# EFFECTS OF THINNING AND CHIPPING ON TREE REGENERATION AND UNDERSTORY PLANT COOMUNITIES IN A PONDEROSA PINE FOREST, BOULDER COUNTY, COLORADO.

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## 1. ABSTRACT OF THESIS


**EFFECTS OF THINNING AND CHIPPING ON TREE REGENERATION AND UNDERSTORY PLANT COOMUNITIES IN A PONDEROSA PINE FOREST, BOULDER COUNTY, COLORADO.**

Novel fire mitigation treatments that chip harvested biomass on site are increasingly prescribed to reduce the density and number of small diameter trees, yet the ecological effects of these innovative treatments are unknown. The purpose of my research is to investigate the impacts of mechanical thinning and chipping on ponderosa pine (*Pinus ponderosa*) regeneration and understory plant communities to guide applications of these new fuel disposal methods. I sampled vegetation in three treatments: 1) thinned forests with harvested biomass chipped and broadcast on site (TC), 2) thinned forests with harvested biomass removed (TR), and 3) unthinned forests (CO). Plots were located in a ponderosa pine forest of Colorado and vegetation was sampled 3-5 growing seasons following treatment using a modified-Whitaker plot design. First year ponderosa pine seedlings were observed most frequently in the TC treatment. Forest litter depth, augmented with chipped biomass, had a negative relationship with cover of understory plant species. I've observed that *in situ* chipping often produces a mosaic of chipped patches tens of meters in size, creating a range of woodchip depths including areas lacking woodchip cover within the TC treated forests. My data indicated that at the stand level, TR and TC treatments had similar abundance and species richness of understory plants, but at smaller spatial scales, areas within TC stands that were free of woodchip cover had an increased abundance of understory vegetation compared to all other treatments. TC treated forests also had a significantly different understory plant community composition compared to other treatments. Thinning and chipping had enhanced frequency of plants that can spread vegetatively and reduced frequency of non-rhizomatous plant species. Exotic plant species cover and richness was correlated with seasonal variation in climate and did not show any treatment effects, while native graminoid richness was positively correlated with CO treatments. I suggest that different fuel disposal methods could alter conifer regeneration and understory plant species composition in ponderosa pine forests of Colorado. When considering post-treatment responses, managers should be particularly aware of both the depth and the distribution of chipped biomass that is left in forested landscapes.

Brett Wolk
Department of Forest, Rangeland and
Watershed Stewardship
Fort Collins, CO 80523
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## 2. INTRODUCTION

Population growth in the wildland urban intermix (WUI) and large, catastrophic fires have made fuels reduction a high priority for federal, state, and local forest managers. Following a century of fire suppression in the United States, many dry pine forests have experienced dramatic increases in the density and number of small diameter trees. In ponderosa pine (*Pinus ponderosa*) forests, fire hazard reduction and forest restoration are often compatible goals achieved by reducing the quantity and homogeneity of small diameter trees and reinstating natural fire cycles (Allen et al. 2002, Kaufmann et al. 2003). However, strict air quality regulations limit the use of prescribed fire and a dearth of markets for small diameter timber discourage traditional harvesting practices, limiting management options to reduce stem densities. To overcome these implementation barriers, novel management techniques that dispose of small diameter trees by chipping harvested biomass *in situ* are now common in many conifer forest of North America.

Thinning and chipping biomass on site reduces stem densities and ladder fuels similar to other restoration treatments, but little is known about the ecological effects caused by the unique disturbance of simultaneously opening the tree canopy and adding chipped woody debris to the forest floor. In particular, information regarding tree regeneration and understory plant responses is needed to assess the value of these treatments for altering fire behavior and enhancing biodiversity, which are key components of ponderosa pine restoration (Moore et al. 1999, Kaufmann et al. 2006). Understory plant community responses to these treatments likely depend on both the degree of thinning and the depth and distribution of material added to the litter layer.

Thinning increases the vigor, abundance and diversity of understory vegetation in many ponderosa pine forests (Naumburg and DeWald 1999, Laughlin et al. 2006, Metlen and Fiedler 2006). However, chipped biomass adds to the depth of the litter layer, limiting light availability for germination and growth of new propagates (Knapp and Seastedt 1986) and potentially diminishing total cover. Plant litter generally moderates seasonal and diurnal variation in soil temperatures and increases soil moisture (Facelli and Pickett 1991). When woodchips are below saturation, they could intercept and retain more moisture than needle litter during small precipitation events, reducing soil moisture (Massman et al. 2006) and negatively effecting the understory plant community. Deep layers of woodchips also provide a physical barrier to seedling emergence, inhibiting establishment of new individuals and potentially diminishing understory plant cover and diversity.

Thinning followed by on site chipping, when applied at the operational scale of typical fuel reduction projects (e.g. tens to hundreds of hectares) usually results in areas tens of square meters in size covered with chips distributed in a mosaic pattern throughout the treatment. Over time, the distribution of woodchips likely will change the plant community composition as individual species exhibit unique responses to the uneven distribution of chipped biomass. For example, thinning operations that remove all harvested material from the site typically reduce ground cover of litter and duff and increase the amount of exposed soil within the treatment area, reducing competition and creating favorable habitat for many non-native understory herbaceous plants to invade (Dodson and Fiedler 2006). Alternatively, thinning followed by chipping harvested biomass on site reduces exposed bare soil, limiting safe sites for annual plant species and aggressive colonizers to invade. While thinning and *in situ* chipping of harvested biomass have independent and often contradicting ecological consequences, the interaction of these processes likely creates novel and unexpected ecological patterns not previously observed in managed or unmanaged forests.

In Colorado, approximately 75 percent of communities at risk from wildfire are characterized by, or lie within, one mile of ponderosa pine dominated stands (CSFS 2004). Because of the management priority to reduce fuels in the WUI and the abundance of small diameter trees, thinning and *in situ* chipping treatments are especially common in ponderosa pine dominated habitats in the Front Range of Colorado. Following several large, catastrophic fires in the early 2000s, collaborative efforts such as the Front Range Fuels Treatment Partnership formed in search of regional solutions to reduce fire danger and restore ecological functions in Colorado's forests. These organizations have encouraged forest management based on ecological principles, yet thinning and chipping treatments continue to be implemented with little to no knowledge of their ecological impacts.

The goal of this research is to provide information about conifer regeneration and responses of understory plant communities as a result of thinning and spreading chipped biomass in ponderosa pine forests along the Front Range of Colorado. In order to describe the ecological effects of thinning and chipping, I sampled understory and overstory vegetation 3-5 years post treatment in three common forest conditions along the Front Range. I hypothesized that ponderosa pine regeneration would be highest in the thinned stands with biomass removed, where light levels and exposed mineral soil favor regeneration. I also expected that thinning would increase cover of understory vegetation regardless of fuel disposal method, but increasing woodchip depths would have a negative relationship with cover of understory vegetation. Finally, I hypothesized that thinning and chipping would create unique plant community assemblages as a result of the novel habit created by these treatments.

## 3. Methods

### *3.1 Study Site*

This study was conducted at the Heil Valley Ranch (HVR), which is owned and managed by Boulder County Parks and Open Space. The HVR is located approximately 3 miles to the northwest of the city of Boulder, Colorado, and includes 1992ha of relatively undeveloped forested landscape (Figure 0). BCPOS has classified the property as one of the highest priority fire risk areas under their oversight. The geology of the area is quite complex. The most common substrate is sandstone of various ages, but siltstone, shale, claystone, pegmatite, silver plume granite, and some limestone and quaternary colluvium and alluvium all occur on the property (Kettler et al. 1996). Climate is

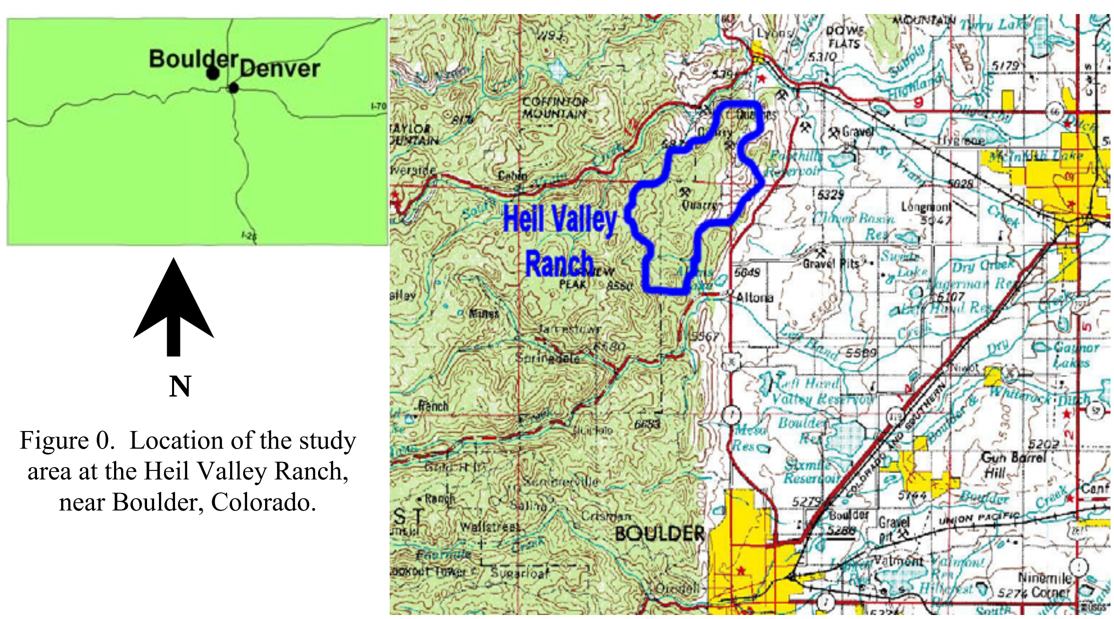


Figure 0. Location of the study area at the Heil Valley Ranch, near Boulder, Colorado.

characterized by generally cool and dry winters with warm summers. Average minimum January temperature at the City of Boulder weather station (5484ft elevation) is 20.6°F, with average highs of 45.5°F. July is the warmest month with average maximum temperatures of 87.6°F and lows of 58.6°F. Average annual precipitation is 19.14 inches and the wettest months are April and May, although precipitation is relatively well distributed throughout the growing season from early spring through late fall (Anonymous, Anonymous 2007). Vegetation at the HVR is characterized by even aged stands strongly dominated (>95%) by *Pinus ponderosa* (ponderosa pine), with occasional individuals of *Pseudotsuga menziesii* (Douglas fir) and *Juniperus scopulorum* (Rocky Mountain juniper). The understory flora is sparse overall, but heavily dominated by herbaceous species with an occasional shrub dotting the landscape. Soft surface recreational trails and an improved dirt road that is restricted to occasional motorized travel by BCPOS are located within the study site.

### *3.2 Treatments and Data Collection*

Multiple thinning treatments have been completed annually since 2000 at the HVR. The treatments are arranged with contiguous borders along a forested east facing plateau. Environmental variables are similar between all thinned stands and are characterized by gentle to moderate slopes (Average 9.5°, range 0° to 18°), generally eastern aspects and elevations ranging from 1900m to 2100m. Each thinned stand ranges in size from approximately 2 to 22 hectares, totaling 141 thinned hectares. Treatments were similar in prescription in that they stipulated complete removal of all ponderosa pine individuals under a maximum cut size of approximately six inches diameter at breast height (DBH) and no removal of larger trees. The thinning treatment prescriptions differed only in the fate of the removed biomass and are classified accordingly: *TR* - thinned stands with harvested biomass removed or chipped and locally piled (total 117ha); *TC* - thinned stands with harvested biomass chipped and broadcasting throughout the treatment area to an average depth of three inches and no greater than six inches (total 24ha); and *CO* - unthinned stands without woodchip augmentations. Thinning treatments consisted of chainsaw felling and hand crew or ATV skidding. Chipping of harvested biomass was completed with a 15in capacity Vermeer BC1500 brush chipper (or similar model) towed by a pickup truck or ATV. For the TR treatments, ATV or hand skidding was used to forward logs to designated landings where they were locally chipped.

Thirty two sampling plots (20m by 50m) were randomly located over two growing seasons (2005 and 2006) according to a stratified random sampling scheme, grouped by thinning treatment and year thinned (TC=12 plots, TR=13 plots, CO=7 plots) (Figure 2). A randomly selected subset of three plots per treatment were sampled in both 2005 and 2006, all other plots were sampled in only one year. Care was taken in plot establishment to avoid old roadbeds or historic areas of human settlement.

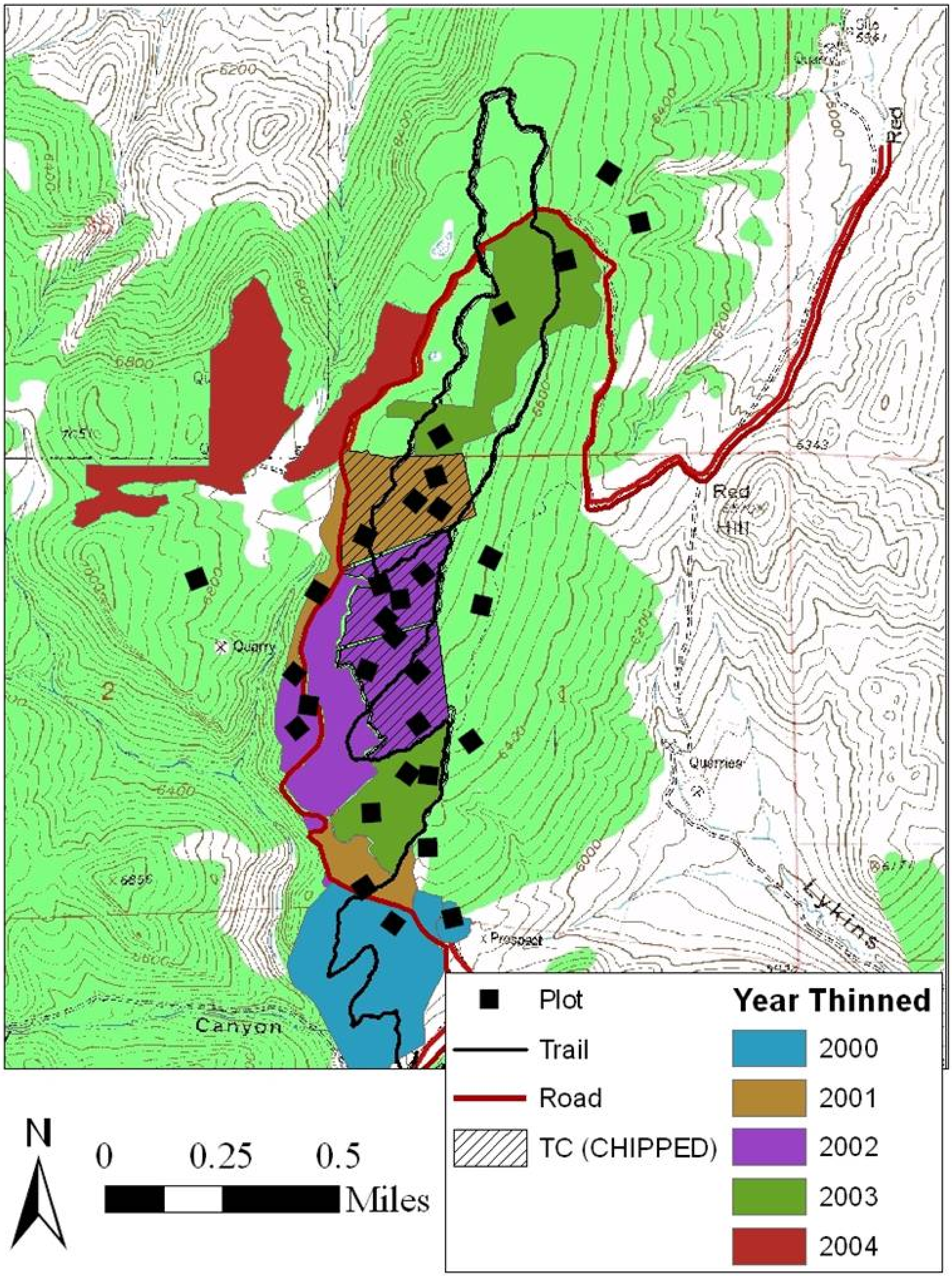


Figure 1. Study site at the Heil Valley Ranch, Boulder County, Colorado.

Vegetation was sampled using a nested intensity modified-Whittaker sampling scheme, similar to the one described in Peet et al. (1998), but with twenty $1m^2$ subplots located in each $1000m^2$ plot (Figure 2). Plants were identified to a minimum of species level and ocular estimates of percent cover for each species were recorded in all $1m^2$ sub-plots. Cumulative additional species were recorded as present in successively larger plots ($100m^2$, $400m^2$, and the entire $1000m^2$ plot). A total of 126 understory plant species were positively identified throughout the course of the study.

Auxiliary and environmental variables were measured at several spatial scales within each plot. Litter depth, duff depth, and ground cover were estimated in each $1m^2$ sub-plot. It was not possible to distinguish chipped biomass from natural litter accumulation because chipped material was applied on top of existing natural forest litter and litter accumulation has occurred since treatments were completed. Therefore, when woodchips were present, litter and duff measurements included chipped and natural litterfall. Canopy closure was measured using a spherical densitometer at four locations within each $1000m^2$ plot. Readings at these four locations were averaged to obtain a mean canopy cover estimate within each plot. Environmental characteristics (slope, aspect, elevation), and digital photos were also recorded within each $1000m^2$ plot. Species, location, DBH, and maximum crown height were measured for all live and dead trees within the $1000m^2$ plot.

I have observed that the distribution of chipped material is often heterogeneous throughout chipping treatments along the Front Range of Colorado, including areas with deeper woodchip cover tens of meters in size and areas of similar size within the chipping treatment that lack chipped material. To investigate the specific effects of the uneven distribution of chipped material, the three stand level treatments were refined to four fine scale treatment classes by grouping 1m$^2$ subplots in each of the twelve 1000m$^2$ plots within the TC treatment into one of two categories: TCCHIP or TCFREE (Figure 2). The *TCCHIP* treatment group includes measurements of species and auxiliary data in 1m$^2$ subplots that contain higher percent ground cover of woodchips compared to natural needle litter. The *TCFREE* group includes the remaining species and auxiliary data in 1m$^2$ subplots that were free of woodchips within plots located in the chipping treatment. *TR* and *CO* treatments were defined as before. In all four treatment groups, only understory plant species observed at the 1m$^2$ scale (92 species) were included in this fine scale analysis. Species recorded as present at scales larger than 1m$^2$ were excluded from all analyses of fine scale treatment effects because woodchip cover was not recorded for subplots larger than 1m$^2$. Cover values and auxiliary variables measured at the 1m$^2$ scale were averaged first by plot then by treatment, while auxiliary variables measured at the 1000m$^2$ scale were assigned at the plot level. Thus, the twelve plots in the TC treatment are partitioned according to the presence of woodchips, resulting in 44 estimates of plant cover at the 1m$^2$ scale over the thirty two 1000m$^2$ plots sampled (TCCHIP=12 plots, TCFREE=12, TR=13, CO=7).

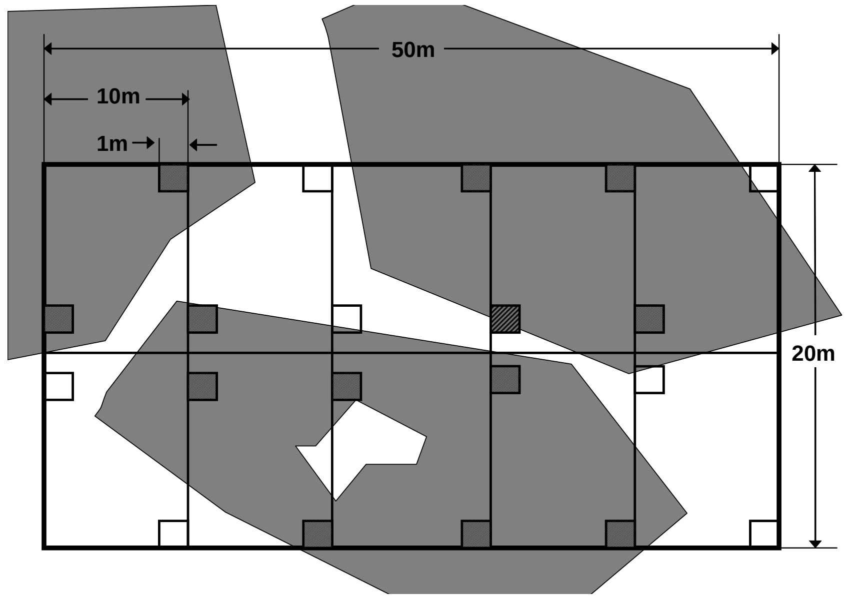


Figure 2. Sample plot design, outlined in black, is overlain on a hypothetical distribution of woodchips within a chipping (TC) treatment. = patches covered by woodchips; = 1m$^2$ subplot lacking woodchip cover (TCFREE treatment); = 1m$^2$ subplots with woodchip cover (TCCHIP treatment).

### *3.3 Data Analysis*

I performed tests on data in the main and second matrices comparing forest auxiliary data, ponderosa pine seedling frequency, total understory vegetation cover and diversity between treatments at the 1m$^2$ and 1000m$^2$ scale using a one-way analysis of variance (ANOVA). Tukey Highest Significant Difference (HSD) was used for post-hoc pairwise comparisons with $\alpha = 0.05$. Calculations were carried out using SAS v9.1 (2003) and attributes were transformed when necessary to approximate assumptions of homoscedastic variance and reduce the influence of outliers in the ANOVA analysis. A 1/sqrt(X) transformation was applied to all understory vegetation cover values, trees per hectare data were log transformed, and all auxiliary variables at the 1m$^2$ scale were square root transformed.

Within each plot, linear correlations were constructed to examine relationships between woodchip depth and cover of understory plant species. Only data in the TCCHIP treatment was included. Cover values in 1$m^2$ subplots were adjusted by multiplying the percent of total plant cover by the percent of substrate that did not support plant growth (rock and coarse woody debris). Only plots that had an $r \geq 0.1$ were included in analysis.

For multivariate analysis on changes in the understory plant community composition, data was divided into two sets: a main matrix containing plant species cover values and a second matrix containing auxiliary and environmental variables. These data categories were then divided again according to the two treatment classifications defined above to investigate fine scale (1$m^2$, four treatment groups) and stand level (1000$m^2$, three treatment groups) effects of treatments on understory vegetation. All tree species (Douglas fir [*Pseudotsuga menziesii*], ponderosa pine [*Pinus ponderosa*], and Rocky Mountain juniper [*Juniperus scopulorum*]) were deleted from the dataset. Plants that were not identified to the species level were identified as shrub, forb, or graminoid and included in total cover estimates but excluded from all other analyses (average 3.0% unidentified species per plot). In two cases, similar species were easily confused when sampling occurred outside peak phonological development. All data for plants identified as cheatgrass (*Bromus tectorum*) and field brome (*Bromus arvensis*), both annual, non-native and noxious grasses, were combined for this reason and analyzed as cheatgrass (*Bromus tectorum*). A second grouping combined all data for Ross' sedge (*Carex rossii*) and White Mountain sedge (*Carex geophila*), both perennial, native sedges, and was analyzed as carex (*Carex* spp.). All species were classified as either native or non-native, according to Webber and Whitman (1996) and the USDA NRCS Plants Database (2007). In case of a discrepancy, I defaulted to Webber and Whitman. Plant species were also classified into functional groups based on life form, annual-biennial or perennial duration, and the ability to spread vegetatively using a variety of reference sources.

For all multivariate analyses, species occurring in less than three plots were deleted from each dataset. The procedure of deleting rare species reduces the noise in the dataset by eliminating haphazard occurrences while still permitting a robust assessment of community responses to treatment and environmental variability (McCune and Grace 2002). Additionally, all species cover values in the stand level main matrix were transformed to present or absent in each plot, giving equal weight to rare and abundant species and focusing multivariate analysis at the stand level on changes in the overall plant community. Species cover values in the fine scale ($1m^2$) main matrix were left as estimates of percent cover to examine trends in abundant species and changes in plant community structure between treatments. All multivariate analyses were carried out in PCORD version 5.0 (McCune and Mefford 1999).

I performed indicator species analysis at both the fine scale and stand level to examine distributions of specific dominant and rare understory plant species. Indicator values represent the frequency and abundance of a species within a group compared to other groups, but because cover values at the stand level were transformed to present or absent, rare and abundant species were given equal weight in the analysis. For each species, a Monte-Carlo randomization test was used to calculate a p-value, which measures the proportion of randomized trials in which the indicator value equals or exceeds the observed indicator value.

To analyze changes in overall species composition at the stand level, I used multi-response permutation procedure (MRPP) to test for differences in plant community composition between the three treatments. MRPP produces an agreement statistic (*A*), which measures how similar within group (e.g. treatment) homogeneity is compared to random chance, and a significance value measuring the probability that within group agreement could be determined by chance. With MRPP I used a Sorensen distance measure and *n*/sum(*n*) as the weighting measure.

Ordination of sample plots was performed on species presence/absence data at the stand level using nonmetric multidimensional scaling (NMS) to display relationships of plant community composition between treatments. NMS finds the best solution to minimize stress through multiple iterations based on ranked values in a multi-dimensional distance matrix. Calculations were performed using Sorensen distance measure and a random starting configuration. A Monte Carlo randomization test comparing stress reductions between 50 runs with real data and 50 runs with randomized data was used to determine the optimal number of dimensions to retain in the ordination. Stability was assessed graphically by comparing stress vs. iteration number to ensure levels stabilized and plateau in later iterations. It was determined that a three dimensional solution would provide optimal dimensionality for minimizing stress, reducing instability, and allowing ecological interpretation of results. A final NMS ordination was then performed using Sorenson distance measure and starting coordinates for plots from the previous NMS procedure in order to achieve stability and aid in finding the true minimum rather than reaching stability in localized minima. A joint plot was overlain on the ordination to describe significant ($r^2 \geq 0.2$) associations of auxiliary and environmental variables with variation in plant community composition between plots.

The ordination was then rotated to align with the auxiliary or environmental variable that explained the most variation among plots.

## 4. RESULTS

T-tests on stand level data for plots sampled in both growing seasons (3 plots per treatment, 18 plot-years) showed no differences in species cover ($p = 0.29$) or richness ($p = 0.66$) between years. Additionally, an MRPP test on presence/absence data in the re-sampled plots found no differences in overall species composition between years ($A$= 0.017, p-value= 0.151). Therefore, data for all thirty two plots were combined over sampling years and analyzed together (

Table 1). For plots that were sampled in both years, only one plot-year (either 2005 or 2006) was randomly selected to include in analysis, within the bounds of approximating equal sample sizes among treatments between years.

Table 1. Plots included in analysis by treatment and year sampled.

| | Year Sampled | | |
|---|---|---|---|
| | 2005 | 2006 | Total |
| CO | 3 | 4 | 7 |
| TC | 6 | 6 | 12 |
| TR | 7 | 6 | 13 |
| Total | 16 | 16 | 32 |

### *4.1 Treatment Effects on Habitat*

Compared to CO treatments, both TC and TR treatments significantly reduced canopy closure (50% to ~30%), trees per hectare (1750 to ~525), and basal area ($33m^2$/ha to ~$20m^2$/ha). As would be expected, TCCHIP treatments had significantly deeper average litter depths compared to all other treatments. Plots in the TCCHIP treatments also had the lowest percent cover of bare ground and the highest average duff depths ( Table 2).

Table 2. Mean values of forest characteristics in treatments at the $1m^2$ scale and the $1000m^2$ scale. Values within a row are not significantly different ($\alpha < 0.05$) unless marked with different letters (Tukey HSD analysis of the one-way ANOVA).

| | CO | TR | TC | TCCHIP | TCFREE |
|---|---|---|---|---|---|
| Canopy Cover (%) | 50.4 **a** | 32.1 **b** | 28.8 **b** | -- | -- |
| Trees per hectare | 1749 **a** | 529 **b** | 520 **b** | -- | -- |
| Basal Area ($m^2$/ha) | 33.4 **a** | 20.9 **b** | 18.4 **b** | -- | -- |
| Bare ground (%) | 11.6 **ab** | 12.6 **a** | -- | 6.4 **b** | 10.3 **ab** |
| Litter depth (cm) | 1.6 **b** | 1.5 **b** | -- | 3.6 **a** | 1.9 **b** |
| Duff depth (cm) | 1.0 **ab** | 0.8 **b** | -- | 1.5 **a** | 0.7 **b** |

### *4.2 Ponderosa Pine Regeneration*

No seedlings were present in the TR treatment plots, while only one individual seedling was recorded in one CO plot. Due to small sample sizes, there was no statistical difference between TC and CO at $\alpha < 0.05$ using the conservative Tukey HSD adjustment for multiple comparisons. However, first year ponderosa pine seedlings were far more frequent in the TC treatment compared to either TR ($p = 0.008$) or CO ($p = 0.036$). There was no difference in the percent of subplots per plot that contained seedlings between TCFREE and TCCHIP ($p = 0.62$).

### *4.3 Total Cover and Dominant Species Trends*

All plots had an overall negative relationship between increasing woodchip depth and decreasing cover of understory species within each plot (Figure 3), average slope for all plots = -0.15). The average y-intercept was 4.6cm, which is the average woodchip depth at which understory vegetation is reduced to zero. The range of average suppression depths varies between plots from 3.1 to 6.5cm.

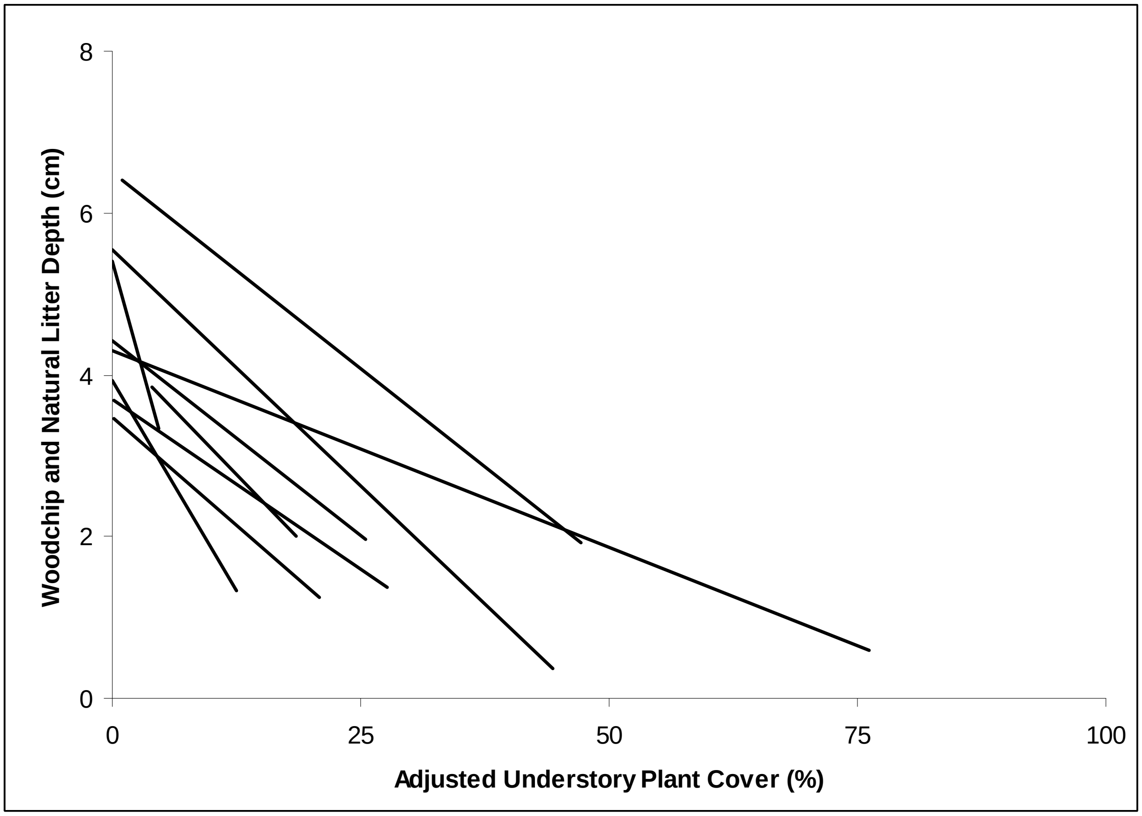


Figure 3. Each line is a linear correlation between adjusted understory species cover and woodchip depth within a plot. Only plots within the TCCHIP treatment that had an $r < 0.1$were included in analysis.

TCFREE had the highest cover values compared to all other treatments and was significantly different from CO (Figure 4). Average cover was not significantly different between any other treatments using a conservative Tukey HSD analysis of an ANOVA, although there was a clear trend of higher cover in the thinned areas and elevated cover in the TCFREE treatment.

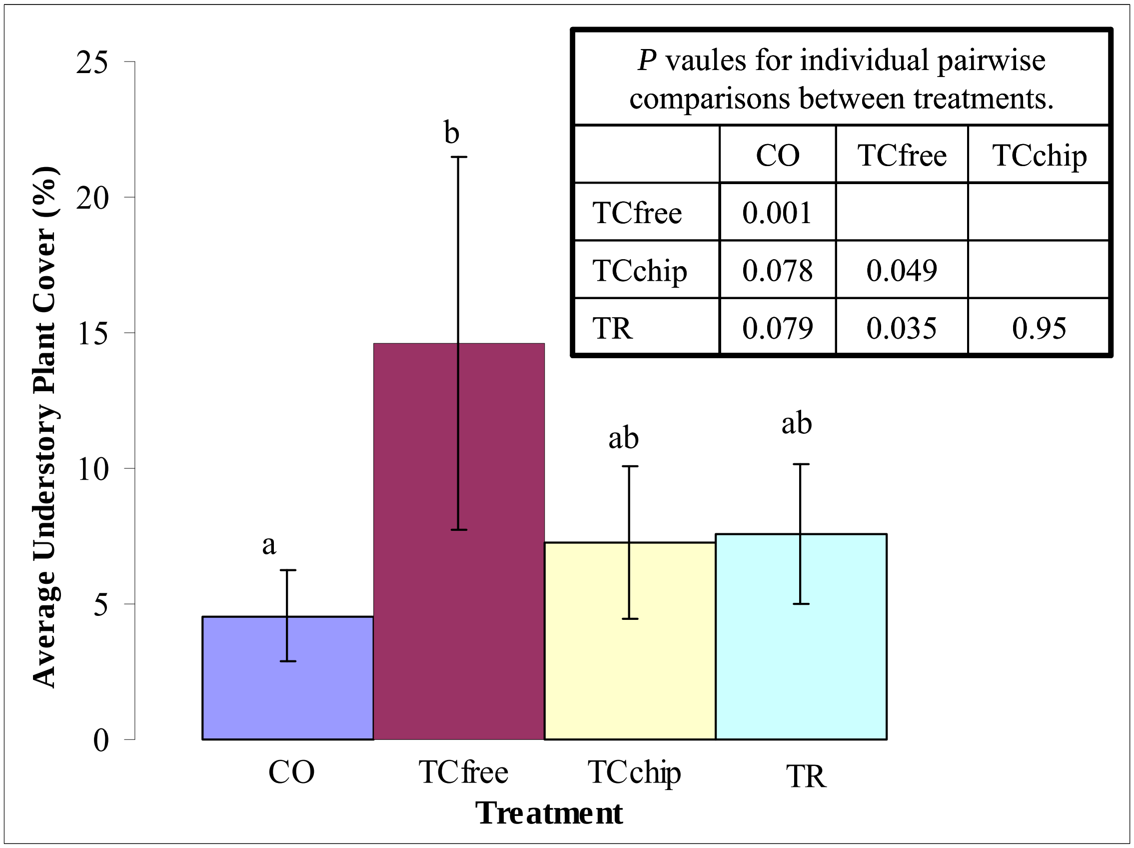


Figure 4. Average cover values between treatments. Sample sizes: CO (7 plots, 140 $1m^2$ subplots); TCFREE (12, 74, range 3 to 10 $1m^2$ subplots per plot), TCCHIP (12, 166, range 10 to 17); TR (13, 260). Values within a row are not significantly different ($\alpha < 0.05$) unless marked with different letters. Statistical differences were determined using Tukey HSD analysis of an ANOVA.

A total of 126 understory plants were positively identified to the species level throughout my study. For multivariate analysis, deletion of species that occurred in fewer than three plots resulted in a main species matrix of fine scale ($1m^2$) data containing 44 plots and 68 species (24 species deleted). At the stand level ($1000m^2$ scale), 94 species remained on 32 plots after 30 species were deleted.

Indicator species analysis identified eight species that were significantly ($\alpha < 0.1$) associated with one of the four treatments and had an indicator value above 30 based on cover values averaged by plot then by treatment (Table 3). *Opuntia fragilis (*brittle pricklypear) was the only species indicative of CO treatments. Plots within the TR treatment were associated with the highest abundances of *Harbouria trachypleura* (whiskbroom parsley), *Bromus tectorum* (cheatgrass), *Elymus elymoides* (squirreltail), *Phacelia heterophylla* (varileaf phacelia), and *Penstemon virens* (Front Range beardtongue). *Danthonia spicata* (poverty oatgrass), the most abundant species in all treatments at the HVR, as well as *Luzula parviflora* (smallflowered woodrush), were significant indicators of the TCFREE treatment. No species were significantly associated with the TCCHIP treatment.

Table 3. Indicator species, based on average cover values that are significantly ($\alpha < 0.1$) associated with treatments at the HVR and have an indicator value above 30. Indicator values can range from 0 to 100. Large indicator values indicate a species is both very frequent and has a higher average cover in that treatment compared to other treatments.

| Treatment | Species | Indicator Value | Indicator p-value |
|---|---|---|---|
| CO | *Opuntia fragilis* | 33.3 | 0.085 |
| | | | |
| TR | *Harbouria trachypleura* | 54.6 | 0.004 |
| | *Bromus tectorum* | 50.3 | 0.034 |
| | *Elymus elymoides* | 43.6 | 0.063 |
| | *Phacelia heterophylla* | 37.6 | 0.059 |
| | *Penstemon virens* | 34.6 | 0.063 |
| | | | |
| TCCHIP | None | | |
| | | | |
| TCFREE | *Danthonia spicata* | 57.6 | 0.032 |
| | *Luzula parviflora* | 32.8 | 0.08 |

*4.4 Plant Community Responses*

The number of species per plot was not significantly different between treatments at the $1m^2$ scale ($p = 0.11$) or the $1000m^2$ scale ($p = 0.81$) (Table 4). TCCHIP had the lowest species richness among treatments at the $1m^2$ scale, while species richness was highest in the CO treatment at the stand level ($1000m^2$ scale).

Table 4. Species diversity trends at two spatial scales. No significant differences (α < 0.05) were found between treatments at either scale (Tukey HSD analysis of a one-way ANOVA).

| | CO | TR | TC | TCCHIP | TCFREE |
|---|---|---|---|---|---|
| Average species per $1000m^2$ plot | 49.6 | 47.4 | 48.3 | -- | -- |
| Average species per $1m^2$ subplot | 4.1 | 4.9 | -- | 3.6 | 4.9 |

Overall plant community composition at the stand level was significantly different between the three treatment groups as indicated by MRPP ($A$ = 0.046, p = 0.0006). Pairwise comparisons suggest there was a marginally significant difference between species composition in the TR and CO treatments ($A$ = 0.021, p = 0.071), but the TC treatment was very different from both CO ($A$ = 0.064, p = 0.0005) and TR ($A$ = 0.027, p = 0.009). Fifteen species were identified as significantly associated (α < 0.1) with one of the three treatments and had an indicator value above 30 based on indicator species analysis of presence/absence data at the stand level (Table 5). Seven native species were indicators of plots in the CO treatment, including four perennial graminoids and the perennial forbs *Yucca glauca* (soapweed yucca) and *Liatris punctata* (dotted blazing star), and the shrub *Rhus trilobata* (skunkbush sumac). TR treatments were generally associated with the highest cover of *Poa compressa* (Canada bluegrass) and more occurrences of *Juniperus communis* (common juniper) compared to other treatments. Five of the six species indicative of TC treatments have high vegetative spread rates, including *Luzula parviflora* (smallflowered woodrush) and *Cirsium arvense* (Canada thistle), which showed the highest species indicator values in this treatment. *Luzula parviflora* was the only species that was a significant indicator in both fine scale and stand level indicator species analyses.

Table 5. Indicator species, based on presence/absence values, that are significantly ($\alpha < 0.1$) associated with treatments at the HVR and have an indicator value above 30.

| Treatment | Species | Indicator Value | Indicator p-value |
|---|---|---|---|
| CO | *Elymus albicans* | 51 | 0.025 |
| | *Rhus trilobata* | 49.9 | 0.014 |
| | *Poa fendleriana* | 48.9 | 0.039 |
| | *Hesperostipa comata* ssp. *comata* | 47.7 | 0.009 |
| | *Yucca glauca* | 44.8 | 0.046 |
| | *Dichanthelium oligosanthes* var. *scribnerianum* | 39 | 0.037 |
| | *Liatris punctata* | 39 | 0.054 |
| | | | |
| TR | *Poa compressa* | 37.5 | 0.028 |
| | *Juniperus communis* | 31.6 | 0.09 |
| | | | |
| TC | *Luzula parviflora* | 54.4 | 0.003 |
| | *Cirsium arvense* | 53.2 | 0.006 |
| | *Solidago simplex* | 43 | 0.095 |
| | *Agrostis scabra* | 41.8 | 0.019 |
| | *Poa pratensis* | 38.7 | 0.096 |
| | *Arctostaphylos uva-ursi* | 38.2 | 0.04 |

The NMS ordination used 82 iterations to produce a solution with a final stress of 16.1 and instability of 0.00048. The ordination of understory species captured 77% of the variation in the original dataset with the first three axes. The joint plot with Axes 1 and 2 (37% and 24% respective variation explained) shows sample units in the TC and TR treatment separated from CO, while there was little separation between TC and TR treatments. Significant correlations ($r^2 \geq 0.2$) with auxiliary variables are displayed by red vectors, which are drawn proportional to the magnitude of correlation (Figure 5).

Year sampled was most strongly correlated to the distribution of plots based on species composition (Axis 2, r = -0.792) (Table 6). Exotic species diversity and cover, annual and biennial species diversity and cover, and cover of forbs and species that spread vegetatively were all greater in 2005 than 2006. However, sample sizes were nearly equal within treatments among years and none of these variables were strongly correlated with any treatment group when data was analyzed in plots sampled only in 2005 or 2006 (data not shown). Depth of forest floor biomass (needle litter, duff, and/or woodchips) was not well correlated with the distribution of sample plots. However, increasing cover of woodchips was positively correlated with Axis 2 (r = 0.51), indicating a treatment effect from the presence of chipped material. Graminoid diversity and cover showed divergent trends, as diversity was positively correlated with CO plots, while average graminoid cover was greatest in the thinned plots regardless of fuel disposal method. Other than sampling year, distance from thinning treatment edge explained the most variation among plots as represented by strong correlations of elevation (r = 0.536) and UTM East (r = -0.481) with Axis 2 in the NMS ordination. Treatments grouped by year since thinning did not show any interpretable trends and was weakly correlated with the first three Axes.

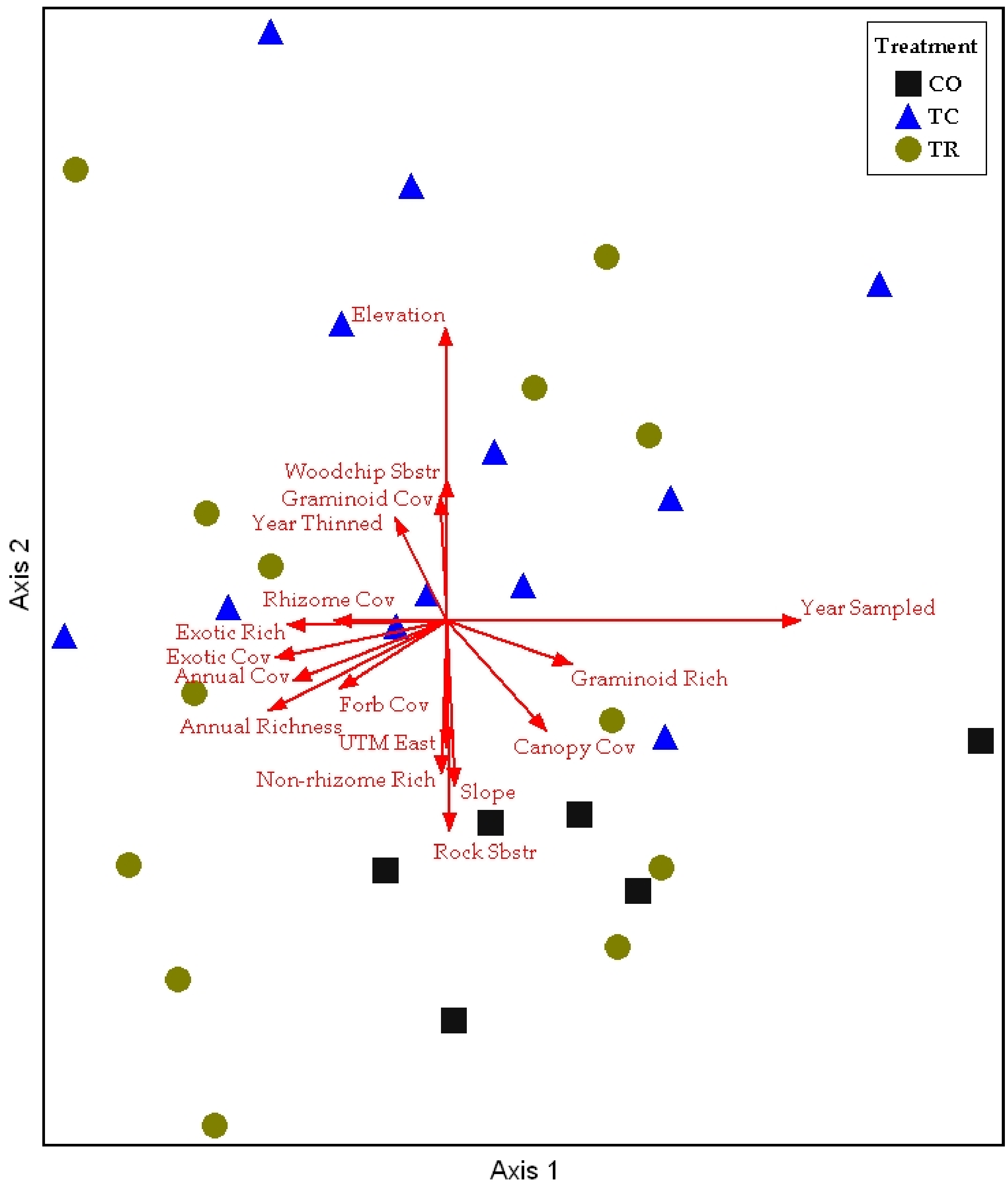


Figure 5. Joint plot of a nonmetric multidimensional scaling ordination of species presence/absence values at the stand level. Scores for plots in the CO (■), TC (▲), and TR (●) treatments are plotted on Axis 1 and 2 of the ordination. Auxiliary variables or plant functional group cover and richness that were strongly correlated ($r^2 > 0.2$) with ordination axis are indicated by red line vectors, where the vector length is proportional to the correlation strength. Abbreviations used for correlated variables are: Cov = cover, Sbstr = substrate, UTM = Universal Transverse Mercator.

Table 6. Pearson correlation coefficients indicating relations between ordination axes with auxiliary variables or plant functional types. Variables with strong correlations (r ≥ 0.5) are highlighted in bold font.

| Variable | Axis 1 | Axis 2 | Axis 3 |
|---|---|---|---|
| Auxiliary Variables | | | |
| Treatment | -0.33 | 0.16 | -0.23 |
| **Year Sampled** | **0.80** | 0.02 | 0.02 |
| Year Thinned | -0.31 | 0.43 | -0.03 |
| Aspect | -0.21 | 0.42 | -0.06 |
| **Slope** | 0.12 | **-0.55** | -0.03 |
| **Elevation** | -0.05 | **0.73** | 0.10 |
| UTM_E | 0.03 | -0.48 | 0.20 |
| UTM_N | -0.14 | 0.19 | 0.33 |
| Habitat | | | |
| Canopy Cover | 0.43 | -0.45 | -0.15 |
| Quadratic Mean Diameter | -0.28 | 0.20 | -0.21 |
| Basal Area | 0.33 | -0.40 | -0.16 |
| Trees $Ha^{-1}$ | 0.32 | -0.32 | 0.07 |
| Duff Depth | 0.14 | 0.16 | -0.08 |
| Litter Depth | 0.29 | 0.34 | 0.20 |
| **Woodchip Substrate** | -0.01 | **0.51** | 0.28 |
| Litter Substrate | 0.01 | -0.44 | -0.27 |
| Mineral Soil Substrate | -0.06 | -0.29 | 0.07 |
| **Rock Substrate** | 0.08 | **-0.62** | -0.23 |
| Plant Functional Group Cover | | | |
| **Annual Cover** | **-0.53** | -0.33 | -0.23 |
| Perennial Cover | -0.22 | 0.33 | 0.38 |
| Forb Cover | -0.44 | -0.35 | -0.03 |
| Graminoid Cover | -0.11 | 0.48 | 0.38 |
| Native Cover | -0.21 | 0.33 | 0.37 |
| **Exotic Cover** | **-0.56** | -0.26 | -0.08 |
| Non-rhizomatous Cover | -0.13 | 0.25 | 0.34 |
| Rhizomatous Cover | -0.46 | 0.02 | 0.01 |
| Total Understory Cover | -0.30 | 0.25 | 0.32 |

Continued on following page…..

| | | | |
|---|---|---|---|
| Plant Functional Group Species Richness | | | |
| **Annual Richness** | **-0.57** | -0.41 | -0.04 |
| Perennial Richness | 0.26 | -0.24 | 0.45 |
| Forb Richness | -0.35 | -0.40 | 0.35 |
| Graminoid Richness | 0.48 | -0.28 | 0.12 |
| Native Richness | 0.23 | -0.38 | 0.41 |
| **Exotic Richness** | **-0.54** | -0.09 | 0.06 |
| **Non-rhizomatous Richness** | -0.10 | **-0.53** | 0.23 |
| **Rhizomatous Richness** | 0.08 | 0.18 | **0.52** |
| Total Understory Richness | -0.07 | -0.40 | 0.36 |

## 5. DISCUSSION AND CONCLUSIONS

While thinning and chipping significantly increased the depth of forest floor litter compared to no management action or thinning and removing trees off site, understory plant cover and diversity was not reduced in chipping treatments at fine or stand level spatial scales 3-5 years post treatment.

### *5.1 Ponderosa Pine Regeneration*

Ponderosa pine regeneration along the Front Range of Colorado is episodic and depends on coincident years of favorable seed crops and local moisture availability related to broad-scale climatic variation (League and Veblen 2006, Shepperd et al. 2006). I found that seedlings were most abundant in the chipped treatments, contradicting my hypothesis that opening the tree canopy would increase ponderosa seedling establishment regardless of fuel disposal method. Chipped and unchipped plots in the TC treatment contained similar amounts of ponderosa pine seedlings, indicating that substrate type was unimportant and suggesting seedling germination was enhanced do to stand level effects within the chipping treatments. My observations disagree with other research that suggest ponderosa pine seedling establishment along the Front Range is higher on scarified soils compared to areas covered with natural forest litter (Shepperd et al. 2006). Initial results indicate chipping lowers densities of small mammals and rodents (Marchand et al. 2006), which could lead to reduced seed predation and increase the seedfall available for germination (Shepperd et al. 2006). However, whether this trend of reduced densities of small animals persists more than one year post treatment has not been determined. Significant amounts of woodchips applied in small experimental plots reduce daily and annual fluctuations in soil temperature and increase soil moisture (Binkley et al. 2003, Massman et al. 2006), which enhances germination conditions for ponderosa seedlings, but whether these trends are amplified, reversed, or unchanged when entire forest stands are covered with a mosaic of chipped biomass remains unknown. If the observed enhanced germination rates in chipping treatments lead to increased survivorship of ponderosa seedlings, thinning and chipping could lead to prolific regeneration. Seedlings were only observed in the final year of this study and I could not determine if chipping treatments increase survivorship.

### *5.2 Total Cover and Dominant Species Trends*

Deciding what depth of chipped biomass to leave on site is a big concern to managers due to direct impacts on understory plant recovery, tree regeneration, fire behavior, and aesthetic value of the treatments. Woodchips significantly augmented litter depths, resulting in suppression of understory vegetation (Figure 3). Areas with low total understory production showed no relationship between plant cover and woodchip depth, indicating that localized site characteristics, such as poor soils or species composition, could also be important in limiting understory vegetation productivity at my study site. Organic litter generally has a negative relationship with understory plant cover (Xiong and Nilsson 1999), but deep litter and duff depths in ponderosa pine forests are often strongly correlated with dense forest cover and/or disruption of natural fire cycles, which also suppresses understory vegetation (Covington and Moore 1994, Naumburg and DeWald 1999, Gildar et al. 2004). Because thinning and chipping increases litter depth while removing overstory trees, my observations are robust to support results in other ponderosa pine forests that deep litter layers indeed suppress understory vegetation.

At the stand level, both thinning treatments increased total understory cover compared to unthinned stands regardless of fuel disposal method (Figure 4), supporting my hypothesis. Stands with biomass removed, a measure of the effect of thinning on understory cover, had the same amount of understory vegetation as thinned areas covered with woodchips, indicating that average woodchip depths at the HVR were moderate and did not suppress total understory plant cover 3-5 years post treatment. Elevated levels of total plant cover, dominated by perennial graminoids (Figure 5, Table 3) in the TCFREE treatment, indicate that the spatial application of chipped material is perhaps more important than woodchip depth in determining understory plant cover. Chipped areas add to the texture of the forest floor, which increases retention of wind dispersed seeds. These additional seeds are likely transported through the litter layer to the soil during rain events when conditions for germination and establishment would be favorable. Facilitated recruitment of plants on the margins of woodchip patches where moderate depths of woodchips occur would increase propagule pressure and understory plant cover in the surrounding area. Benefits of increased soil moisture and moderated soil temperatures near chipped areas could also lead to greater cover of understory plants, but as discussed earlier, stand level effects on soil properties within the matrix of woodchip distributions is unknown. While differences in plant cover between treatments were marginally statistically significant using a conservative test, trends appear clear and I would expect that if a variety of size classes were removed understory, vegetation would exhibit a similarly large increase in cover (Laughlin et al. 2006). Low environmental variability within the study site make it unlikely that treatment areas selected for thinning and chipping are significantly more productive than other stands, refuting an alternative conclusion that woodchips suppressed understory vegetation following thinning at the HVR. Coarse scale pre-treatment data indicated relatively equal amounts of understory vegetation between TR and TC treatment areas (BCPOS, unpublished data). Clearly there is a need to assess plant dispersal mechanisms and effects on soil properties at the operational scale of chipping treatments in order to affirm mechanisms for the increase in vegetation within chipping treatments observed in this study.

### *5.3 Plant Community Responses*

Thinning and chipping significantly altered the composition of understory plant species by enhancing the frequency of plants that can spread vegetatively (Table 5) and reducing plants that lack rhizomes (Table 6, Figure 5 and Figure 6). Exotic plant species cover and diversity was dependent on seasonal variation in climate and did not show any treatment effects. However, strong differences were observed between treatments in individual species distributions and plant functional groups defined by life form and life history traits. Five of six indicator species in the thinned and chipped treatments had significant vegetative spread rates (Table 5), and non-rhizomatous species were negatively correlated with chipped forests (Table 6, Figure 5), indicating that rhizomatous species were more prevalent in the chipped areas compared to other treatments at the stand level. I've frequently observed plants with high vegetative spread rates growing horizontally through the top portion of the woodchip layer (Figure 7), while plants that lack vegetative spread are typically absent from heavily chipped areas. Although causal mechanisms could not be determined from this observational study, the small, uniform physical structure of woodchips creates a tight mat of litter that likely impedes soil-seed contact of new propagules, provides a substantial mechanical barrier to seedling emergence, and reduces light availability for seeds, resulting in suppression of seed reproduction (Facelli and Pickett 1991). Additionally, significant amounts of fungal decomposers can result in compaction of forest litter layers (Facelli and Pickett 1991), and initial findings in a central Colorado forest indicate elevated levels of fungus in chipped areas (Marchand et al, 2006). Plants that spread vegetatively through the upper layer of woodchips receive more light at the base of their tillers and likely have more energy available to allocate towards reproduction than germinated seedlings, which must expend significant amounts of energy to germinate and penetrate through deep litter layers (Knapp and Seastedt 1986). Areas that received no manipulations of thinning or chipping had the highest diversity of native graminoids (Figure 5, Table 5), indicating that some species are more competitive in dense stands of ponderosa pine (Naumburg and DeWald 1999) or that dispersal of these species into the managed areas occurs on time scales longer than the course of this study.

Figure 6. Chipping treatment at the HVR. Notice the *Cirsium arvense* flowering in the bottom left of the photo growing in the heavily chipped area, while *Danthonia spicata* and *Luzula parviflora* are very abundant on the margins and interspaces of the chipped areas.

Figure 7. *Scutellaria brittonii* spreading vegetatively through the top layer of woodchips that are two inches deep.

*5.4 Management Implications*

Forest management objectives to increase thinning in ponderosa pine forest under economic and logistical constraints have resulted in implementation of a wide variety of new mechanical thinning techniques to chip small diameter trees on site. *In situ* disposal of small diameter trees can generally be categorized into two categories based on the type of equipment used: 1) chipping, as described in this paper, and 2) mastication or chunking, which includes treatments completed using a HydroAxe with a rotary ax mower, tracked Timbco with a Fecon head, or similar heavy machinery. These two methods are often both lumped and called chipping treatments in the literature and the field, but observations made in this study suggest that the ecological effects of the two treatments could be drastically different. Chipping produces uniformly shaped, small (~1/2 to ~3 inch on a side) woody debris typically deposited in clumps of various sizes that are heterogeneously spaced throughout the treatment as determined by the forester. In contrast, mastication produces many size classes of woody debris (splinters to portions of intact tree boles) that are randomly scattered throughout the treatment area. While chipping treatments typically reduce exposed bare soil and form a tight mat on top of the existing forest floor, mastication equipment throws woody debris at the ground with great force, often mixing woody biomass with soil and creating new areas of exposed soil. My study on chipping suggest that these differences in the size, distribution, and soil mixing properties between chipping and mastication likely lead to different impacts on tree regeneration and the understory plant community. Future research should focus on describing ecological differences between forest management practices with careful consideration of the exact equipment and methods used.

The present study provides evidence that thinning followed by *in situ* chipping, thinning and removing biomass from the site, and taking no forest management action each have unique consequences on tree regeneration and the understory plant community in a ponderosa pine forest in the Front Range of Colorado. If promoting ponderosa pine regeneration is a treatment goal, thinning and chipping could enhance stocking levels compared to other treatments. However, in order to maintain fuel reduction benefits of the treatment, chipping should be followed by prescribed fire to kill the enhanced regeneration. Prescribed fire in chipped areas of loblolly pine flatwoods in South Carolina shows promise (Glitzenstein et al. 2006), but we are in desperate need of information on prescribed fire effects in chipped ponderosa pine forests.

Often thinning and chipping management plans only stipulate average and maximum woodchip depths, but my results suggest that understory plant cover and composition responds to both the depth and distribution of chipped biomass. Therefore, future prescriptions should stipulate specific distributions of chipped material to maximize desired effects on understory plants. For example, homogenous distribution of deep woodchip layers applied throughout a stand would likely have detrimental effects on the understory plant community. However, guided localized application of deep, homogenous woodchips layers could reduce local infestations of short lived non-rhizomatous noxious weeds. While individual species and several functional groups of understory plants differed in abundance and diversity between treatments, cover of noxious weeds was similar between treatments. However, careful consideration should be made to monitor the occurrence of Canada thistle and other rhizomatous noxious weeds in chipping treatments and cheatgrass in thinned stands where the biomass is removed. I urge caution in extrapolating results from this study to other locales outside the study area, especially observed individual species responses to treatments, until regional assessments of chipping are completed. Thinning and chipping is a novel forest management tool that remains poorly understood and there are likely unexpected consequences in different ecosystems and forest types that were not described in this study. If results of this study are robust to other locations and forest types, careful control of the quantity and distribution of chipped biomass could enhance ponderosa pine regeneration, reduce cover of cheatgrass or similar undesirable species, and increase the spatial heterogeneity and overall cover of understory vegetation within the treatment area. In order to maximize diversity and resiliency of understory plant communities on the landscape, a variety of management approaches should be implemented in ponderosa pine forests along the Front Range of Colorado, including the three treatment options evaluated in this study.

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