999). Mulch greatly decreases soil movement (Wagenbrenner et al. 2006) by providing an organic, physical barrier to the topsoil until vegetation can establish in abundance (Dodson and Peterson 2010, Robichaud et al. 2013). Mulching is one of the most effective emergency stabilization techniques to use post-fire (Robichaud et al. 2000, Bautista et al. 2009). It has repeatedly been shown to stabilize soil, reduce sediment flow, prevent loss of soil productivity, and reduce risk of flooding (Bautista et al. 1996, Dean 2001, Robichaud et al. 2000, Williams et al. 2014). Due to expense, mulching is used as a very targeted and strategic treatment of areas of high soil erosion potential and risk of loss of downstream values (Bautista et al. 2009, Williams et al. 2014).

Mulch alters local habitats, sometimes enough that some plant species may not be able to establish after a fire. Some species such as ponderosa pine (*Pinus ponderosa*) require bare soil to germinate (Curtis et al. 1965). The physical barrier caused by mulching forces plants to expend extra energy to push through, and prevents new seeds from hitting bare ground (Facelli and Prickett 1991). Mulching also lowers soil temperature by blocking solar radiation, and acts as an insulator at night to prevent freezing (Facelli and Prickett 1991). This may result in a longer growing season for mulched areas compared to unmulched areas (Facelli and Prickett 1991). The lowered temperature and reduced solar radiation mulch provides also reduces evaporation which increases soil moisture (Mulumba and Lal 2008). These effects combine to increase available nitrogen available for developing plants, and microbial activity (Berryman et al. 2014).

Many short-term studies of vegetation response to post-fire mulching have been done, with mixed results. In examining only mulched sites on the 2005 Tripod Fire (Washington), Dodson and Peterson (2010) found plant cover, species richness, and tree seedling density were all positively associated with cover of straw mulch in a dry-mixed conifer forest. However, when mulch reached over 70% cover, it began to have a negative effect on vegetation (Dodson and Peterson 2010). In the ponderosa pine forests burned in the 2000 Bobcat Fire in Colorado, Wagenbrenner et al. (2006) found reduced sediment movement, increased ground cover by the mulch, and more vegetation cover on mulched compared to unmulched plots. Conversely, others have found that mulching can inhibit plant establishment and introduce non-native species (Beyers 2004, Kruse et al. 2004, Robichaud 2005). Kruse et al. (2004) found that two years post-fire, mulched areas had a higher occurrence of non-native species, less overall vegetation cover, and reduced conifer tree seedling density when compared to similar unmulched areas. If these differences persist, mulching very well could be a tradeoff between short-term reduction of soil erosion potential and altered vegetation recovery for many years post-fire.

Initial vegetation response can have lasting influences on vegetation trajectories. The first species to colonize post-fire will often persist far into the future (10 years, Abella and Fornwalt 2015; 29 years, Engel and Abella 2011). Areas burned with high severity, which make up a majority of all wildfire area mulched, have reduced ecosystem resistance to fire and reduced resiliency, a diminished ability to return to the pre-fire state (Abella and Fornwalt 2014). Of particular concern is that high severity wildfire could be causing large areas to be converted from forest to shrub or grass communities (Savage and Mast 2005, Johnstone et al. 2016). By altering the physical characteristics of a site's microclimate, it is possible that mulching could change which species will be able to colonize first and persist.

LandTrendr (Landsat-based Detection of Trends in Disturbance and Recovery) offers a useful way to compare vegetation trajectories over time and space. Kennedy et al. (2010) developed this algorithm to analyze a time series of Landsat images on an annual, pixel-by-pixel basis. The algorithm is sensitive enough to capture short-term disturbance, while reducing intra-annual variance to clarify trends over the long-term record (Kennedy et al. 2010). This method has been previously used to estimate live versus dead basal area in forests with bark beetle-induced tree mortality (Bright et al. 2014), monitor multiple forest disturbances when combined with Forest Inventory and Analysis plots (Schroeder et al. 2012), estimate CO_2 flux (Smithwick et al. 2013), as well as other applications. Thus far, LandTrendr has not been applied to chart the vegetation trajectories on individual fires, or to compare vegetation trajectories in contrasting conditions across the landscape, yet it has strong potential to the degree to which initial conditions alter vegetation trajectories and if early differences persist 10 or more years.

Research on ecological effects of mulching have not kept up with its prevalent use as a post-fire treatment. Studies conducted on vegetation response from mulching have been short-term (usually less than three years), on single fires, and have yielded mixed results (Dodson and Peterson 2010, Morgan et al. 2015). This study will help us to understand vegetation recovery trajectories, including if, where and how altering immediate post-fire conditions by mulching results in different vegetation composition and plant species diversity across the landscape. This is part of a larger study to better understand vegetation trajectories after fire across the western United States by combining field data with vegetation trajectories inferred using LandTrendr (Hudak et al. 2014). Together with the larger project, we offer a unique perspective as we combine long-term (~10 years) data from multiple fires in multiple forest types with remotely-sensed data across the interior western United States. Managers will be able to assess the long-term implications of mulching and better weigh potential impacts on post-fire vegetation against the benefits of reducing soil erosion potential.

1.1 *Research objectives*

We assessed the differences in understory vegetation and tree seedling differences between mulched and unmulched areas on multiple large fires sampled in the field, and for annual vegetation trajectories interpreted from satellite imagery. Specifically, our hypotheses were:

1) Understory plant species richness and diversity was higher on mulched sites,

2) Tree seedling density was lower on mulched sites, but tree seedling height growth per year is higher, and

3) Mulching increases the rate of long-term vegetation recovery, as inferred using LandTrendr.

2. **Methods**

2.1 *Study Areas*

We focused on older (nine or more years) wildfires in the Interior Western United States where a minimum of 40 hectares of straw mulch was aerially applied in steep areas burned with high severity (Figure 1). In order to stratify across forest types, fires were chosen from two dry forest types (ponderosa pine-dominated and dry mixed conifer), as described by LANDFIRE existing vegetation type (LANDFIRE 2008), in which mulching treatments are often applied (Table 1). We used the Burned Area Emergency Response (BAER) database to select fires where mulch had been applied nine to 13 years prior (Robichaud 2017). Information on post-fire actions that have been recommended, requested, and completed by BAER teams are all contained in the BAER database. We then confirmed the location of treated areas with local forest managers. We used Monitoring Trends in Burn Severity (MTBS) maps for areas burned with high severity (MTBS 2016). MTBS analysts use the differenced normalized burn ratio (dNBR) and expert opinion to define burned areas into three severity classes: low, moderate, and high severity. NBR is a remotely-sensed vegetation index that is sensitive to vegetation and soil reflectance within a given pixel, (Landsat bands 4 - band 7) / (band4 + band 7) (Key and Benson 2005). On the ground, areas burned with high severity usually correspond to greater than 70% basal area removed (Agee 1993); those areas burned with high soil burn severity also have loss of ground cover, surface discoloration due to ash or oxidation, loss of soil structure, consumption of fine roots, and possible formation of water repellent layer (Parson et al. 2010).

2.2 *Plot setup*

We collected data from five fires in the summer of 2015, and from the Cascade Fire in summer 2016 (Table 1). We sampled 58 plots (29 plot pairs). To ensure that plot pairs represented a range of conditions on each fire, four strata, including elevation (high and low) and transformed aspect (high and low), were sampled. Not all four strata were sampled on all fires. Number of plots within each fire varied by number and size of mulch units and accessibility by road (Table 1).

Plot centers were randomly located within the strata. To avoid edge effects, mulched and unmulched plots were located at least 60 m from a patch boundary. Patch boundaries included edge of strata class borders, as well as roads and trails. Additionally, plots were usually within 0.5 km from roads for ease of access and time constraints.

Within each plot there were five subplots (Figure 2). The arrangement of the plot was similar to that described by Morgan et al. (2015). One subplot was placed at the center of each plot, with four other subplots located 30 m away (Figure 2). The first peripheral subplot was placed directly upslope with the others at azimuths of 90, 180, and 270 degrees from upslope.

2.3 *Understory plant measurements*

At each subplot, we measured ocular fractional cover and percent canopy cover by species within a 1-m^2 quadrat. The categories for fractional cover were green vegetation, non-photosynthetic vegetation (NPV), soil, and rock. NPV includes woody debris and other dead plant material. Cover components were recorded as a percentage of area occupied as viewed from above, therefore fractional cover for these classes in each subplot added up to 100%. Also, we estimated the percent canopy cover of every plant by species. For many subplots, the sum of percent canopy cover by all species exceeded 100%, as there was often multiple layers of vegetation. Plant species not identified in the field were given a unique unknown code, collected, and later identified with help of the experts at the University of Idaho Stillinger Herbarium.

2.4 *Tree seedling measurements*

Tree seedling density by species was measured within a 5.6-m radius circle from the center of the subplot (Figure 2). This circle was divided into quarters. Tree seedling counts started in a randomly selected quarter and continued in additional whole quarters until six of the most dominant species had been measured. To avoid measuring tree seedlings from the same clump, especially in areas of high density, measurements were spread throughout the quarter(s) being measured and counted. Total height and distance between terminal bud scars (the resulting scar after terminal bud scales fall off) were measured to estimate yearly height growth (Urza and Silbold 2013). Seedlings without nodes (i.e., less than one year old) were not counted. Every seedling older than one year within the measured quarter(s) was counted; counts were converted to density for a given subplot using the recorded area sampled on one or more quarters. Tree seedling counts were then converted to stems/hectare (ha).

2.5 *Analyses*

Measured variables were averaged for the five subplots in each plot. All statistical analyses were done in R version 3.3.1 (R Core Team 2016). The same fixed terms were used for all MANOVA and linear mixed models: differenced normalized burn ratio, elevation, transformed aspect, treatment, and fire. Differenced normalized burn ratio (dNBR) is the change in the normalized burn ratio caused by a fire. Transformed aspect (trasp) is a cosine transformation of aspect from degrees to a continuous variable, where 0 represents 30 degrees north-northeast-facing (NNE) and 1 is 210 degrees, a south-southwest-facing (SSW) aspect. Thus, plots with low trasp are typically cooler and wetter, and those with high trasp are hotter and drier. Treatment is a binary factor, mulched or unmulched. The 'Fire' variable is a factor that encompasses a wide variety of climatic and soil variables that seedlings and understory plants of a particular fire would share but are understandably different across all fires. The random effect, or grouping variable, was 'Pair'. This allowed for comparing mulched and unmulched plots in a pair across multiple environmental conditions in a way that allowed us to focus on the differences caused by mulch. In our mixed modeling framework, predictor variables would have been removed if the pairwise correlation coefficient of 0.7 or higher (Dormann et al. 2012).

2.5.1 *Understory Plant Species Diversity and Richness*

The *vegan* package (Oksanen et al. 2017) was used to calculate both Shannon-Weiner diversity index and species richness for each plot. The total number of species from all five subplots (richness) were used together along with percent canopy cover to calculate diversity at the plot level. A paired t-test was used to analyze for differences in diversity and richness between plot pairs (Zuur et al. 2009).

2.5.2 *Plant Growth Form*

Each plant was assigned to a growth form (shrubs, forbs, graminoid or other) based on the USDA plant database (NRSC 2017). In cases where a species had multiple growth types one was assigned based on the most common form. Cover for each growth form was the total for the individual species in that growth form for each subplot; then the subplot totals were averaged to obtain cover for each growth form on each plot.

A multivariate analysis of variance (MANOVA) was used on the three growth forms being tested. We analyzed the *difference* between the mulched and unmulched plot pairs in order to take advantage of the paired data in this analysis. The cover values for each growth form on a mulched plot was subtracted from the value for the unmulched plot of the same pair. Where the overall mulching treatment was significant ($\alpha \leq 0.1$) we then examined differences for each growth form using a linear mixed effects model. An $\alpha \leq 0.1$ was chosen as a first pass, as a MANOVA test is not an ideal way to look in-depth at paired data, whereas a mixed effects framework is more adept.

Mixed effects model selection was based on an improved Akaike information criterion (*AIC*) score, when shown significantly different from a more complex model with an ANOVA test (Zuur et al. 2009). The base model started with all variables, interactions between mulch treatment and other fixed variables, and then non-significant fixed variables were dropped sequentially. To conform to linear mixed effects assumptions, unequal variance and spatial autocorrelation were accounted for in the model structure using R package *nlme* (Pinheiro et al. 2016) as opposed to any transformations (O'Hara 2010). This facilitated easier interpretations of model coefficients.

2.5.3 *Fractional Cover*

Cover values from each subplot were averaged to the plot level. A MANOVA test was initially used on the difference in mean fractional cover estimates of vegetation, NPV, soil, and rock. Only if a cover material showed significance ($\alpha \leq 0.1$) would an individual variable be investigated further with a mixed effects model. Model selection was again done in a mixed effects framework and proceeded similarly to the growth form analysis.

2.5.4 *Tree Seedling Density and Height Growth*

For each plot, the tree seedling density (stems/ha) for each species in each subplot were aggregated to total count for the plot and converted to stems/ ha by dividing by total area sampled If one of the five sub-plot had zero seedlings, but seedlings were present in the other subplots, it was still included as a plot with seedlings. This was done for total tree seedlings, as well as for each tree seedling species on a given plot.

Because some plots did not have any tree seedlings present, a two-step modeling process was used to ask two questions. First, does mulch influence whether or not there will be seedlings in an area? Second, *where seedlings are present* does mulch influence the density or play a role in which tree species are regenerating? To answer the first question, density at a given plot was converted to binary presence/absence data. This was then analyzed in a mixed effects modeling framework similar to that used in growth form and fractional analyses. dNBR, elevation, trasp, treatment, and fire were used as predictor variables for whether or not seedlings were present on a plot. On plots with seedlings present, seedling density was calculated in stems/ ha. Using these values, we compared average seedling density on mulched and unmulched plot pairs. We used a mixed effects modeling framework with the same treatment and physical predictor variables as before.

To examine mulch influences at the tree species level, two analyses were done: one for ponderosa pine, and one for Douglas-fir (*Pseudotsuga menziesii*). Zeros were also removed so as not to skew model results, but to ask the focused question: where there are seedlings, is mulch having an influence on density? These two species were chosen because they were by far the most common species found throughout the study, representing enough data for a mixed effect model. The composition models were run on all plots that had ponderosa pine, and then in plots that had Douglas-fir seedlings present, regardless if the corresponding paired plot had seedlings or not.

Tree height analysis was more complex, as there were variable tree heights at any given plot, with a different combination of species and ages. We used a mixed effect model initially as opposed to testing with a MANOVA first. In addition to the fixed variables mentioned above, seedling age and species were also included in this model. Age was estimated from the total number of terminal bud scars counted along the individual seedling (Urza and Silbold 2013). Only trees that established within the first three years after the fire were used in the model to address the question of how mulch affects tree growth. Most of the mulch likely dissipated after three years.

Due to the large amount of possible combinations (Treatment, Species, and Fire having multiple levels) it was not possible to run every interaction all at once. Instead, a stepwise process was used, including one interaction combination each time. Initially 'Treatment' was tested interacting with every other variable, and this produced a better model when combined with trasp. Next, tree species variable was tested with interactions of other variables, as the growth rates could vary between species. In the end, both the interaction between seedling treatment to trasp, and age to species were modeled together.

2.5.6 *LandTrendr and vegetation trajectories*

Landsat series images from 1984 to 2012 were processed using LandTrendr (Kennedy et al. 2010). For each pixel where a field observation was collected annual NBR, and recovery magnitude were extracted. Normalized recovery magnitude (RMag) as defined by LandTrendr as the increase in NBR from disturbance until 2012, the last year for which we have values on these plots (Kennedy et al. 2010), was divided by the number of years since fire (as of 2012) to calculate the rate of recovery, resulting in recovery rate metric (RRate. While the yearly NBR values are useful for illustrating the disturbance and recovery, a single value such as RRate is easier to test statistically and to interpret relative to our field data. Model selection was again done in a mixed effects framework and proceeded similarly to the growth form analysis. dNBR, elevation, trasp, 'Fire' and 'Treatment' were all used as possible predictor variables, 'Treatment' was tested for interactions with all other variables as well. A LandTendr product for pre-disturbance NBR was also included as a possible predictor variable. We also grouped each trajectory pair into one of six post-fire recovery categories: 1) mulched and unmulched had similar post-fire NBR values, 2) mulched plots appeared to be recovering at a faster rate initially, but at the end the unmulched plot recovered both at a higher rate and to a higher value, 3) mulch seemed to suppress recovery for the first three years, but then the mulched plot eventually recovered to a higher point than the unmulched, 4) mulch always showed a higher NBR recovery, 5) unmulched always showed a higher NBR recovery, and 6) 'not a true pair'; this was caused by LandTrendr error, different pre-fire conditions, or different NBR burn severity. A difference in prefire conditions is determined by non-overlapping error bars for a pair the previous five years before a fire, whereas different burn severity is non-overlapping error bars the year of fire. Because each plot only represented five pixels, the error bars are twice the standard error (2SE). By only using LandTrendr values from plots we visited on the ground we are able to make comparisons of possible differences to collected vegetation data, as well as guarantee these areas were actually mulched.

3. Results

3.1 *Diversity and Species Richness*

A total of 352 species were sampled across the six fires, with 247 on mulched plots, and 248 on unmulched plots. Neither plant species richness (P = 0.4396, t = -0.7841, df = 28) nor diversity (P = 0.7479, t = -0.3246, df = 28) differed for mulched and unmulched plot pairs. Species richness ranged from 8 to 37 species on mulched plots (median of 22), and 9 to 33 on unmulched plots (median of 21). Plant species diversity ranged from 1.9 to 3.4 on mulched plots (median of 2.9), and 2.0 to 3.3 on unmulched plots (median of 2.8). While diversity varied among fires, there was no discernable pattern across all fires (Figure 3), or by average summer precipitation (Table 1).

3.2 *Plant Growth Form Cover*

Graminoid canopy cover was slightly higher on mulched than on unmulched plots (P = 0.004, MANOVA), but cover of forbs and shrubs did not differ between mulched and unmulched plot pairs across the six fires (P = 0.209 for forbs and P = 0.144 for shrubs). The best-fitting mixed effects model predicting graminoid cover had only treatment and fire as predictor variables (Table 2), with mulched plots having slightly higher graminoid cover than unmulched plots (Figure 4). Including physical variables made the model much worse, most likely because the differences in trasp, elevation, or dNBR between plot pairs were too small to influence graminoid cover. Interactions between mulch and these physical variables also did not improve the model. However, a semi-variogram showed high levels of spatial autocorrelation within 4 km, which indicates plots of the same fire are correlated. Fitting the model with a correlation structure addressed this, but did not alter the model *AIC*, coefficients, or P values significantly (Table 2).

3.3 *Fractional Cover*

Fractional cover values varied greatly between pairs, and between fires (Figure 4). The Ricco Fire had the most vegetation (ranging from 60-74%, median of 68%) and the Hayman Fire had the least (ranging from 27-49%, median of 37%). Percent bare soil also varied by fire, ranging widely within the study, from 0% (median of 2%, high of 6%) on an unmulched plot on the Ricco Fire, to 43% (median 29%, low 16%) on an unmulched plot on the Hayman Fire. Overall, the Hayman had the most bare soil cover at 16% to 43%, all others were below 16%.

While mulching treatment had no influence on green vegetation, NPV, or rock cover (P = 0.405 P = 0.436, and P= 0.903, respectively using the MANOVA test), there was more bare soil on mulched than unmulched plot pairs (*P* = 0.058, MANOVA). The final soil cover model had mulch as non-significant (*P* = 0.732) once unequal variance and spatial autocorrelation were accounted for. Adding the treatment variable increased *AIC* score slightly from 424.6, to 424.8. The final model contained only elevation and aspect, both very significant (*P* < 0.001) with positive coefficients (Table 2). As elevation increased, so did bare soil, and as aspect became more SSW, bare soil increased regardless of whether or not a plot was mulched (Table 2).

Mulch was present but with very low or zero cover on plots sampled. When mulch was encountered in a subplot it was treated as NPV for fractional cover. This happened on three sub-plots on the Tripod Fire. Mulch was seen on all fires except for the Ricco Fire, however it often took determined effort to find it so many years after these areas were mulched.

3.4 *Tree Seedling Density and Height Growth*

Density of tree seedlings by species varied greatly within and among fires (Figure 5). As an example, density on the Ricco Fire ranged from 0 to 102 stems per hectare (median of 37), with only ponderosa pine present. In contrast, on the Myrtle Creek Fire, tree seedling density ranged from 0 to 23,000 stems/ ha (median of 6535) with ponderosa pine, Douglas-fir, Engelmann spruce (*Picea engelmannii*), western larch (*Larix occidentalis*), and lodgepole (*Pinus contorta*) all present (Figure 5). On 14 of the 58 plots, no tree seedlings were present: nine mulched plots and five unmulched plots. Of these, four plot pairs had no seedlings present, accounting for eight of the 14 plots without seedlings. The Hayman, Ricco, and Myrtle Creek Fires all had plots with no tree seedlings present.

Mulch had no effect on whether or not tree seedlings were present in a plot using a binary 0/1 model (*P* = 0.216 for the mulched variable, *AIC* increased by adding it to the model). Nor did mulch influence tree density (*P* = 0.645) where seedlings were present. However, species composition of seedlings varied with the interaction of mulch and aspect for density of both ponderosa pine and Douglas-fir. Where Douglas-fir seedlings were present, mulch was significant in the mixed effects model (*P* = 0.004), and in the ponderosa pine model, mulch was also significant (*P* = 0.015) (Table 2). By itself, mulch decreased density of both ponderosa and Douglas-fir, however, for each species there was an interaction effect between mulch and aspect (Table 2). For ponderosa pine on unmulched plots, seedling density increased as aspect became more SSW. However the interaction effect between mulch and aspect was negative, meaning as aspect moved from NNE to SSW in mulched plots, the density of ponderosa pine decreased (Table 2). Douglas-fir showed the opposite trend. The interaction effect of mulch and increasing aspect (becoming more SSW) was enough to overcome the negative coefficients of mulch and trasp separately, and have a net gain of Douglas-fir in mulched plots on drier aspects (Table 2). Douglas-fir and ponderosa pine were each found on just under half of the plots.

Tree seedling height varied with tree seedling age (as indicated by number of terminal bud scars), species, dNBR, trasp, and whether the plot was mulched. The interactions between age and species, and between trasp and mulching treatment are also included in the model. Tree seedling height increased with increasing burn severity (as captured by dNBR) and increasing trasp (from NNE to SSW). Growth also depended on tree seedling age, as older seedlings grew more in one year (Table 2).

Mean tree seedling height was on average 5.7 cm greater ($P = 0.017$) on mulched than unmulched plots where other variables were accounted for (Table 2). Whereas height growth was positively correlated with both mulch and increasing trasp, the interaction was not statistically significant. Rather, as trasp increased on plots that were mulched, the interaction predicts less height, almost enough to cancel out the influence of mulch all together. The tree seedling height model was improved by including the interaction effect of mulch with trasp. This interaction was statistically significant at $\alpha = 0.1$, but not at $\alpha = 0.05$, however it did significantly lower the model *AIC*, so it was included.

3.5 *LandTrendr and Post-fire Vegetation Trajectories*

Annual post-fire vegetation recovery rate, as inferred from the LandTrendr normalized recovery metric RRate, was influenced by elevation, dNBR and an interaction effect between 'Treatment' and the 'Fire' variable. As dNBR increased, the recovery magnitude rate increased, therefore the more severe a fire was, the greater it recovered from post-fire to 2012 (Table 2). Elevation was negatively correlated with recovery; as elevation increased the recovery decreased (Table 2). The 'Fire' variable was significant by itself, and as an interaction effect with 'Treatment', however 'Treatment' only improved *AIC* as an interaction effect, as a separate variable it increased *AIC*. Adding it as an interaction showed significant model improvement ($P < 0.001$, Table 2). There does not seem to be any pattern in the coefficient of each fire to summer precipitation, however we had only a small number of plot pairs on a given fire and only average annual precipitation, rather than the precipitation in the first few years following mulching, all of which make it difficult to detect a pattern and explain this possible cause of variability in post-fire tree establishment.

Yearly NBR values were plotted by plot pair for visual examination and interpretation. Every plot showed some recovery, however the extent of recovery varied greatly. A plot pair on the Ricco Fire had similar NBR values in 2012 to the pre-fire values for both the mulched and unmulched plots, whereas one plot pair on the Myrtle Creek Fire showed only the slightest upward trend in NBR values.

In the most common pattern of post-fire vegetation trajectories, evident for 41% of plot pairs, the annual NBR of mulched and unmulched plot pairs showed little to no difference from each other since the time of fire to 2012 where the available LandTrendr data stops (Figure 6A). This was consistent with the results from paired plots measured in the field, suggesting that mulch has minimal influence on long-term post-fire vegetation recovery.

On some plot pairs, mulch influenced the post-fire vegetation trajectories through time. The second pattern was that in 3% of plots mulch seemed to suppress recovery in the first few years (Figure 6B). After four years, however, the NBR values of the mulched plot surpassed that of its unmulched paired plot. The third pattern, evident in 3% of our plot pairs, is that vegetation is more abundant on mulched than unmulched plot initially, but after four years post fire, vegetation was more abundant on the plots that had not been mulched (Figure 6C).

There were also pairs where one of the plot pairs recovered faster initially and always had a higher NBR (Figure 6D). On 7% of our plots, mulch seemed to suppress initial recovery, and did not catch up to the unmulched plots by 2012, suggesting a long-term difference in vegetation recovery.
On 10% of our plot pairs, mulched plots recovered faster initially, and the unmulched plots did not catch up to the mulched plots for the duration of the LandTrendr algorithm (Figure 6E). On these plots, vegetation recovery was enhanced slightly by mulching.

Due to non-overlapping pre-fire NBR values the remaining 36% of plots were not classified into one of the above patterns.

4. Discussion

4.1 *Mulch had Minimal Effects on Understory Vegetation Response 9 to 13 Years Post-Fire*

Our results suggest that any initial effects of straw mulch on understory vegetation are not long-term, ecosystem-altering effects. Nine to 13 years post-fire we saw few statistically significant differences in species richness, diversity, canopy cover or fractional cover between mulched and unmulched plot pairs. The one statistically significant difference we did find, slightly increased cover of graminoids by about 2% in mulched plots (Table 2), is not enough to be ecologically significant. Post-fire mulching with agricultural straw has numerous, but often conflicting short-term influences on vegetation recovery. Mulching has been linked to decreased plant diversity and suppression of native plants (Kruse et al. 2004) by providing a physical barrier on the forest floor, as well as introducing non-native species (Dodson and Peterson 2010). Conversely, mulching has also been shown to increase plant diversity and tree seedling growth (Dodson and Peterson 2010), increase vegetation cover post-fire, and most importantly reduce soil erosion potential (Wagenbrenner et al. 2006).

All plots had less than 50% bare soil total coverage. Pannkuk and Robichaud (2003) described this as a tipping point of when large soil erosion events can happen post-fire. Thus, mulched or not, vegetation on all the plots recovered enough so that soil erosion potential is low. Unfortunately, we cannot infer from the Landtrendr trajectories how long it took for vegetation to reach this threshold, nor how long post-fire the mulch was present.

Introduced weeds are a significant concern of managers and ecologists alike. Although analyzing non-native plants was not a main objective of this study, we found very few non-native plants and none were abundant. While it is now common practice to use certified weed free straw in post-fire mulching, it is unlikely that agricultural straw used is ever truly ‘weed free’ (Robichaud et al. 2000). The Hayman Fire is infamous for using cheatgrass (*Bromus tectorum)*-contaminated straw for a majority of the mulching operation (Robichaud et al. 2003). We found cheatgrass in nine of 14 plots we sampled on the Hayman Fire; two were not mulched while seven were mulched. However cheatgrass density across these plots was low, averaging just 1% on the unmulched plots and under 3% on the mulched plots. It is also possible in the 13 years between mulching with cheatgrass-tainted straw and sampling, that cheatgrass moved from mulched to unmulched plots, as we attempted to have plot pairs as close as possible within the same physical conditions. When Dodson and Peterson (2010) sampled the Tripod Fire during the second growing season after the fire, they found 14 non-native plant species in mulched areas, five with occurrences over 50 (an occurrence in their study being one individual found in a sampling unit). In contrast, our assessment of the Tripod nine growing seasons after mulching, only one of these weeds, the common dandelion (*Taraxacum officinale*) was present on five plots (three unmulched and two mulched), with the highest canopy cover at 3%. Although our study design and plot locations were different, it is likely we would have detected other non-native species if they were still present and so abundant. Further investigation is warranted to assess the long-term implications of weed introduction in mulched areas. It is also possible these non-native species were in the area before the fire, however to our knowledge there is no pre-fire vegetation data. . However, these results from the Hayman and Tripod Fires suggest that while non-native species may have been a concern initially, they did not persist in large abundances across the landscape.

4.2 *Mulch Resulted in Increased Tree Height Growth and Altered Species Composition*

Although mulch has no influence on overall tree seedling density, the differences in tree species composition suggests the relative importance of mulch as a barrier helping to retain soil moisture. In plots where Douglas-fir trees naturally regenerated, seedlings were at a higher density in mulched plots on the SSW aspect. In plots where ponderosa pine established, there were fewer in mulched plots than in similar plots that were not mulched. Perhaps this is because ponderosa pine requires bare mineral soil to establish (Curtis et al. 1965) and were thus less likely to establish until the straw mulch decomposed. Douglas-fir can establish with some organic matter present over bare mineral soil (Herman and Lavender 1990), so we think the greater soil moisture holding capacity in mulched areas is more important to Douglas-fir. This could explain why more Douglas-fir than ponderosa pine seedlings were found on the drier aspects.

However, just because Douglas-fir is able to establish at higher densities than ponderosa pine on mulched plots on SSW aspects does not mean they are better off than their counterparts growing on opposite facing slopes. Generally, northern aspects of a given area are more productive than southern aspects (Stage and Salas 2007). While all seedlings grew taller on mulched plots than on those not mulched, the height difference on the SSW aspects is less than as on the NNE aspects (Table 2).

Mulch influences on tree seedling species composition and height growth, while subtle, are potentially long-lasting. By decreasing soil temperature and increasing soil moisture, mulch created an environment where Douglas-fir were able to establish where they otherwise may not have been able to grow. Trees that grow faster initially are most likely to continue in this way (Mattsson 1997). If the seedlings that established on areas that were mulched continue to grow taller than their counterparts on similar areas without mulch, this could possibly provide a distinct advantage in surviving the next fire if those future saplings also have thicker bark and canopy base heights well above the flames when fires occur (Harper 1977.

4.3 *LandTrendr Vegetation Trajectories Show Little Contrasts for Mulched and Unmulched Areas*

Perhaps the most important potential effect of mulching is the altering of vegetation trajectories. As a management technique, mulch has been shown to greatly reduce soil erosion potential, however there is potential for an ecosystem shift caused by microsite alterations. Mulching applications alone did not impact vegetation trajectories. To complement our statistical analysis of vegetation sampled in th field on paired plots, we identified multiple different vegetation trajectory patterns based upon LandTrendr annual NBR, which offers visual interpretation of LandTrendr results (Figure 6). From these we can infer potential reasons for the differences observed. Each of these patterns appear on multiple plot pairs in both forest types, on all aspects, and all fires, and with no discernable pattern based on summer precipitation. This suggests that other variables, such as pre-fire vegetation, also influence vegetation trajectories (Engel and Abella 2011).

On 3% of plot pairs NBR was lower on the mulched plot and higher on the umulched, but after a few years they switched and the mulched plot had the higher NBR value. Kruse et al. (2004) noted how straw mulching suppressed establishment of tree seedlings and native species in the first year after mulching. This could be due to the barrier effect that mulch has when it reduces the available area where plant regeneration can occur on bare soil, thus delaying revegetation of the plot. However, after three years, presumably once the mulch was decomposed or blown away, the annual NBR values increased more on mulched than on unmulched pixels (Figure 6B). While it may have slowed down recovery initially, the mulch seems to have set up the plot for rapid vegetation growth after the mulch was gone. This could be an influence of increased tree growth seen on mulched plots, or increased nitrogen and microbial activity in mulched areas (Berryman et al. 2014).

Conversely, on 3% of plot pairs had initial NBR that was higher on the mulched plot and lower on the umulched, but after a few years they switched and the unmulched plot had the higher NBR value. Bautista et al. (1996), Dodson and Peterson (2010) and Wagenbrenner et al. (2006) all noted how mulched areas increased in vegetation on mulched rather than unmulched areas in the three years post-fire that they sampled. Mulch provides a microclimate with cooler soils (Facelli and Prickett 1991) and more moisture (Berryman et al. 2014). While mulch also acts as a barrier, it is likely that many plants species can exploit this moisture enough to grow up through the mulch to show higher NBR values. However, this effect can be short lived, for after the mulch has decomposed or blown away, the unmulched plot recovers faster (Figure 6C). It is also possible that the plants recovering on the mulched plots thrive in the milder microclimate, and once the mulch is removed they do not continue to grow as well, allowing the unmulched plot to catch up.

Monitoring mulched and unmulched areas for only the first two to three years, as done in most studies and management projects, may not be enough to show whether mulch is suppressing or accelerating vegetation trajectories post fire. Over a quarter of our plots showed a switch in recovery rates after two years (see Figure 6). Any initial differences have attenuated nine to 13 years later with the exception of tree seedling composition and height growth. It is very possible that these would not have been seen only a few years after mulching where the seedlings would not be very tall to begin with, and chance of survival for seedlings would be low. Long-term effects of mulching would be most evident on the recruitment of trees, the most long-lived plant growth forms

4.4 *Limitations*

Our analysis of understory vegetation and tree seedlings from plot pairs were just one “snapshot” in time. We offset this by inferring vegetation trends using LandTrendr. Despite the greater sensitivity in detecting treatment differences using paired plots instead of pooled plots, it is possible that the plot pairs differed for reasons other than mulch. This was accounted for as much as possible by the inclusion of physical variables to keep pairs within a single strata and limiting distance between mulched and unmulched plots of a pair.

Inclusion of site specific climatic variables, sampling more paired plot pairs, more paired pixels, and LandTrendr data through year of sampling could all strengthen the conclusion. While we sampled relatively few plots, the paired design allowed us to draw conclusions of the response to mulch in a variety of environmental conditions. This approach helped us conclude that differences in vegetation response was likely due to being treated with mulch or not. We also do not know the depth of mulch on the plots where we sampled. Aerial application can result in uneven distribution of mulch (Lewis and Robichaud 2011, Dodson and Peterson 2010). Our main hypothesis for difference in tree seedling species composition with mulch is moisture related, however we did not assess drought stress or soil moisture.

Using LandTrendr, we found no statistically significant difference in rates of vegetation recovery trajectories due to mulch nine to 13 years post-fire, despite differences in tree seedling composition and height growth we found on plot pairs. This is most likely due to spectral and spatial resolution that was too coarse to see such subtle differences such as tree seedling height and composition between plot pairs, as well as only sampling five pixels as a plot. However this could also be caused by only having LandTrendr products and NBR values until 2012. It is possible that recovery trajectories could have become different between the last year of LandTrendr and our sampling. Other remote sensing tools may be useful for monitoring if they have higher spatial and spectral resolution. LandTrendr is also highly processed and aggregated data. While it has been shown to be useful in detecting both disturbance and recovery the modeling method can also cause errors when performing disturbance recovery calculations on a pixel-by-pixel basis (Kennedy et al. 2010). Another possible approach would have been to use LandTrendr on the many pixels burned with high severity and fitting a regression model to see if there were differences between the mulched and unmulched areas at the landscape scale. This was not done for two reasons. First, we would not be able to explain what the differences were in anything besides NBR values. Only using pixels we sampled on the ground allows us to directly compare field results to LandTrendr results. Second, mulching data is not always accurate. Many of the GIS layers and maps that describe areas that were mulched include *planned* straw mulching, not what actually happened. For example, an area of the Hayman Fire that was supposed to have been mulched had to be disregarded when we found no mulch. On the Tripod Fire we found mulch in an area that was supposed to have been untreated. These errors could affect conclusions drawn from analysis at the landscape scale, and illustrate the need for field monitoring. On a single fire event, with well mapped treatment areas this may be a viable option, however the mulching data at our disposal ranged from planned mulching to simply a PDF map that had to be geocoded. We felt that in this case the risk of omission/commission errors was great enough to warrant the use of field sites only. Comparing NBR each year at plot pairs also allowed us to see where the LandTrendr disturbance detection algorithm was inconsistent.

4.5 *Management Implications*

While straw mulch can greatly reduce soil erosion potential, it is expensive when applied over large areas and should be limited to where values are at risk and soil erosion potential is high (Robichaud et al. 2003). As large fires continue to occur with portions burning at high severity, and with more federal funds going toward fire management and suppression (GAO 2009, Ellison et al. 2015), strategic use of mulch will be important to avoid similar rises in BAER treatment costs.

Seeding is a popular alternative to mulching, and can also be used in conjunction with it. Seeding is much cheaper than aerial application of straw or other mulch (Robichaud et al. 2014). However, after an extensive review of studies of seeding, Peppin et al. (2010) concluded that there is little evidence to support claims that seeding is an effective post-fire restoration strategy. They concluded that seeding does little to protect soil in the short term, can hinder vegetation recovery, especially if seeding introduces non-native species. Although more expensive than seeding, mulching is a much more predictable way to stabilize soil and promote vegetative regrowth (Robichaud et al. 2005, Williams et al. 2014).

We found that mulching with agricultural straw had little effect on the vegetation trajectories nine to 13 years post fire. Vegetation will eventually establish with or without mulch, and all of our plots had far less than 50% exposed soil, which is at a higher risk of erosion. We found no evidence to suggest that mulch will alter ecosystem function. Managers can expect similar densities of naturally regenerating tree seedlings with or without mulching, even if the tree species composition is different. However, differences in tree seedlings could alter longer-term vegetation trajectories if differences in species composition and height growth continue to develop.

5. Conclusions

Post-fire mulching with agricultural straw has subtle, but potentially long-term impacts on vegetation nine to 13 years post fire. We found greater tree seedling height growth and differences in species composition, as well as higher graminoid cover on mulched plots. Mulching with agricultural straw favored Douglas-fir seedlings on south- and southwest-facing aspects, and ponderosa pine on north- and northeast-facing aspects, though total tree seedling densities were similar on mulched and unmulched plots. Mulching increased tree height growth, more so on the NNE, more productive aspects. There was an average of 2% more grass cover in the vegetation component on mulched plots. Plant species diversity, species richness, and fractional cover were all similar with or without mulching. Any significant differences in vegetation trajectories between mulched and unmulched plots were on a fire by fire basis, there was no overarching trend. The yearly NBR values provided by the LandTrendr algorithm were useful in exploring the likely ecological reasons behind vegetation response after mulching.

There are slight differences between mulched and unmulched plots, but our study finds both recover given enough time. We recommend both short and long-term monitoring of mulched areas to detect if these are general trends across multiple vegetation types of the Interior West, or may change within or between specific wildfire events. While mulching is not appropriate for every area, it is important that managers know that using agricultural straw mulch to reduce soil erosion potential has minimal impacts on the long-term vegetation trajectories and site recovery post fire.

References

Abella, S., Fornwalt, P. 2014. Ten years of vegetation assembly after a North American mega fire. Global Change Biology, 21(2), 789-802

Agee, J. 1993. Fire Ecology of pacific Northwest Forests. Island Press, Washington, D.C. 3-25

Baker, W. L., Veblen, T. T., Sherriff, R. L. 2007. Fire, fuels and restoration of ponderosa pine–Douglas fir forests in the Rocky Mountains, USA. Journal of Biogeography, 34(2), 251-269.

Bautista, S., Bellot, J., Vallejo, V. R. 1996. Mulching treatment for postfire soil conservation in a semiarid ecosystem. Arid Land Research and Management, 10(3), 235-242.

Bautista, S., Robichaud, P., Blade, C. 2009. Post-fire Mulching. In: Cerda, A. (Eds.), Fire effects on soils and restoration strategies (Vol. 5). Enfield, NJ. CRC Press. 353-372

Beauchamp, J., Olson, J. 1973. Corrections for bias in regression estimates after logarithmic transformation. Ecology, 54(6), 1403-1407.

Berryman, E. Morgan, P. Robichaud, P., Page-Dumroese, D. 2014. Postfire erosion control mulches alter belowground processes and nitrate reductase activity of a perennial forb, heartleaf arnica (*Arnica cordifolia*). Research Note RMRS-RN-69. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Beyers, J. 2004. Postfire seeding for erosion control: effectiveness and impacts on native plant communities. Conservation Biology, 18(4), 947-956.

Bright, B.C., A.T. Hudak, R.E. Kennedy and A.J.H. Meddens. (2014) Landsat time series and lidar as predictors of live and dead basal area across five bark beetle-affected forests. *IEEE Journal of Selected Topics in Applied Earth Observations and Remote Sensing* 7(8): 3440-3452.

Curtis, James D., Donald W. Lynch. 1965. Ponderosa pine (*Pinus ponderosa* Laws.). in: Fowells H., (Comp.), Silvics of forest trees of the United States. U.S. Department of Agriculture, Agriculture Handbook 271. Washington, DC. 417-431

Dean, A. 2001. Evaluating effectiveness of watershed conservation treatments applied after the Cerro Grande Fire, Los Alamos, New Mexico. University of Arizona, Tucson, AZ (Thesis).

Dennison, P., Brewer, S., Arnold, J., Moritz, M. 2014. Large wildfire trends in the western United States, 1984–2011. *Geophysical Research Letters*, 41(8), 2928-2933.

Dodson, E., Peterson, D. 2010. Mulching effects on vegetation recovery following high severity wildfire in North-Central Washington State, USA. *Forest Ecology and Management* 260(10), 1816–23.

Dormann, C., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carré, G., Marquez, J., Gruber, B., Lafourcade, B., Leitao, P., Münkemüller, T., Mclean, C., Osborn, P., Reineking, B., Schoder, B., Skidmore, A., Zurell, D., Lautenbach, S. 2013. Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. *Ecography*, 36(1), 27-46.

Ellison, A., Moseley, C., Bixler, R. 2015. Drivers of Wildfire Suppression Costs. Literature Review and Annotated Bibliography. Ecosystem Workforce Program. Working Paper Number 53. Eugene, OR.

Engel, E., Abella, S. 2011. Vegetation recovery in a desert landscape after wildfires: influences of community type, time since fire and contingency effects. *Journal of Applied Ecology*, 48(6), 1401–1410.

US Government Accountability Office (GAO). 2009. “Wildland fire management: Federal agencies have taken important steps forward, but additional action is needed to address remaining challenges.” GAO-09-906-T. Washington, DC.

Facelli, J., Pickett, S. 1991. Plant litter: its dynamics and effects on plant community structure. *The Botanical Review*, 57(1), 1-32.

Harper, J. L. 1977. Population Biology of Plants. London: Academic Press.

Hudak, A., Newingham, B., E. Strand, E., Morgan, P. 2014. How vegetation recovery and fuel conditions in past fires influences fuels and future fire management in five western U.S. ecosystems. $450,000. Joint Fire Science Program. JFSP_14-1-02-27.

Johnstone, J. F., Allen, C. D., Franklin, J. F., Frelich, L. E., Harvey, B. J., Higuera, P. E., Mack, M. C., Meentemeyer, R. K., Metz, M. R., Perry, G., L., Schoennagel, T, Turner, M. G. 2016. Changing disturbance regimes, ecological memory, and forest resilience. *Frontiers in Ecology and the Environment*. 14(7), 369-378.

Kennedy, R, Yang, Z, Cohen, W. 2010. Detecting trends in forest disturbance and recovery using yearly Landsat time series: 1. LandTrendr — Temporal segmentation algorithms. *Remote Sensing of the Environment*, 114 (12), 2897-2910.

Key C.H., Benson N.C. (2005) Landscape assessment: Remote sensing of severity, the Normalized Burn Ratio, in: D.C. Lutes (Ed.), et al., FIREMON: Fire effects monitoring and inventory system, General Technical Report, RMRS-GTR-164-CD: LA1-LA51, USDA Forest Service, Rocky Mountain Research Station, Ogden, UT

Kruse, R., Bend, E., Bierzychudek, P. (2004). Native plant regeneration and introduction of non-natives following post-fire rehabilitation with straw mulch and barley seeding. *Forest Ecology and Management*, 196(2), 299-310.

LANDFIRE, 2008, Existing Vegetation Type Layer, LANDFIRE 1.1.0, U.S. Department of the Interior, Geological Survey. Accessed 28 May 2015 at http://landfire.cr.usgs.gov/viewer/.

Larson, M., Schubert, G. 1969. Root competition between ponderosa pine and grass. USDA Forest Service Research Paper RM-54.

Herman, R. Lavender, D. 1990. Douglas-fir. in: Burns, R. M., & Barbara, H. Silvics of North America: 1. conifers; 2. hardwoods. 1080-1108

Lewis, S. A., Robichaud, P. R. 2011. Using QuickBird imagery to detect cover and spread of post-fire straw mulch after the 2006 Tripod Fire, Washington, USA. Research Note RMRS-NR-43. Rocky Mountain Research Station. U.S. Department of Agriculture, Forest Service. Fort Collins, CO.

Lyon, L. J., and Stickney, P. F. 1976. Early vegetal succession following large northern Rocky Mountain wildfires. In: *Proceedings of the Montana Tall Timbers Fire Ecology Conference and Fire and Land Management Symposium*, 14, 355-375.

Morgan, Penelope, Marshell Moy, Christine A. Droske, Leigh B. Lentile, Sarah A. Lewis, Peter R. Robichaud, and Andrew T. Hudak. 2014. Vegetation response after post-fire mulching and native grass seeding. *Fire Ecology*, 10(3): 49-62.

Morgan, P., Moy, M., Droske, C. Lentile, B., Lewis, S., Robichaud, P., Hudak, A. Williams, C. 2015. Vegetation response to burn severity, seeding and salvage logging after the 2005 School Fire, Washington. *Fire Ecology*, 11(2): 31-58.

Mattsson, A. 1997. Predicting field performance using seedling quality assessment. *New Forests*, 13(1-3), 227-252.

Monitoring Trends in Burn Severity. 2012. Individual fire- level geospatial data. Available at www.mtbs.gov/. (Accessed June 30, 2016).

Mulumba, L., Lal, R. 2008. Mulching effects on selected soil physical properties. *Soil and Tillage Research*, 98(1), 106-111.

Neary, D., Klopatek, C., DeBano, L., Folliott, P. 1999 Fire effects on belowground sustainability: a review and synthesis. *Forest Ecology and Management*, 122(1–2), 51–71.

O'Hara, R., Kotze, D. 2010. Do not log-transform count data. *Methods in Ecology and Evolution*, 1(2), 118-122.

Oksanen, J., Guillaume Blanchet, F., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P., O'Hara, R.B., Simpson, G. , Solymos, P., Stevens, M. , Szoecs, E., Wagner, H. 2017. vegan: Community Ecology Package. R package version 2.4-2. https://CRAN.R-project.org/package=vegan.

Pannkuk, C., Robichaud, P. 2003. Effectiveness of needle cast at reducing erosion after forest fires. *Water Resources Research*, 39(12), 1-1 – 1-9

Parson, A., Robichaud, P., Lewis, S., Napper, C., Clark, J. 2010. Field guide for mapping post-fire soil burn severity. General Technical Report RMRS-GTR-243. Rocky Mountain Research Station. U.S. Department of Agriculture, Forest Service. Fort Collins, CO.

Peppin, D., Fulé, P. Z., Sieg, C. H., Beyers, J. L., Hunter, M. E. (2010). Post-wildfire seeding in forests of the western United States: an evidence-based review. *Forest Ecology and Management*, 260(5), 573-586.

Pinheiro J., Bates D., DebRoy S., Sarkar D. and R Core Team. 2016. _nlme: Linear and Nonlinear Mixed Effects Models_. R package version 3.1-128, <URL: http://CRAN.R-project.org/package=nlme>

PRISM Climate Group, Oregon State University. 2004. “30 Year Normals”. Accessed 28 May 2016. http://www.prism.oregonstate.edu/normals/

R Core Team. 2016. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Robichaud, P. R. 2017. “BAER Burned Area Report DB”. Rocky Mountain Research Station, United State Forest Service. Accessed 28 May 2015. https://forest.moscowfsl.wsu.edu/cgi-bin/BAERTOOLS/baer-db/index.pl

Robichaud, P. R. 2005. Measurement of post-fire hillslope erosion to evaluate and model rehabilitation treatment effectiveness and recovery. *International Journal of Wildland Fire*, 14(4), 475-485.

Robichaud, P. R., & Ashmun, L. E. 2013. Tools to aid post-wildfire assessment and erosion-mitigation treatment decisions. International Journal of Wildland Fire, 22(1), 95-105.Robichaud, P. R., Beyers, J. L., Neary, D. G. 2000. Evaluating the effectiveness of postfire rehabilitation treatments. General Technical Report RMRS-GTR-63. Rocky Mountain Research Station. U.S. Department of Agriculture, Forest Service. Moscow, ID.

Robichaud, P. R., MacDonald, L., Freeouf, J., Neary, D., Martin, D., Ashmun, L. 2003. Postfire rehabilitation of the Hayman Fire. General Technical Report RMRS-GTR-114. Rocky Mountain Research Station. U.S. Department of Agriculture, Forest Service. Moscow, ID.

Robichaud, P. R., Rhee, H., & Lewis, S. A. 2014. A synthesis of post-fire Burned Area Reports from 1972 to 2009 for western US Forest Service lands: trends in wildfire characteristics and post-fire stabilisation treatments and expenditures. *International Journal of Wildland Fire*, 23(7), 929-944.

Savage, M., Mast, J. 2005. How resilient are southwestern ponderosa pine forests after crown fires? *Canadian Journal of Forest Research*, 35(4), 967-977.

Schroeder, T., Moisen, G., Healey, S., Cohen, W. 2012. Adding value to the FIA inventory: combining FIA data and satellite observations to estimate forest disturbance. Moving From Status to Trends. General Tchnical Report GTR-NRS-P-105. Northern Research Station. U.S. Department of Agriculture, Forest Service. Baltimore, MD.

Smithwick, E., Kennedy, R., Naithani, K., Davis, K., Keller, K., Parker, L., MacEachren, A. 2013. Influence of landscape disturbance patterns on modeled carbon fluxes and associated uncertainty. In AGU Fall Meeting Abstracts. 1, 7.

Stage, A. R., Salas, C. 2007. Interactions of elevation, aspect, and slope in models of forest species composition and productivity. *Forest Science*, 53(4), 486-492.

Urza, A., Sibold, J., 2013. Nondestructive aging of postfire seedlings for four conifer species in northwestern Montana. *Western Journal of Applied Forestry*, 28(1), 22-29.

USDA, NRCS. 2017. The PLANTS Database National Plant Data Team, Greensboro, NC 27401-4901 USA. Accsedded 29 January2017 http://plants.usda.gov.

Wagenbrenner, J., MacDonald, L., Rough, D. 2006. Effectiveness of three post-fire rehabilitation treatments in the Colorado Front Range. *Hydrological Processes*, 20(14), 2989-3006.

Westerling, A., Hidalgo, H., Cayan, D., Swetnam, T. 2006. Warming and earlier spring increase western US forest wildfire activity. *Science*, 313(5789), 940-943.

Westerling, A.L., 2016. Increasing western US forest wildfire activity: sensitivity to changes in the timing of spring. *Phil. Trans. R. Soc. B*, 371(1696): 20150178.

Williams, C., Pierson, F., Robichaud, P., & Boll, J. 2014. Hydrologic and erosion responses to wildfire along the rangeland–xeric forest continuum in the western US: a review and model of hydrologic vulnerability. *International Journal of Wildland Fire*, 23(2), 155-172.

Zuur, A., Ieno, E., Walker, N., Saveliev, A., Smith, G. 2009. Mixed Effects Models and Extensions in Ecology with R. Springer New York. 101-142.

Fire	Location (State	Year of Fire	Fire Size (ha)	High Severity (ha)	Total Area Mulched (ha)	Plot Pairs Sampled	Forest Type	Elevation (m)	Latitude	Longitude	30-Year Average Summer Precipitation (cm)
Shake Table	OR	2006	4,320	1,418	130	3	Dry Mixed	1,890	44.2852	-119.2510	11.7
Cascade	ID	2007	128,350	39,350*	19,780	4	Dry Mixed	1,640	44.5925	-115.7209	17.8
Tripod	WA	2006	70,750	28,210	5,570	8	Dry Mixed	1,540	48.5856	-119.9970	20.2
Myrtle	ID	2003	1,430	104*	120	4	Dry Mixed	1,050	44.2148	-116.5810	27.3
Hayman	CO	2002	52,370	2,270	490	7	Ponderosa	2,430	39.1626	-105.3375	25.6
Ricco	SD	2005	1,440	290	80	3	Ponderosa	1,230	44.2468	-103.4713	29.8
* Myrtle Creek and Cascade Complex had scan-line issues from Landsat 7, non-processing area for each was 245 ha and 23,510 ha respectivly											

Table 1.Study areas. List of fires sampled, total acres of straw mulch and number of plot pairs sampled on each. Fire size and severity was derived from MTBS data (2012), area mulched courtesy of National Forests via Freedom of Information Act requests. Forest type determined by dominant trees pre-fire (LANDFIRE 2008); fires are ordered by forest type, and then average summer (May through August) precipitation (PRISM Climate Group 2004).

Response	Intercept			Mulch (yes/no)			Transformed Aspect (0-1)			Elevation (m)			dNBR (NBR units)			Fire (name)			Mulch*Trasp			ANOVA w/o Mulch
	Coefficient	SE	p-value	Coefficient	SE	p-value	Coefficient	SE	p-value	Coefficient	SE	p-value	Coefficient	SE	p-value	Coefficient	SE	p-value	Coefficient	SE	p-value	p-value
Ponderosa Seedling Density (s/ha)	184.6	225.5	0.430	-32.6	8.0	0.015	249.3	48.8	0.007	-0.1	0.1	0.619	-	-	-	*	*	*	-56.5	35.6	0.187	<0.001
Douglas-fir Seedling Density (s/ha)	123.9	14.6	<0.001	-103.6	12.6	0.000	-153.8	59.4	0.049	-	-	-	-	-	-	-	-	-	398.9	84.6	0.006	<0.001
Graminoid Cover (%)	4.1	1.4	0.006	1.9	0.5	0.002	-	-	-	-	-	-	-	-	-	*	*	*	-	-	-	<0.001
Soil Cover (%)	-21.1	*	<0.001	-	-	-	15.9	*	<0.001	0.0	*	<0.001	-	-	-	-	-	-	-	-	-	0.889
LandTrendr (RRate)	62.5	37.6	0.098	-12.9	15.9	0.415	-	-	-	-0.1	<0.1	0.764	0.2	<0.1	0.004	*	*	*	-	-	-	<0.001
	Intercept			Mulch			Transformed Aspect			Seedling Species			dNBR			Seedling Age			Mulch*Trasp			ANOVA w/o Mulch
	Coefficient	SE	p-value	Coefficient	SE	p-value	Coefficient	SE	p-value	Coefficient	SE	p-value	Coefficient	SE	p-value	Coefficient	SE	p-value	Coefficient	SE	p-value	p-value
Seedling Height (cm)	-49.7	29.7	0.120	12.7	2.4	0.005	24.8	3.2	0.005	*	*	*	0.0	2.4	0.0137	9.1	4.3	0.033	-12.4	2.7	0.093	0.0002

Table 2. Mixed effects models. Final mixed effects models. For each variable the coefficient, standard error (SE) and *P*-value are shown. (*) represents when a factored variable was used in the model, however because coefficients, standard error, and *P*-values varied for each factor level they were not included in this table. Transformed aspect is a cosine transformation of aspect from degrees to a continuous variable, where 0 represents 30 degrees (NNE) and 1 is 210 degrees (SSW) aspect. ANOVA value between final model and alternative model with or without 'Treatment' as a category (thus justifying the inclusion or exclusion of treatment in the model). Not shown are the values for the 'Fire' variable, which is a factor containing six variables (the six fires used in the study), each having its own coefficient, standard error, and p-value. The LandTrendr recovery model had an interaction between Treatment and 'Fire'. The coefficients, standard errors, and *P*-values for each fire interacting with mulching are as follows: Hayman: Mulched, 11.5, 18.0, 0.522; Myrtle Creek: Mulched, 26.2, 26.2, 0.319; Ricco: Mulched, 2.5, 28.8, 0.930; Shake Table: Mulched, -43.5, 22.6, 0.056; Tripod: Mulched, 37.2, 19.8, 0.061. Cascade Complex coefficient and P-value was omitted as it was the base comparison in the mixed effects model.

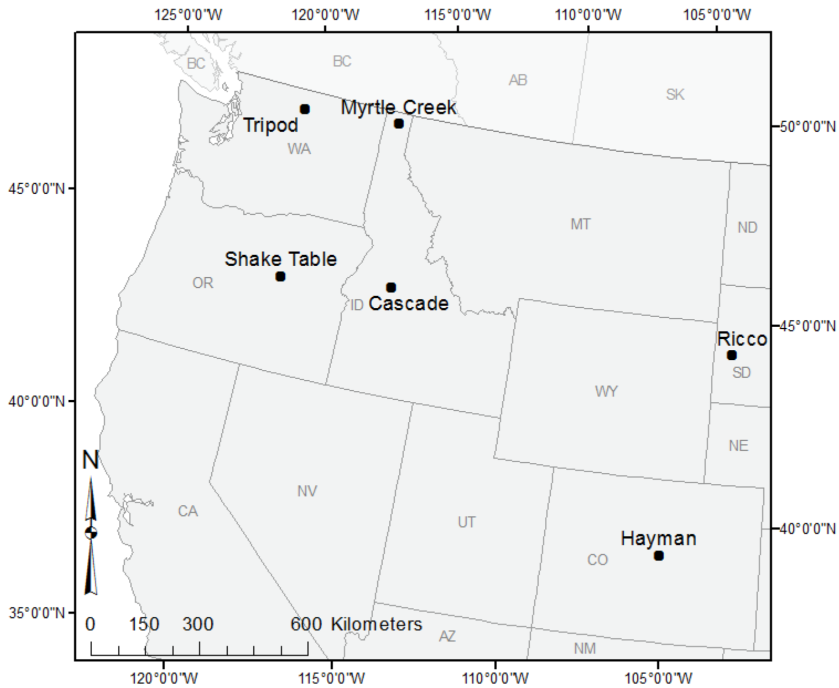

Figure 1. Study areas. Location of the six large fires sampled in the interior western United States. Albers conical projection.

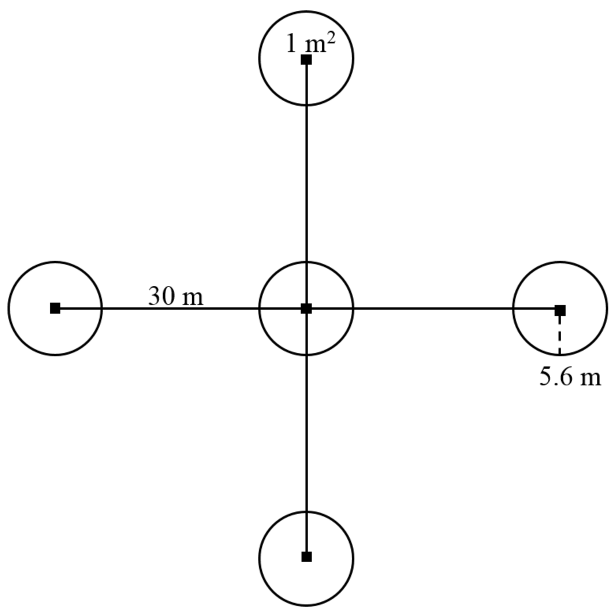

Figure 2. Study plot design. Sampling plot with five subplots. At each 1-m^2 quadrat (black square) each understory plant was identified and percent canopy cover of each species was ocularly estimated and recorded. Fractional cover of green vegetation, NP, soil and rock were also recorded. At each 5.6-m radius plot (circles), total tree seedling density was recorded, as well at total and yearly height growth on a subsample of seedlings.

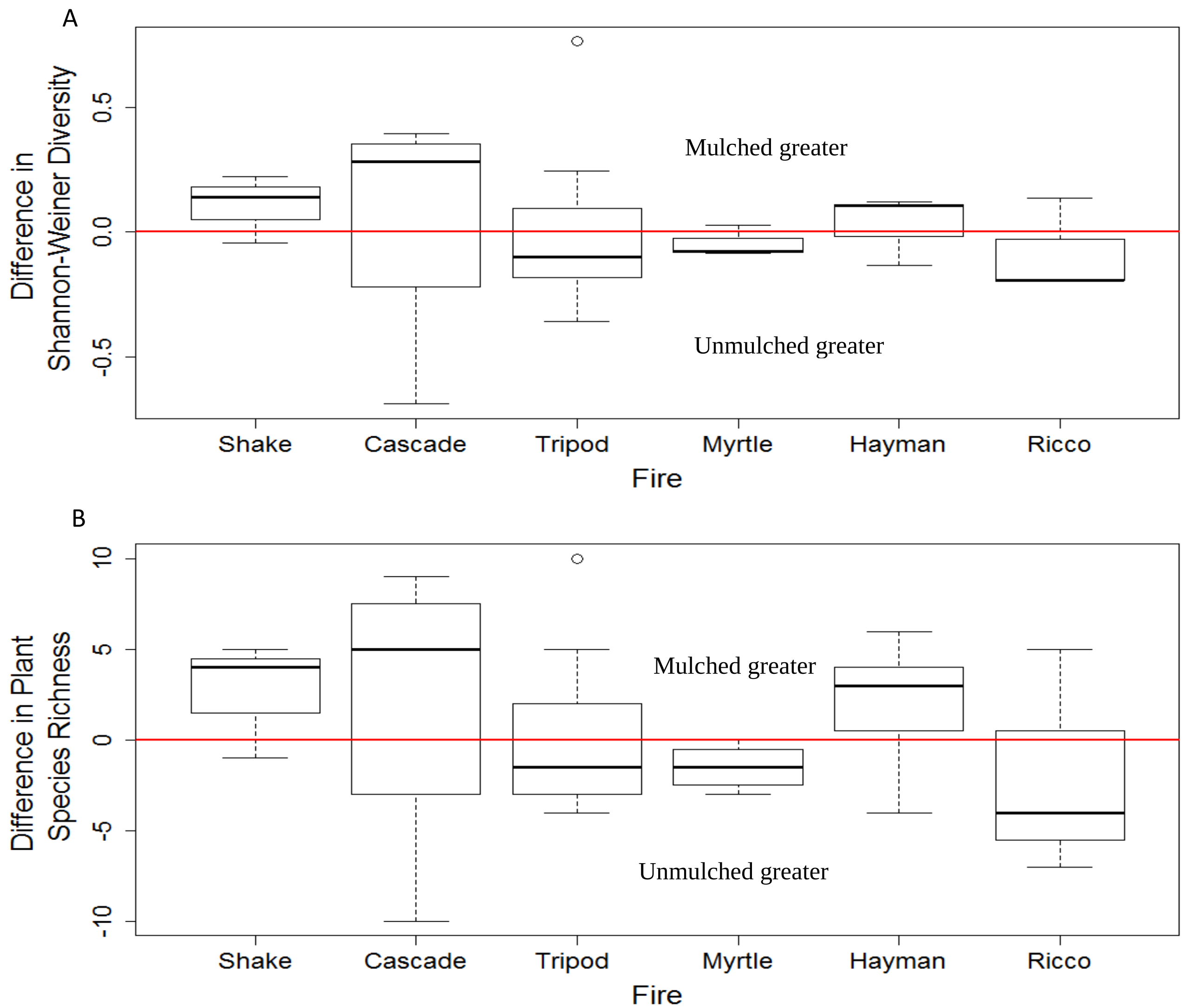

Figure 3. Differenced plant diversity and species richness. These are box plots of differences by plot pair. Any point above the zero line represents a higher value on the mulched plot, while any value below is a higher value on the unmulched plot of the pair. (A) Differenced plant species diversity for mulched and unmulched plot pairs on each fire. (B) Differenced species richness (number of plants species) for plot pairs on each fire. For the box plots, the thick line is the median (50% quantile), and top and bottom of boxes are 75% quantile and 25% quantile respectively. Ends of whiskers extend a maximum of 1.5 times the median to 75% quantile or 25% quantile, or to the farthest point within that range, whichever is closest to the median. Circles are outliers beyond this range.

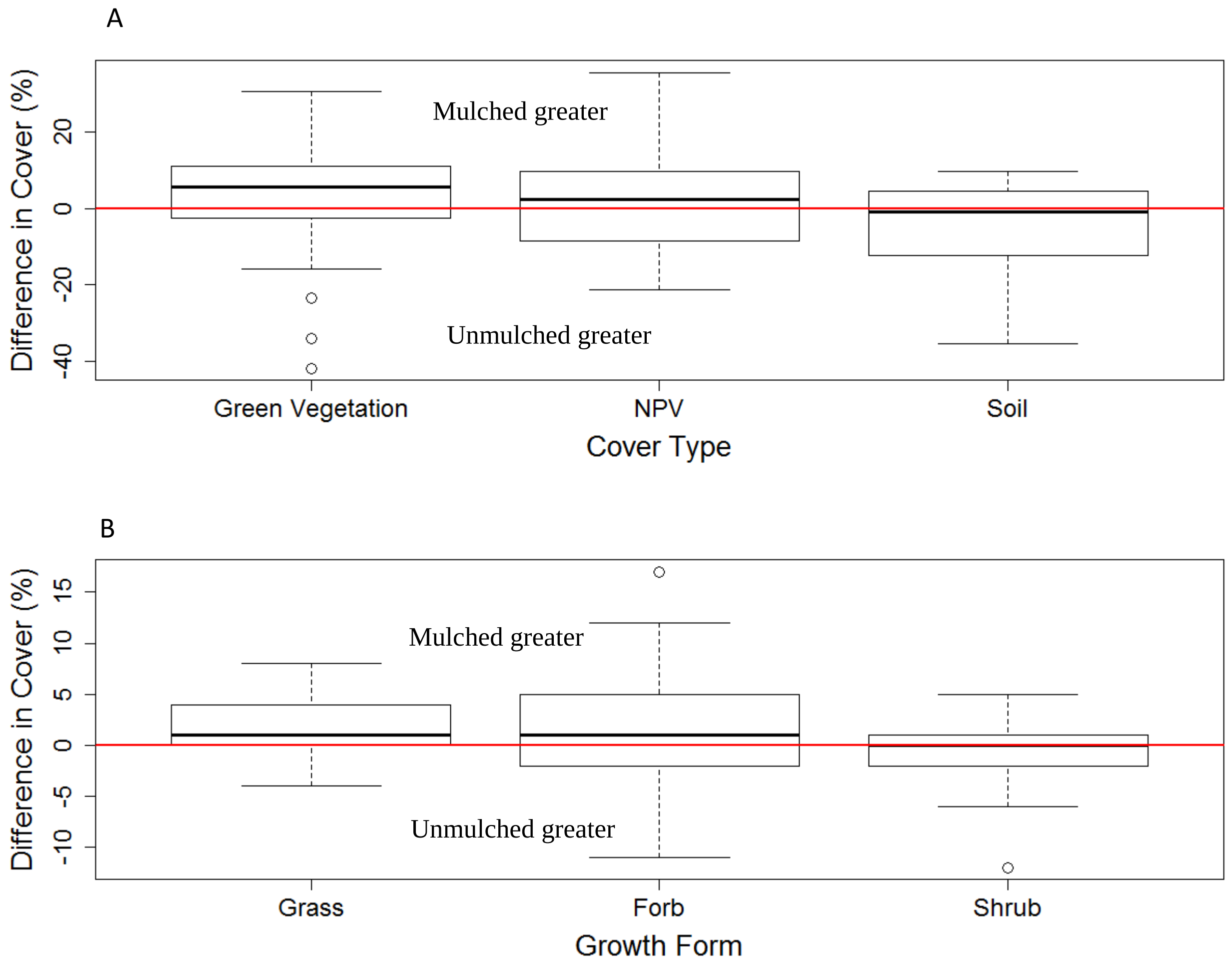

Figure 4. Differenced fractional cover and cover by growth form. Understory ground cover boxplots differenced by plot pair. Any point above the zero line represents a higher value on the mulched plot, while any value below is a higher value on the unmulched plot of the pair. (A) Differenced fractional cover for green and non-photosynthetic vegetation (NPV) and soil for plot pairs by fire. (B) Differenced percent canopy cover of three main vegetation cover groups by plot pair. Three subplots on the Tripod Complex contained residual mulch; this was added to the NPV class for that plot and the values were 11%, 3% and 0.4%. Thick line represents median (50% quantile), top and bottom of box are 75% quantiles and 25% quantiles respectively. Ends of whiskers extend a maximum of 1.5 times the median to 75% quantile or 25% quantile, or to the farthest point within that range, whichever is closest to the median. Circles are outliers beyond this range.

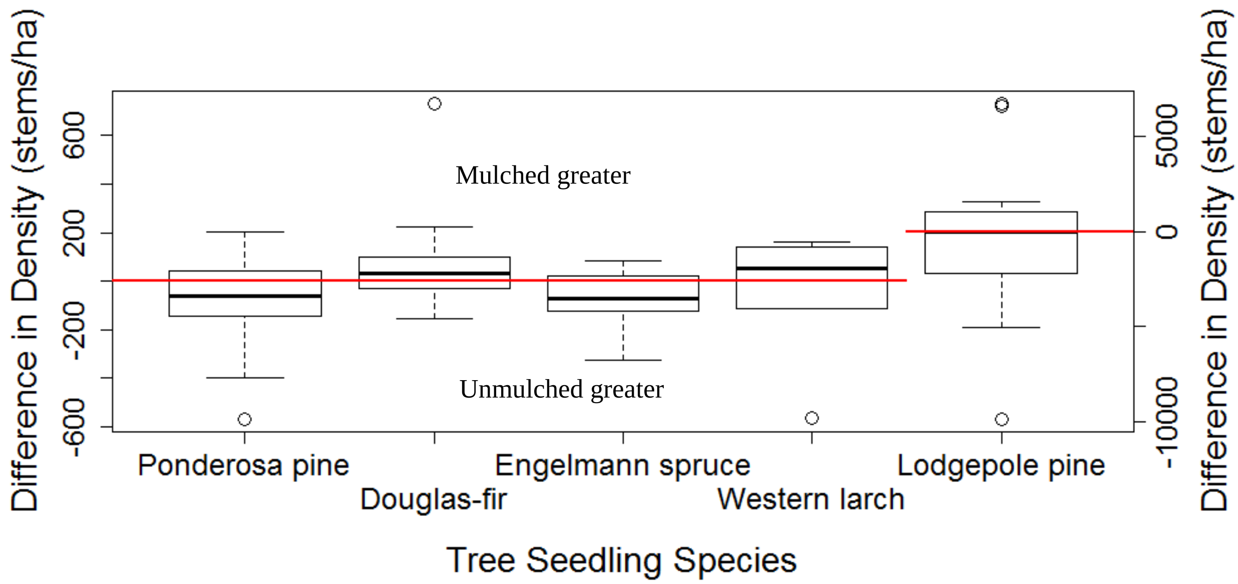

Figure 5. *Differenced tree seedling density by species*. For each plot pair, unmulched density is subtracted from mulched density. Any point above the zero line represents a higher value on the mulched plot, while any value below is a higher value on the unmulched plot of the pair. Only plot pairs with non-zero values were included. Note scale difference for lodgepole pine. For box plots, the thick line represents median (50% quantile), top and bottom of box are 75% quantiles and 25% quantiles respectively. Ends of whiskers extend a maximum of 1.5 times the median to 75% quantile or 25% quantile, or to the farthest point within that range, whichever is closest to the median. Circles are outliers beyond this range.

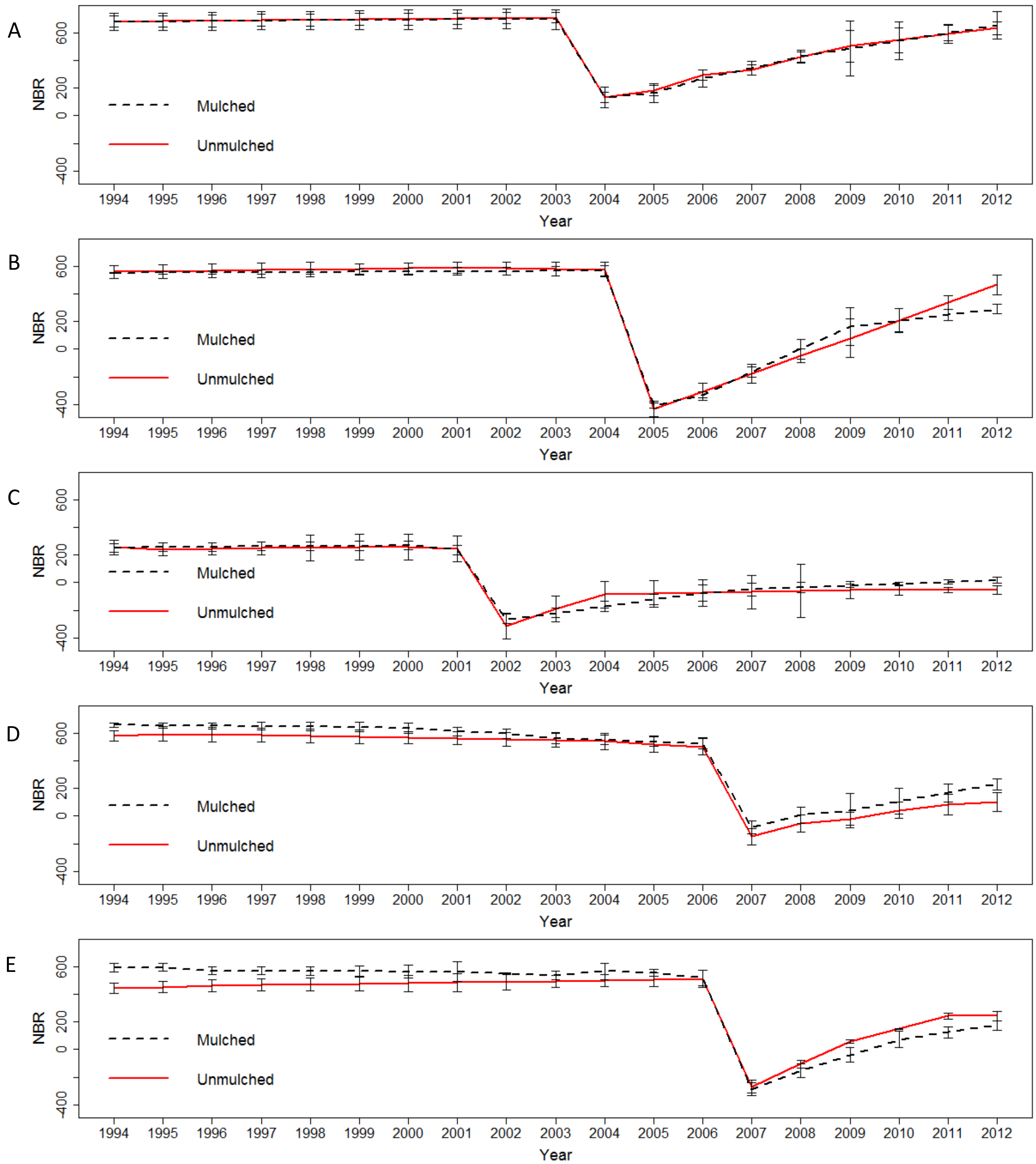

Figure 6. Visual interpretation of LandTrendr recoveries. Example NBR trajectories before, during and after fire for mulched and unmulched plot pairs. Each row includes representative pattern of NBR trajectories for the average of five pixels representing the mulched (dashed line) and unmulched (solid line) plot pair. NBR is a remotely-sensed vegetation index that is sensitive to vegetation and soil within a given pixel. (A) Mulched and unmulched plot pairs had similar NBR values, 41%. This was

the most common across all fires, this specific plot pair was from Myrtle Creek. (B) Mulched plots appear to be recovering at a faster rate initially, but after five years, the unmulched plots recover both at a higher rate and to a higher value, 3%. This is most likely caused by mulch presenting a physical barrier early on, but this barrier was reduced as mulch decomposed or blew away. This example is from the Ricco Fire. (C) Mulch seemed to suppress recovery for the first three years, but then the mulched plot eventually recovered to a higher point than the unmulched, 3%. This is most likely caused by mulch altering microsite conditions initially, allowing what vegetation was there to thrive and increase once the mulch was gone. This example is from the Hayman Fire. (D) Vegetation recovered faster on mulched than on unmulched, 7% paired plot. This example is from the Tripod Complex. (E) Vegetation recovered more slowly on mulched than on unmulched, 10% paired plot. This example is from Tripod Complex. Error bars are twice the standard error calculated from the five subplot observations at these representative selected sites.

APPENDIX A: Plant Species

					Ricco		Hayman		Tripod		Shake		Cascade		Myrtle	
Growth Form	**Family**	**Scientific Name**	**Symbol**	**Common Name**	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched
Forb	Apocynaceae	Apocynum androsaemifolium	APAN2	spreading dogbane	X	X								X		
Forb	Araliaceae	Aralia nudicaulis	ARNU2	wild sarsaparilla											X	
Forb	Asteraceae	Achillea millefolium	ACMI2	common yarrow			X	X	X	X	X	X	X	X	X	X
Forb	Asteraceae	Agoseris glauca	AGGL	pale agoseris			X									
Forb	Asteraceae	Antenria microphylla	ANMI3	littleleaf pussytoes					X	X						
Forb	Asteraceae	Antenria parvifolia	ANPA4	small-leaf pussytoes			X	X								
Forb	Asteraceae	Antenria sp.	ANTEN	pussytoes	X	X										
Forb	Asteraceae	Aphalis margaritacea	ANMA	western pearly everlasting					X	X	X	X			X	X
Forb	Asteraceae	Arnica cordifolia	ARCO9	heartleaf arnica					X	X	X					
Forb	Asteraceae	Artemisia campestris	ARCA12	field sagewort			X	X								
Forb	Asteraceae	Artemisia vulgaris	ARVU	common wormwood			X	X								
Forb	Asteraceae	Aster	ASTER	aster									X			
Forb	Asteraceae	Aster sp.	UNKAST1	NA					X	X						
Forb	Asteraceae	Asteracea sp.	UNKWAS	NA			X									
Forb	Asteraceae	Cirsium arvense	CIAR4	Canada thistle	X	X										
Forb	Asteraceae	Cirsium canescens	CICA11	prairie thistle			X									
Forb	Asteraceae	Cirsium sp.	CIRSIUM3	NA			X									
Forb	Asteraceae	Cirsium vulgare	CIVU	bull thistle									X			
Forb	Asteraceae	Crepis sp.	CREPI	hawksbeard									X			
Forb	Asteraceae	Echicea angustifolia	ECAN2	blacksamson echinacea	X											
Forb	Asteraceae	Elephantopus elatus	ELEL3	tall elephantsfoot			X									
Forb	Asteraceae	Erigeron subtrinervis	ERSU2	threenerve fleabane	X											
Forb	Asteraceae	Erigeron vetensis	ERVE2	early bluetop fleabane			X									
Forb	Asteraceae	Eurybia conspicua	EUCO36	western showy aster						X			X	X		X
Forb	Asteraceae	Heterotheca villosa	HEVI4	hairy false goldenaster	X	X	X	X								
Forb	Asteraceae	Hieracium	HIERA	Linea borealis									X	X		
Forb	Asteraceae	Hieracium albiflorum	HIAL2	white hawkweed					X	X						
Forb	Asteraceae	Hieracium scouleri	HISC2	Scouler's woollyweed						X					X	X
Forb	Asteraceae	Hieracium scouleri	HISCA	Scouler's woollyweed									X			
Forb	Asteraceae	Lactuca serriola	LASE	prickly lettuce	X	X	X									X
Forb	Asteraceae	Liatris punctata	LIPU	dotted blazing star	X											
Forb	Asteraceae	Machaeranthera parviflora	MAPA	smallflower tansyaster			X									
Forb	Asteraceae	Madia glomerata	MAGL2	mountain tarweed					X							
Forb	Asteraceae	Madia exigua	MAEX	small tarweed											X	X
Forb	Asteraceae	Senecio pseudoarnica	SEPS	seaside ragwort									X			
Forb	Asteraceae	Senecio streptanthifolius	PAST10	NA					X	X						
Forb	Asteraceae	Senecio wootonii	SEWO	Wooton's ragwort			X									
Forb	Asteraceae	Solidago	SOLID	goldenrod									X	X		
Forb	Asteraceae	Solidago canadensis	SOCA6	Canada goldenrod											X	X
Forb	Asteraceae	Solidago missouriensis	SOMI2	Missouri goldenrod	X	X										
Forb	Asteraceae	Solidago mollis	SOMO	velvety goldenrod	X	X										
Forb	Asteraceae	Solidago nana	SONA	baby goldenrod			X	X								
Forb	Asteraceae	Taraxacum officile	TAOF	common dandelion	X	X	X	X	X	X			X	X	X	X
Forb	Asteraceae	Tetraneuris acaulis	TEAC	stemless four-nerve daisy	X											
Forb	Asteraceae	Tragopogon dubius	TRDU	yellow salsify	X	X	X	X							X	X
Forb	Boraginaceae	Boragenacea sp.	UNK1	NA									X			
Forb	Boraginaceae	Mertensia lanceolata	MELA3	prairie bluebells			X									
Forb	Brassicaceae	Alyssum simplex	ALSI8	alyssum			X	X								

Growth Form	Family	Scientific Name	Symbol	Common Name	Ricco		Hayman		Tripod		Shake		Cascade		Myrtle	
					Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched
Forb	Brassicaceae	Arabis cobrensis	ARCO	sagebrush rockcress									X	X		
Forb	Brassicaceae	Arabis hoffmannii	ARHO	Hoffmann's rockcress									X	X		
Forb	Brassicaceae	Brassicacea sp.	UNKBLL	NA	X											
Forb	Brassicaceae	Cardamine	CARDA	bittercress									X			
Forb	Brassicaceae	Noccaea monta	NOMO2	alpine pennycress			X	X								
Forb	Brassicaceae	Rorippa sarmentosa	ROSA	longrunner									X			
Forb	Brassicaceae	Sisymbrium altissimum	SIAL2	tall tumblemustard									X			
Forb	Campanulaceae	Campanula rotundifolia	CARO2	bluebell bellflower	X	X										
Forb	Caprifoliaceae	Linea borealis	LIBO3	twinflower					X				X	X		
Forb	Caryophyllaceae	Arenaria	ARENA	sandwort									X	X		
Forb	Caryophyllaceae	Arenaria congesta	ARCO5	ballhead sandwort							X					
Forb	Caryophyllaceae	Cerastium arvense	CEAR4	field chickweed			X									
Forb	Caryophyllaceae	Minuartia michauxii	MIMI2	Michaux's stitchwort							X					
Forb	Caryophyllaceae	Pink family	CARYFY	NA			X	X								
Forb	Caryophyllaceae	Silene menziesii	SIME	Menzies' campion					X							
Forb	Caryophyllaceae	Silene sp.	SILEN	catchfly									X			
Forb	Clusiaceae	Hypericum perforatum	HYPE	common St. Johnswort											X	X
Forb	Crassulaceae	Sedum lanceolatum	SELA	spearleaf stonecrop					X							
Forb	Crassulaceae	Sedum radiatum	SERAC	Coast Range stonecrop											X	
Forb	Crassulaceae	Sedum sarmentosum	SESA	stringy stonecrop							X					
Forb	Cyperaceae	Carex	CAREX	sedge	X	X	X	X	X	X	X	X	X	X	X	X
Forb	Dennstaedtiaceae	NA	PTAQ	western brackenfern											X	
Forb	Equisetaceae	Equisetum arvense	EQAR	field horsetail						X			X			
Forb	Fabaceae	Astragalus cadensis	ASCA11	Canadian milkvetch					X							
Forb	Fabaceae	Astragalus crassicarpus	ASCR2	groundplum milkvetch	X											
Forb	Fabaceae	Astragalus laxmannii	ASLAR	prairie milkvetch	X	X										
Forb	Fabaceae	Dalea purpurea	DAPU5	purple prairie clover	X											
Forb	Fabaceae	Fabacea sp.	FAB	NA			X									
Forb	Fabaceae	Glycyrrhiza glabra	GLGL	cultivated licorice							X					
Forb	Fabaceae	Lupinus caudatus	LUCA	tailcup lupine							X	X				
Forb	Fabaceae	Lupinus sericeus	LUSE4	silky lupine					X	X						
Forb	Fabaceae	Medicago lupuli	MELU	black medick	X											
Forb	Fabaceae	NA	LUPIN	lupine											X	X
Forb	Fabaceae	Oxytropis lambertii	OXLA3	purple locoweed	X											
Forb	Fabaceae	Oxytropis multiceps	OXMU	Nuttall's oxytrope			X									
Forb	Fabaceae	Pediomelum argophyllum	PEAR6	silverleaf Indian breadroot	X	X										
Forb	Fabaceae	Pediomelum esculentum	PEES	large Indian breadroot	X	X										
Forb	Fabaceae	Thermopsis rhombifolia	THRH	prairie thermopsis			X									
Forb	Fabaceae	Vicia america	VIAM	American vetch	X											
Forb	Geraniaceae	Geranium sp.	GERAN	NA			X									
Forb	Geraniaceae	Geranium viscosissimum	GEVI2	sticky purple geranium									X			
Forb	Hydrophyllaceae	Phacelia hastata	PHHA	silverleaf phacelia							X	X				
Forb	Hydrophyllaceae	Phacelia heterophylla	PHHE2	varileaf phacelia			X	X								
Forb	Hydrophyllaceae	Phacelia sp.	PHASE	NA							X					
Forb	Lamiaceae	Agastache urticifolia	AGUR	nettleleaf giant hyssop							X					
Forb	Lamiaceae	Morda fistulosa	MOFI	wild bergamot	X											
Forb	Lamiaceae	Nepeta cataria	NECA2	catnip	X											
Forb	Liliaceae	Liliaceae family	LILFY	NA			X									
Forb	Liliaceae	Maianthemum stellatum	MAST4	starry false lily of the valley	X											X

Growth Form	Family	Scientific Name	Symbol	Common Name	Ricco		Hayman		Tripod		Shake		Cascade		Myrtle	
					Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched
Forb	Liliaceae	Zigadenus elegans	ZIEL2	mountain deathcamas	X											
Forb	Nyctaginaceae	Boerhavia gracillima	BOGR	slimstalk spiderling			X									
Forb	Onagraceae	Chamerion angustifolium	CHAN9	Fireweed					X	X	X	X	X	X	X	X
Forb	Onagraceae	Epilobium sp.	EPILOA	NA					X							
Forb	Onagraceae	Epilobium brachycarpum	EPBR3	tall annual willowherb											X	X
Forb	Onagraceae	Gayophytum sp.	GAYOP	NA							X	X				
Forb	Polemoniaceae	NA	PHHO	spiny phlox											X	
Forb	Polygonaceae	Polygonum SP.	POLYG4	knotweed									X			
Forb	Primulaceae	Androsace septentriolis	ANSE4	pygmyflower rockjasmine			X	X								
Forb	Ranunculaceae	Actaea rubra	ACRU2	red baneberry							X	X				
Forb	Ranunculaceae	Anemone lancifolia	ANLA	mountain thimbleweed									X			
Forb	Ranunculaceae	Anemone piperi	ANPI	Piper's anemone									X			
Forb	Ranunculaceae	Clematis columbia	CLCO2	rock clematis			X	X								
Forb	Ranunculaceae	Clematis columbia	CLCOT	rock clematis	X	X										
Forb	Ranunculaceae	Pulsatilla patens	PUPA5	eastern pasqueflower	X	X	X	X								
Forb	Ranunculaceae	Ranunculus	RANUN	buttercup									X	X		
Forb	Ranunculaceae	Thalictrum occidentale	THOC	western meadow-rue									X			
Forb	Ranunculaceae	Anemone quinquefolia	ANMI	nightcaps									X	X		
Forb	Rosaceae	Fragaria sp.	FRAGA	strawberry									X	X		
Forb	Rosaceae	Fragaria vesca	FRVE	woodland strawberry											X	X
Forb	Rosaceae	Fragaria virginia	FRVI	Virginia strawberry		X			X							
Forb	Rosaceae	Potentilla arguta	POAR7	tall cinquefoil	X											
Forb	Rosaceae	Potentilla fissa	POFI3	bigflower cinquefoil			X	X								
Forb	Rosaceae	Potentilla SP.	POTEN	cinquefoil			X									
Forb	Rubiaceae	Galium boreale	GABO2	northern bedstraw	X	X										
Forb	Rubiaceae	Galium sp.	GALIU	NA									X			
Forb	Saxifragaceae	Heuchera cylindrica	HECY2	roundleaf alumroot									X			
Forb	Scrophulariaceae	Collinsia parviflora	COPA3	maiden blue eyed Mary									X	X		
Forb	Scrophulariaceae	Penstemon glaber	PEGL3	sawsepal penstemon			X									
Forb	Scrophulariaceae	Penstemon sp.	PENST	NA	X				X				X			
Forb	Scrophulariaceae	Verbascum thapsus	VETH	common mullein	X	X	X	X								
Forb	Violaceae	Viola glabella	VIGL	pioneer violet											X	
Forb	Violaceae	Viola orbiculata	VIOR	darkwoods violet											X	
Forb	Violaceae	Viola sp.	VIOLA	violet						X			X			
Graminoid	Cyperaceae	Carex geophila	CAGE	White Mountain sedge					X							
Graminoid	Cyperaceae	Carex rossii	CARO5	Ross' sedge					X	X						
Graminoid	Cyperaceae	Eleocharis elegans	ELEL	elegant spikerush			X									
Graminoid	Poaceae	Achtherum nelsonii	ACNEN2	Columbia needlegrass			X									
Graminoid	Poaceae	Bouteloua curtipendula	BOCU	sideoats grama			X									
Graminoid	Poaceae	Bromus carinatus	BRCA5	California brome											X	
Graminoid	Poaceae	Bromus ciliatus	BRCI2	fringed brome			X									
Graminoid	Poaceae	Bromus inermis	BRIN2	smooth brome	X	X									X	X
Graminoid	Poaceae	Bromus inermis	BRINP	Pumpelly's brome			X									
Graminoid	Poaceae	Bromus sp.	BROMU	NA			X									
Graminoid	Poaceae	Bromus tectorum	BRTE	cheatgrass			X	X							X	X
Graminoid	Poaceae	Bromus arvensis	BRAR5	field brome											X	
Graminoid	Poaceae	Bromus vulgaris	BRVU	Columbia brome											X	
Graminoid	Poaceae	Calamagrostis cadensis	CACA4	bluejoint					X							
Graminoid	Poaceae	Calamagrostis rubescens	CARU	pinegrass					X	X			X	X	X	X
Graminoid	Poaceae	Danthonia spicata	DASP2	poverty oatgrass	X	X										

Growth Form	Family	Scientific Name	Symbol	Common Name	Ricco		Hayman		Tripod		Shake		Cascade		Myrtle	
					Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched
Graminoid	Poaceae	Dichanthelium oligosanthes	DIOLS	Scribner's rosette grass	X											
Graminoid	Poaceae	Elymus cadensis	ELCA4	Canada wildrye	X											
Graminoid	Poaceae	Elymus elymoides	ELEL5	squirreltail			X	X								
Graminoid	Poaceae	Elymus glaucus	ELGL	blue wildrye							X	X				
Graminoid	Poaceae	Elymus trachycaulus	ELTR7	slender wheatgrass			X									
Graminoid	Poaceae	Elymus trachycaulus	ELTRT	slender wheatgrass	X	X										
Graminoid	Poaceae	Eragrostis cilianensis	ERCI	stinkgrass			X									
Graminoid	Poaceae	Festuca idahoensis	FEID	Idaho fescue							X	X			X	X
Graminoid	Poaceae	Festuca occidentalis	FEOC	western fescue					X							
Graminoid	Poaceae	Hesperostipa comata	HECO26	needle and thread	X		X									
Graminoid	Poaceae	Koeleria macrantha	KOMA	prairie Junegrass				X							X	
Graminoid	Poaceae	Muhlenbergia monta	MUMO	mountain muhly			X	X								
Graminoid	Poaceae	Nassella viridula	NAVI4	green needlegrass	X	X										
Graminoid	Poaceae	Pascopyrum smithii	PASM	western wheatgrass	X	X										
Graminoid	Poaceae	Poa fendleria	POFE	muttongrass			X	X								
Graminoid	Poaceae	Poa nemoralis	PONE	wood bluegrass			X									
Graminoid	Poaceae	Poa pratensis	POPR	Kentucky bluegrass	X	X										
Graminoid	Poaceae	Poa secunda	POSE	Sandberg bluegrass			X	X	X	X						
Graminoid	Poaceae	Poa tracyi	POTR	Tracy's bluegrass			X	X					X			
Graminoid	Poaceae	Pseudoroegneria spicata	PSSP6	bluebunch wheatgrass			X	X				X			X	X
Graminoid	Poaceae	Ventenata dubia	VEDU	North Africa grass											X	X
Shrub	Agavaceae	Yucca glauca	YUGL	soapweed yucca			X	X								
Shrub	Anacardiaceae	Rhus glabra	RHGL	smooth sumac	X	X										
Shrub	Anacardiaceae	Toxicodendron rydbergii	TORY	western poison ivy	X	X										
Shrub	Asteraceae	Artemisia frigida	ARFR4	prairie sagewort			X	X								
Shrub	Asteraceae	Dubautia pauciflorula	DUPA5	Wahiawa Bog dubautia	X											
Shrub	Berberidaceae	Mahonia nervosa	MANE2	Cascade barberry											X	X
Shrub	Berberidaceae	Mahonia repens	MARE11	creeping barberry									X		X	
Shrub	Betulaceae	Alnus	ALNUS	alder									X			
Shrub	Betulaceae	Alnus sinuata	ALSI3	NA					X	X						
Shrub	Betulaceae	Alnus viridis	ALVI5	green alder											X	X
Shrub	Caprifoliaceae	Sambucus racemosa	SARA2	red elderberry					X	X		X				
Shrub	Caprifoliaceae	Sambucus sp.	SABMU	NA							X					
Shrub	Caprifoliaceae	Symphoricarpos albus	SYAL	common snowberry		X	X								X	X
Shrub	Celastraceae	Celastrus scandens	CESC	American bittersweet	X											
Shrub	Celastraceae	Paxistima myrsinites	PAMY	Oregon boxleaf					X	X						
Shrub	Elaeagnaceae	Shepherdia cadensis	SHCA	russet buffaloberry					X	X			X	X		
Shrub	Ericaceae	Arctostaphylos nevadensis	ARNE	pinemat manzanita					X	X						
Shrub	Ericaceae	Arctostaphylos uva-ursi	ARUV	kinnikinnick	X			X								
Shrub	Ericaceae	Rhododendron albiflorum	RHAL2	Cascade azalea					X							
Shrub	Ericaceae	Vaccinium membraceum	VAME	thinleaf huckleberry					X	X			X	X		
Shrub	Ericaceae	Vaccinium scoparium	VASC	grouse whortleberry					X	X						
Shrub	Ericaceae	Vaccinium sp.	VACCI	blueberry									X			
Shrub	Ericaceae	Vaccinium caespitosum	VACA	dwarf bilberry					X							
Shrub	Fabaceae	Amorpha canescens	AMCA6	leadplant	X	X										
Shrub	Grossulariaceae	Ribes cereum	RICE	wax currant			X	X								
Shrub	Grossulariaceae	Ribes lacustre	RILA	prickly currant					X							
Shrub	Grossulariaceae	Ribes viscosissimum	RIVI3	sticky currant							X	X	X			
Shrub	Pyrolaceae	Chimaphila umbellata	CHUM	pipsissewa					X	X						
Shrub	Pyrolaceae	Orthilia secunda	ORSE	sidebells wintergreen					X	X						

					Ricco		Hayman		Tripod		Shake		Cascade		Myrtle	
Growth Form	**Family**	**Scientific Name**	**Symbol**	**Common Name**	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched	Mulched	Unmulched
Shrub	Pyrolaceae	pyrola sp.	PYROL	NA							X					
Shrub	Rhamnaceae	Ceanothus fendleri	CEFE	Fendler's ceanothus			X	X								
Shrub	Rhamnaceae	Ceanothus herbaceus	CEHE	Jersey tea	X											
Shrub	Rhamnaceae	Ceanothus sanguineus	CESA	redstem ceanothus							X	X			X	X
Shrub	Rhamnaceae	Ceanothus velutinus	CEVE	snowbrush ceanothus					X	X	X	X	X	X	X	
Shrub	Rosaceae	Amelanchier alnifolia	AMAL2	Saskatoon serviceberry	X	X				X			X	X	X	
Shrub	Rosaceae	Holodiscus discolor	HODI	oceanspray											X	
Shrub	Rosaceae	Physocarpus malvaceus	PHMA5	mallow ninebark											X	X
Shrub	Rosaceae	Physocarpus monogynus	PHMO4	mountain ninebark	X											
Shrub	Rosaceae	Prunus virginia	PRVI	chokecherry	X	X		X							X	
Shrub	Rosaceae	Rosa acicularis	ROAC	prickly rose			X									
Shrub	Rosaceae	Rosa gymnocarpa	ROGY	dwarf rose									X		X	X
Shrub	Rosaceae	Rosa nutkana	RONU	Nootka rose											X	
Shrub	Rosaceae	Rosa sp.	N/A	NA			X									
Shrub	Rosaceae	Rosa SP.	ROSA5	rose		X							X			
Shrub	Rosaceae	Rosa woodsii	ROWO	Woods' rose											X	
Shrub	Rosaceae	Rubus deliciosus	RUDE	delicious raspberry			X									
Shrub	Rosaceae	Rubus idaeus	RUID	American red raspberry			X	X								
Shrub	Rosaceae	Rubus parviflorus	RUPA	thimbleberry									X		X	X
Shrub	Rosaceae	Rubus sp.	RUBUS	blackberry					X							
Shrub	Rosaceae	Spiraea betulifolia	SPBE2	white spirea	X	X			X	X			X	X	X	X
Shrub	Salicaceae	Salix sp.	SALIX	willow					X	X			X	X	X	X
Tree	Aceraceae	Acer glabrum	ACGL	Rocky Mountain maple											X	X
Tree	Arecaceae	Acrocomia media	ACME2	grugru palm			X	X								
Tree	Fagaceae	Quercus macrocarpa	QUMA2	bur oak	X	X										
Tree	Pinaceae	Abies lasiocarpa	ABLA	subalpine fir											X	X
Tree	Pinaceae	Larix occidentalis	LAOC	western larch							X				X	X
Tree	Pinaceae	Picea engelmannii	PIEN	Engelmann spruce									X			
Tree	Pinaceae	Pinus contorta	PICO	lodgepole pine					X	X			X	X	X	X
Tree	Pinaceae	Pinus ponderosa	PIPO	ponderosa pine	X					X						
Tree	Pinaceae	Pseudotsuga menziesii	PSME	Douglas-fir			X		X		X					X
Tree	Salicaceae	Populus balsamifera	POBAT	black cottonwood											X	
Tree	Salicaceae	Populus tremuloides	POTR5	quaking aspen					X	X						X

APPENDIX B: Plot Data

Fire	Pair ID	Treatment	Fractional Cover			Vegetation Cover			Stems/ ha			Diversity	Rmag	Physical	
			Green Vegetation	NPV	Soil	Forb	Grass	Shrub	All Species	Douglas-fir	Ponderosa pine			dNBR	Trasp
Cascade	A	Unmulched	44	35	21	20	5	2	832.3	0	0	2.6145	359.4	436.2	0.93
Cascade	A	Mulched	51	23	21	26	5	4	1603.7	203	0	2.866	162.6	407.8	0.96
Cascade	B	Unmulched	63	36	1	19	5	14	13398.2	0	101.5	2.6993	183.2	620.6	0.03
Cascade	B	Mulched	65	27	7	29	5	15	3207.5	81.2	40.6	3.0959	216.4	477.6	0.16
Cascade	C	Unmulched	51	18	27	25	5	5	751.1	277.4	0	2.9082	177	677.8	0.79
Cascade	C	Mulched	44	19	21	19	5	6	263.9	243.6	0	2.2184	187	652.4	0.93
Cascade	D	Unmulched	45	24	15	9	5	15	6090.1	0	0	2.5245	354.4	688.8	0.32
Cascade	D	Mulched	51	35	13	21	5	14	2436	0	0	2.8394	181	703.4	0.42
Hayman	E	Unmulched	30	41	26	26	5	1	0	0	0	3.0472	258.8	472.6	0.45
Hayman	E	Mulched	31	31	36	29	5	6	0	0	0	3.1552	453	560	0.07
Hayman	F	Unmulched	31	26	25	35	5	7	697	500.7	196.2	3.3244	463	576.2	0.01
Hayman	F	Mulched	49	29	16	44	7	5	162.4	101.5	40.6	3.44	376.2	444	0.02
Hayman	G	Unmulched	35	21	43	29	8	9	40.6	0	40.6	3.0201	261	687.6	0.14
Hayman	G	Mulched	44	26	29	29	7	3	0	0	0	2.9613	57.2	173.2	0.96
Hayman	H	Unmulched	39	26	34	43	8	4	203	142.1	20.3	3.2014	263.2	458	0.09
Hayman	H	Mulched	36	14	40	32	14	4	0	0	0	3.2288	281.6	484.8	0.65
Hayman	I	Unmulched	34	18	38	32	5	5	0	0	0	3.0632	274.6	461.8	0.89
Hayman	I	Mulched	48	22	19	34	8	2	0	0	0	3.184	430.8	698	0.3
Hayman	J	Unmulched	27	15	56	34	6	6	0	0	0	3.2611	508.8	635	0.68
Hayman	J	Mulched	41	35	23	33	14	5	0	0	0	3.1267	164.6	257	0.99
Hayman	K	Unmulched	37	33	30	27	7	5	121.8	81.2	40.6	3.1146	325.6	379.6	0.65
Hayman	K	Mulched	47	24	22	30	13	2	0	0	0	3.2206	314.6	586.8	0.26
Myrtle	L	Unmulched	75	24	1	8	6	26	0	0	0	2.9624	446.6	287.2	0.97
Myrtle	L	Mulched	33	59	5	18	11	21	0	0	0	2.889	391.6	137	0.98
Myrtle	M	Unmulched	12	47	30	25	17	7	20.3	0	20.3	2.9884	180.6	220.2	0.92
Myrtle	M	Mulched	24	36	33	22	16	8	0	0	0	2.9031	340.6	149.6	0.98
Myrtle	N	Unmulched	50	43	3	13	8	12	3552.6	182.7	40.6	3.0845	93.6	352.4	0.99
Myrtle	N	Mulched	34	38	13	11	11	9	4290.1	40.6	771.4	3.0055	148.8	456	0.98
Myrtle	O	Unmulched	59	39	2	8	2	22	21356	81.2	162.4	2.7761	501.4	472.2	0.97
Myrtle	O	Mulched	58	41	1	7	7	22	23061.2	0	243.6	2.8033	520.6	514.6	0.95
Ricco	P	Unmulched	60	32	3	37	7	11	0	0	0	3.2607	1022.8	687.8	0.37
Ricco	P	Mulched	71	21	6	35	13	12	101.5	101.5	0	3.3979	1020	744.8	0.44
Ricco	Q	Unmulched	72	27	0	26	13	15	40.6	40.6	0	3.1977	894.8	780.4	0.02
Ricco	Q	Mulched	74	19	6	15	14	11	0	0	0	3.0024	884.2	753.8	0.2
Ricco	R	Unmulched	57	38	2	25	12	17	40.6	40.6	0	3.2979	898.4	844.8	0.01
Ricco	R	Mulched	66	33	1	22	15	5	40.6	40.6	0	3.105	700.4	784.8	0.17
Shake	S	Unmulched	45	27	21	27	6	3	60.9	40.6	0	2.8799	282	608	0.16
Shake	S	Mulched	40	56	1	20	12	4	385.7	101.5	101.5	2.8364	222.4	794.2	0.08
Shake	T	Unmulched	70	25	5	11	12	4	162.4	142.1	0	2.4721	445.6	743.8	0.59
Shake	T	Mulched	47	35	10	23	13	3	60.9	20.3	20.3	2.6957	177	625.8	0.05
Shake	U	Unmulched	61	35	1	18	13	2	324.8	243.6	0	2.463	531.8	794.4	0.03
Shake	U	Mulched	59	38	2	23	11	2	872.9	40.6	60.9	2.6046	164.8	557.2	0.15
Tripod	V	Unmulched	53	47	0	19	6	6	3011.2	0	121.8	2.7264	249.6	785.2	0.75
Tripod	V	Mulched	61	25	9	18	7	8	3004.5	0	284.2	2.6736	313.8	712.8	0.69
Tripod	W	Unmulched	52	15	27	25	4	2	852.6	0	20.3	2.8115	300.2	814.4	0.69
Tripod	W	Mulched	59	20	12	16	5	7	365.4	20.3	243.6	2.5903	545.6	1112.2	0.45
Tripod	X	Unmulched	38	39	19	9	0	12	2253.3	0	0	2.5402	459.4	764.6	0.58
Tripod	X	Mulched	65	33	1	10	0	10	8972.8	0	0	2.4699	470	759.8	0.97
Tripod	Y	Unmulched	75	15	0	17	0	8	3857.1	0	0	2.5592	283.6	924.8	0.34
Tripod	Y	Mulched	41	51	2	19	0	9	60.9	0	0	2.8044	521.2	931	0.54
Tripod	Z	Unmulched	29	46	16	8	4	5	223.3	0	20.3	2.3021	380.6	695	0.4
Tripod	Z	Mulched	33	48	8	12	0	4	81.2	0	20.3	1.944	425.8	811.2	0.21
Tripod	ZA	Unmulched	17	30	36	9	1	2	737.6	0	0	1.9826	226.4	661	0.73
Tripod	ZA	Mulched	48	47	1	26	1	5	2050.3	0	101.5	2.7473	538.2	682.6	0.76
Tripod	ZB	Unmulched	36	27	16	4	4	17	5278.1	0	0	2.5826	512	759.4	0.08
Tripod	ZB	Mulched	61	23	4	6	4	19	304.5	20.3	40.6	2.4431	461.4	815.2	0.49
Tripod	ZC	Unmulched	44	52	2	8	3	11	832.3	649.6	142.1	2.7288	224.6	683.2	0.11
Tripod	ZC	Mulched	39	55	4	6	7	9	6861.5	81.2	121.8	2.5999	122.6	582.8	0.01