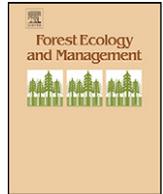




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# Thinning and chipping small-diameter ponderosa pine changes understory plant communities on the Colorado Front Range

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### ABSTRACT

Novel fire mitigation treatments that chip harvested biomass on site are increasingly prescribed to reduce the density of small-diameter trees, yet the ecological effects of these treatments are unknown. Our objective was to investigate the impacts of mechanical thinning and whole tree chipping on *Pinus ponderosa* (ponderosa pine) regeneration and understory plant communities to guide applications of these new fuel disposal methods. We sampled in three treatments: (1) unthinned forests (control), (2) thinned forests with harvested biomass removed (thin-only), and (3) thinned forests with harvested biomass chipped and broadcast on site (thin + chip). Plots were located in a ponderosa pine forest of Colorado and vegetation was sampled three to five growing seasons following treatment. Forest litter depth, augmented with chipped biomass, had a negative relationship with cover of understory plant species. *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 thinned and chipped forest stands. Thin-only and thin + chip treatments had similar overall abundance and species richness of understory plants at the stand scale, but at smaller spatial scales, areas within thin + chip treatments that were free of woodchip cover had an increased abundance of understory vegetation compared to all other areas sampled. Relative cover of non-native plant species was significantly higher in the thin-only treatments compared to control and thin + chip areas. Thin + chip treated forests also had a significantly different understory plant community composition compared to control or thin-only treatments, including an increased richness of rhizomatous plant species. We suggest that thinning followed by either chipping or removing the harvested biomass could alter 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.

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## 1. Introduction

Fuels reduction treatments are becoming widespread on federal, state, and local lands across the western United States. Fire hazard reduction treatments continue to be completed throughout the ever-expanding wildland–urban interface to reduce the risk of home, property, and infrastructure damage from wildfire. Also, managers are increasingly prescribing forest restoration treatments in many western forests to return them to a more ecologically appropriate and sustainable condition; many of these forests have experienced dramatic increases in the density of small-diameter trees and in the occurrence of large stand-

replacing wildfires due, in part, to a century of fire suppression. In *Pinus ponderosa* (ponderosa pine) forests, fire hazard reduction and forest restoration are often compatible goals that can be achieved by removing small trees and by reinstating natural fire cycles (Allen et al., 2002; Kaufmann et al., 2003). Management options for reducing stem densities remain limited, however. Strict air quality regulations and the risk of fire escape restrict the use of prescribed fire, and a dearth of markets for small-diameter timber makes traditional harvesting practices economically impracticable.

To overcome these barriers to implementing fuels treatments, a variety of novel management techniques that dispose of small-diameter trees by shredding (masticating) or chipping harvested biomass *in situ* are now commonly applied throughout conifer forests of North America (Six et al., 2002; Stephens and Moghaddas, 2005; Fettig et al., 2006; Glitzenstein et al., 2006). According to a 2005 survey of forest managers, chipping and

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mastication treatments impact tens of thousands of acres each year in Colorado alone (S.G. Resh, U.S. Forest Service Rocky Mountain Research Station, personal communication). Whole tree chippers are one of the most widely available forestry tools used for mulching small-diameter trees, yet little information is available on the ecological impacts of broadcast chipping in wildland settings.

Thinning and chipping biomass on site reduces stem densities and ladder fuels in a manner similar to other mechanical fire mitigation treatments, but the impact of simultaneously opening the tree canopy and adding chipped woody debris to the forest floor is likely to have unique impacts on conifer regeneration and understory vegetation. Understory plant community responses to thinning treatments likely depend on how much of the overstory canopy cover is removed. Mitchell and Bartling (1991) found that understory vegetation production increased as ponderosa pine canopy cover decreased along the Front Range of Colorado. In other ponderosa pine forests, thinning has been found to increase the abundance and species richness of herbaceous understory vegetation (Naumburg and DeWald, 1999; Laughlin et al., 2006; Metlen and Fiedler, 2006), but not always (Metlen et al., 2004). By facilitating recovery of the herbaceous layer, thinning often helps to achieve restoration objectives in many ponderosa pine forests (Covington et al., 1997; Allen et al., 2002). However, 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, creating favorable habitat for many aggressively colonizing understory herbaceous plants, including non-native invaders (Dodson and Fiedler, 2006). Given the prevalence of non-native species in lower elevation *P. ponderosa* forests prior to restoration treatments (Fornwalt et al., 2003), it is imperative that managers understand how non-native species respond to new management practices.

While opening forest canopies tends to promote vigorous understory growth, adding chipped biomass to the forest floor litter layer may reduce light and substrate availability, limiting opportunities for germination and growth of new propagules (Knapp and Seastedt, 1986). On the other hand, chipped biomass may increase soil moisture retention and moderate seasonal and diurnal variation in soil temperatures (Gower et al., 1992; Greenly and Rakow, 1995), which creates more favorable growing conditions for plants that are able to establish. Over time, the addition of chipped material to the forest floor is likely to change the understory plant community composition as individual species uniquely respond to its presence. After the eruption of Mt. St. Helens in 1980, Antos and Zobel (1985a,b) found that the ability to reproduce vegetatively was an important characteristic of plants that were successful following burial from tephra and volcanic ash. Additionally, Lezberg et al. (1999) found that rhizomatous species may be able to persist in dense coniferous forests over long periods of time better than species that lack the ability for vegetative spread. These studies suggest that the simultaneous disturbances of adding chipped biomass to the forest floor and opening up the tree canopy may create a unique set of circumstances inhibiting some species and favoring others in the understory community.

We examined conifer regeneration and responses of understory plant communities to thinning and spreading chipped biomass in ponderosa pine forests three to five years following treatment in the Front Range of Colorado. We hypothesized that thinned stands with harvested biomass removed, where light levels and exposed mineral soil favor regeneration, would contain more ponderosa pine seedlings than other treatments. We also expected that thinning would increase the cover and species richness of understory vegetation under each fuel

disposal method compared to unthinned forests, and that increasing woodchip depths would decrease cover and species richness of understory vegetation. Finally, in thinned and chipped treatments we expected unique plant community assemblages compared with other treatments, characterized by increased relative cover and richness of rhizomatous species and reduced non-native species abundance.

## 2. Methods

### 2.1. Study site

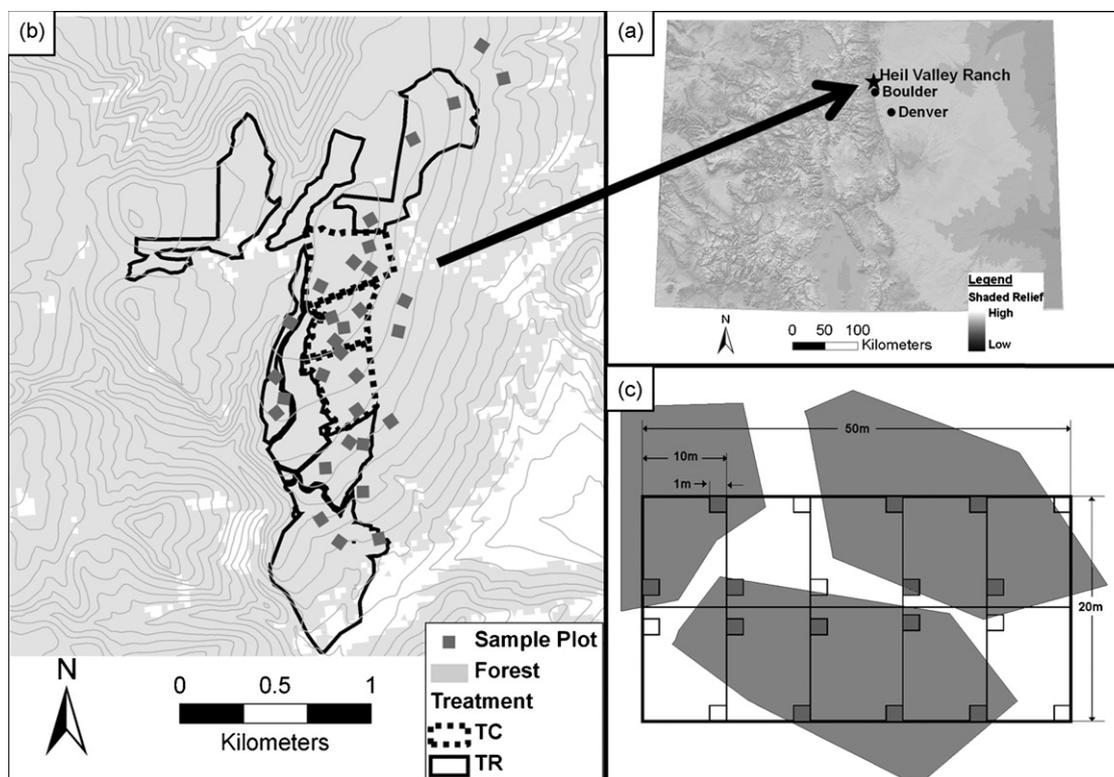
This study was conducted at the Heil Valley Ranch (HVR), which is owned and managed by Boulder County Parks and Open Space. The forested, 1992 ha property is located approximately 5 km northwest of Boulder, Colorado (Fig. 1a). Climate is characterized by generally cool and dry winters with warm summers. Average minimum January temperature at the city of Boulder weather station (1672 m elevation) is  $-6.3^{\circ}\text{C}$ , with average highs of  $7.5^{\circ}\text{C}$ . July is the warmest month with average maximum temperatures of  $30.9^{\circ}\text{C}$  and lows of  $14.8^{\circ}\text{C}$ . Average annual precipitation is 486 mm and the wettest months are April and May. Precipitation is relatively well distributed throughout the growing season from early spring through late fall (WRCC, 2008), with summer and fall precipitation dominated by short and often intense rain events. Soft surface recreational trails and an improved dirt road that is restricted to infrequent motorized travel by Boulder County employees are located within the study site at the HVR.

### 2.2. Study design

Boulder County acquired the HVR property in 1994 and subsequently designated a contiguous block of 141 ha of forest for future thinning, with the dual purposes of creating a large fire break and restoring historic forest structure. To facilitate implementation of the thinning project by private contractors, Boulder County divided the 141 ha into ten smaller stands ranging in size from 7 ha to 30 ha. Stand boundaries were established with regard to recreational trails and existing road access. The thinning prescriptions for all thinning treatments stipulated removal of all ponderosa pine individuals under 15 cm diameter at breast height (DBH). The thinning treatments differed by stand only in the fate of the removed biomass, which was determined by contractor preference and capabilities.

All thinning was implemented via chainsaw felling and hand crew or all-terrain vehicle skidding. In treatments where biomass was chipped, felled trees were fed through a Morbark Hurricane 18 WCL brush chipper (Morbark, Inc., Winn, Michigan) or similar model brush chipper towed by a pickup truck or ATV, and broadcast throughout the stand to an average depth of 7.5 cm and no greater than 15 cm. In treatments where biomass was removed from the site, ATV or hand skidding was used to forward logs to designated landings where they were locally chipped or hauled away. Stands were treated between 2000 and 2004.

Stands selected for study were at least three years post-treatment at the time sampling was initiated in 2005. Stands were classified into three treatment categories: (1) unthinned stands (*control*); (2) thinned stands with harvested biomass removed from the site (*thin-only*); and (3) thinned stands with harvested biomass chipped and distributed on the site (*thin + chip*). Pre-thinning forest conditions and environmental characteristics were similar throughout all stands selected for study (Chad Julian, Boulder County Parks and Open Space, personal communication), characterized by even aged forests strongly dominated (>95%) by



**Fig. 1.** Location of the study area (a) at the Heil Valley Ranch, near Boulder, Colorado, USA, and (b) a detailed view of the 31 sample plot locations and thinning treatments at the study site. The sample plot design (c) is overlain on a hypothetical distribution of woodchips within a chipping (thin + chip) treatment. Patches covered by woodchips are shaded in grey. 1 m<sup>2</sup> subplots lacking woodchip cover (FREE subplots) are small open squares, while 1 m<sup>2</sup> subplots with woodchip cover (CHIP subplots) are dark grey squares.

ponderosa pine, with occasional individuals of *Pseudotsuga menziesii* (Douglas-fir) and *Juniperus scopulorum* (Rocky Mountain juniper). The understory flora was sparse overall, comprised mainly of herbaceous species with an occasional shrub dotting the landscape. The sampled stands have gentle to moderate slopes (median 9.5°, range 4–15°), southeasterly aspects (median 133°, range 98–177°), and elevations ranging from 1921 m to 2069 m (median 2039 m). Soils are of the Pinata-Rock Outcrop Complex, which is a well-drained soil with a very stony loamy fine sand texture (USDA, 2008b).

### 2.3. Data collection

A total of 31 sampling plots, each 1000 m<sup>2</sup> in area, were established throughout the study site, with twelve plots in three thin + chip stands, thirteen plots in four thin-only stands, and six plots in three control stands (Fig. 1b). Plots were distributed among stands and thinning treatments according to a stratified random sampling scheme for each year of the study. All 31 plots were sampled once, either during the summer of 2005 or 2006. An equal number of plots were sampled in each treatment across years, except thin-only treatments which had seven plots sampled in 2005 and six in 2006. Additionally, plots were sampled in alternate treatments each day to ensure even sampling in each treatment as conditions changed throughout the summer. All plots were located a minimum of 50 m from treatment boundaries to avoid any transitional plant communities between treatments.

Vegetation was sampled in 20 m × 50 m plots using a nested-intensity sampling scheme, similar to the one described in Peet et al. (1998), but with twenty 1 m<sup>2</sup> subplots located in each 1000 m<sup>2</sup> plot (Fig. 1c). Vascular plants were identified to the species level and ocular estimates of percent cover for each species were recorded in all 1 m<sup>2</sup> subplots. Additional species were

recorded as present in the entire 1000 m<sup>2</sup> plot. We define understory plants as all vascular plant species excluding trees; therefore, tree species were excluded from understory analyses. Plants that were not identified to the species level were included in total cover estimates but excluded from all other analyses (average 3.0% of all species were unidentified in each 1 m<sup>2</sup> plot). In a few cases, similar species were easily confused when sampling occurred outside peak phenological development. In each case, data for similar plant species were combined and analyzed under one species name (see Appendix). Species richness was calculated as the sum of all species occurring in each sample unit. All the species were classified as either native or non-native to Colorado, according to Weber and Wittmann (2001) and the PLANTS Database (USDA, 2008a). In case of a discrepancy, we defaulted to Weber and Wittmann (2001). Using a variety of reference sources (Weber and Wittmann, 2001; USDA, 2008; USDA, 2008a), plant species were also classified based on the ability to spread vegetatively (rhizomatous, non-rhizomatous). Although the terms rhizomatous, stoloniferous, and vegetative reproduction have unique meanings, we use them interchangeably in the context of this paper to distinguish groups of plants that have the ability to grow horizontally and produce new individuals, vs. caespitose plants that can only produce new individuals through sexual reproduction. Nomenclature follows the PLANTS Database (USDA, 2008a).

Forest floor characteristics and environmental variables were also measured for each plot. In each 1 m<sup>2</sup> subplot, litter depth and duff depth were measured. Litter was defined as fresh and partially decomposed organic forest debris located above the mineral soil (O<sub>1</sub> and O<sub>e</sub> horizons), while duff consisted of highly decomposed organic matter below the litter layer and above mineral soil (O<sub>a</sub> horizon). It was not possible to distinguish woodchip depth from natural litter depth because chipped material was applied on top of

the existing forest floor and additional litter accumulation has occurred since treatments were completed. Therefore, when woodchips were present, litter and duff depth measurements included chipped material as well as natural litterfall, and the presence of woodchips was noted. Also, in each 1 m<sup>2</sup> subplot, ocular cover estimates were made for the cover of six forest floor substrates: exposed rock, mineral soil, litter and duff, living woody material (exposed roots, stems, and tree boles including fresh stumps), dead woody material (greater than 7.6 cm diameter), and woodchips. Environmental variables (slope, aspect, elevation), and digital photos were recorded for each 1000 m<sup>2</sup> plot.

Species, location and DBH were measured within the 1000 m<sup>2</sup> plot for all live and dead trees greater than 15 cm tall. Ponderosa pine seedlings less than 15 cm tall (including new germinants) were counted in each 1 m<sup>2</sup> subplot. Canopy closure was measured using a spherical densiometer at four locations within each 1000 m<sup>2</sup> plot (four measurements per location, 10 m between locations) and averaged.

We have observed that broadcast chipping of harvested trees produces uniformly shaped, small (~1 cm to ~5 cm on a side) pieces of wood typically deposited in clumps tens of meters in size that are heterogeneously spaced throughout the treatment. Thin + chip stands therefore include areas with deep woodchip cover and areas of similar size that lack chipped material. To investigate the specific effects of the uneven distribution of chipped material, every 1 m<sup>2</sup> subplot in each of the twelve 1000 m<sup>2</sup> thin + chip plots was classified as either *CHIP* (subplots covered in chips) or *FREE* (subplots free of chips). The vast majority of subplots were either completely covered or completely *FREE* of woodchips, but for those subplots that fell on chip-patch edges, we categorized it based on whether cover of natural needle litter or that of woodchips was higher. Understory plant species cover and richness, forest floor characteristics, and environmental variables were tabulated for *CHIP* and *FREE* areas by averaging the corresponding 1 m<sup>2</sup> subplots in each 1000 m<sup>2</sup> thin + chip treatment plot. This re-sampling of the data generated twelve data points that represent the differences between subplots with chips and those *FREE* of chips for each 1000 m<sup>2</sup> thin + chip treatment plot (*FREE*–*CHIP*) (Fig. 1c).

#### 2.4. Data analyses

Statistical tests were carried out using SAS v9.1 (SAS Institute, 2003) unless otherwise noted. Attributes were transformed when necessary to approximate assumptions of homoscedastic variance. Trees per hectare numbers were log transformed and a  $1/\sqrt{X}$  transformation was applied to all understory vegetation cover values. Significant differences for all tests were determined using  $\alpha = 0.05$  unless otherwise noted.

A mixed effects ANOVA including stand (i.e. treatment unit) as a random factor and treatment and year sampled as fixed factors revealed no effect of stand (variance parameter estimate  $\approx 0$ ,  $p > 0.2$ ) on forest characteristics, total understory species cover, or species richness. Therefore, the random effect of stand was eliminated from subsequent analyses. Forest structure, forest floor characteristics, and *P. ponderosa* germinants were compared between treatments using a one-way ANOVA. Two-way ANOVAs tested for the effects of treatment and year sampled on total understory species cover and richness, relative (percent of total) non-native species cover and richness, and relative rhizomatous species cover and richness. By analyzing the relative contributions of non-native and rhizomatous species, we are able to separate the effects of treatment on the understory as a whole (as measured by total understory cover and richness) from compositional changes to the understory brought by treatments (as measured by the

proportion of non-native and rhizomatous species). Linear contrasts, established a-priori, were used to compare the effects of each treatment. Tests for differences in these same variables between *CHIP* and *FREE* areas were conducted using paired *t*-tests.

We used quantile regression analysis (Koenker, 2008) to examine the relationship between understory cover and richness and litter depth in the thin + chip plots. Quantile regression allows us to estimate the effects of litter depth not only on average vegetation cover and species richness, but also on other quantiles of the distribution. We are especially interested in understanding the depths at which woodchips suppress understory vegetation cover or exclude species. We therefore report the quantile regression line associated with the 0.95 quantile, which approximates the near-maximum amount of vegetation cover and species richness that could be expected to be observed at a given depth of forest floor litter, realizing that low levels of other unmeasured factors (e.g. water, light, nutrients) may suppress the understory on some of our study plots, even when litter depths are permissive (Cade and Noon, 2003). For this analysis, we focused on the combined effects of thinning and biomass addition and, therefore, only data from the thin + chip 1 m<sup>2</sup> subplots were included. Plant cover values in 1 m<sup>2</sup> subplots were adjusted relative to maximum potential vegetation cover by dividing the percent of total plant cover by the percent of substrate that could support plant growth (not rock, living or dead woody material). Adjusted values were used only in quantile regression analyses.

We used multi-response permutation procedure (MRPP) to statistically test for differences in plant community composition between control, thin-only, and thin + chip treatments at the 1000 m<sup>2</sup> scale and between *CHIP* and *FREE* areas. Species occurring in less than three plots were deleted from each dataset, leaving 95 species at the 1000 m<sup>2</sup> scale and 44 species in the *CHIP* and *FREE* dataset. The procedure of deleting rare species reduces 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). MRPP produces both an effect size statistic (*A*), which measures within-group (e.g. treatment) homogeneity compared to random chance, and a significance value, measuring the probability that within-group similarity could be determined by chance (Mielke and Baerry, 2001; McCune and Grace, 2002). When the MRPP test was significant, we ordinated sample plots using nonmetric multidimensional scaling (NMS) to display relationships of overall plant community composition between treatments. NMS aligns plots on Cartesian axes based on the likeness of their species composition. Calculations were performed using Sorensen distance measure and a random starting configuration. A joint plot was overlain on the ordination to describe strong associations ( $r^2 \geq 0.2$ ) of forest characteristics and environmental variables with variation in plant community composition between plots. The final joint plot was rotated to maximize the variance between plots with regard to correlations with treatment differences.

Indicator species analysis was performed on untransformed average cover values for all species recorded in 1 m<sup>2</sup> subplots. A good indicator species has a high frequency within a group (i.e., the species is always found in samples belonging to that group) and is exclusive to that group (i.e., it has a high relative abundance within a group compared to all groups), with indicator values representing a combination of relative frequency and relative abundance scores (Dufrene and Legendre, 1997). 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. We considered species to be significantly associated with a treatment group when the indicator value surpassed 30 with a *p*-

**Table 1**  
Mean values (standard error) of forest structure characteristics

Forest structure	Control	Thin-only	Thin + chip
Overstory canopy cover (%)	49.5 (3.9) a	32.1 (2.6) b	28.8 (2.7) b
Density (tree/ha)	1687 (138) a	528 (94) b	520 (97) b
Basal area (m <sup>2</sup> /ha)	33 (2.6) a	21 (1.8) b	18 (1.8) b

Values within a row that are significantly different ( $\alpha < 0.05$ ) are marked with different letters.

value less than 0.1. The MRPP, NMS ordination, and indicator species analyses were completed with PCORD version 5.0 software (McCune and Mefford, 1999).

### 3. Results

#### 3.1. Treatment effects on forest structure and forest floor characteristics

Both thin + chip and thin-only treatments reduced canopy cover, trees per hectare, and basal area compared to control treatments (Table 1). Thin + chip treatments had deeper average litter depths compared to all other treatments (Table 2). Plots in the thin + chip treatments also had the lowest percent cover of rock and the highest average duff depths (Table 2). Within the thin + chip treatment, litter and duff depths were greater in CHIP areas than in FREE areas (Table 2).

#### 3.2. *Ponderosa pine* regeneration

Established *P. ponderosa* seedlings were not found in any plots, and *P. ponderosa* seedling germination did not occur in any plots during the summer of 2005. During the summer of 2006, one individual germinant was recorded in a plot within the control treatment ( $n = 5$  plots sampled), no germinants were present in the thin-only treatment ( $n = 5$ ), and five of the seven thin + chip plots surveyed that summer contained at least one germinant (total 75 individuals). There was no significant difference in the number of germinants per 1 m<sup>2</sup> subplot between FREE and CHIP ( $p = 0.585$ ). A subset of three plots in each treatment was revisited during the summer of 2007 and none of the germinants from the previous year had survived.

**Table 2**  
Mean values (standard error) of forest floor variables compared separately at the plot scale (control vs. thin-only vs. thin + chip) and at the subplot scale within chipped stands (CHIP vs. FREE)

Forest Floor	Control	Thin-only	Thin + chip	CHIP	FREE
Litter and/or woodchip depth (cm)	1.6 (0.2) a	1.5 (0.1) a	3.0 (0.2) b	3.6 (0.2) z	1.9 (0.2) y
Duff depth (cm)	0.9 (0.2)	0.8 (0.2)	1.2 (0.2)	1.5 (0.2) z	0.7 (0.1) y
Woodchip cover (%)	0.0 a	0.0 a	62.8 (2.7) b	87.5 (1.5) z	8.6 (2.1) y
Litter and/or duff cover (%)	85.1 (4.0) a	85.0 (2.8) a	26.6 (2.7) b	4.0 (0.8) z	75.8 (3.4) y
Rock cover (%)	11.5 (1.9) ab	11.4 (1.3) a	7.2 (1.4) b	6.2 (1.3)	8.9 (1.5)

No statistical comparisons were made between the two scales. Values within a row and treatment grouping that are significantly different ( $\alpha < 0.05$ ) are marked with different letters.

**Table 3**  
Mean values (standard error) of understory plant cover compared separately at the plot scale (control vs. thin-only vs. thin + chip) and at the subplot scale within chipped stands (CHIP vs. FREE)

Understory cover	Control	Thin-only	Thin + chip	CHIP	FREE
Total (%) <sup>a</sup>	3.9 (3.3–4.6) a	6.1 ab (5.3–7.0)	7.1 (6.0–8.4) b	5.1 (4.1–6.5) z	10.0 (8.4–12.2) y
Non-native (% of total)	3.0 (2.9) a	11.7 (2.0) b	4.6 (2.1) a	3.9 (1.1)	5.5 (2.3)
Rhizomatous (% of total)	45.2 (8.6)	49.9 (5.8)	45.7 (6.1)	51.0 (8.0)	43.4 (8.4)

No statistical comparisons were made between the two scales. Values within a row and treatment grouping that are significantly different ( $\alpha < 0.05$ ) are marked with different letters.

<sup>a</sup> Analysis on total (%) understory plant cover was performed with transformed data; back-transformed means ( $\pm$ standard error) are displayed, see methods for transformation details.

#### 3.3. Understory vegetation cover and richness

Total understory cover remained relatively low in all plots three to five years post-treatment (Table 3). Total cover values were significantly higher in 2005 relative to 2006 ( $p = 0.007$ ), with no significant year by treatment interaction ( $p = 0.438$ ). Averaged across years, both thinning treatments increased the total amount of vegetation in the understory compared to unthinned stands (Table 3). Thin + chip treatments had significantly higher total understory plant cover than did control treatments ( $p = 0.014$ ), while the difference between thin-only and control treatments approached significance ( $p = 0.052$ ).

Non-native species represented more relative cover in thin-only treatments compared with either control ( $p = 0.021$ ) or thin + chip ( $p = 0.021$ ), while species that can reproduce vegetatively had similar relative cover among treatments ( $p = 0.85$ , Table 3). Relative cover of non-native species was significantly greater in plots sampled in 2005 compared with 2006 ( $p = 0.001$ ). There was no difference in the relative cover of rhizomatous species between years sampled ( $p = 0.317$ ).

When the fine-scale patterning of chip distribution within the thin + chip treatment was taken into account by separating the subplots with and without chip cover, chip-free patches emerged as having roughly twice the total understory cover of patches with chips ( $p = 0.006$ ) and two and a half times the cover of untreated areas (Table 3). There were no differences in the percent of total cover between CHIP and FREE for non-native ( $p = 0.433$ ) or rhizomatous ( $p = 0.212$ ) understory plant species (Table 3).

A total of 129 understory plant species were identified across all treatments at the HVR, of which 81% were native and 19% were non-native. There were no differences between years or treatments in total species richness at the 1000 m<sup>2</sup> scale ( $p = 0.765$  for year,  $p = 0.139$  for treatment) or the 1 m<sup>2</sup> scale ( $p = 0.080$  for year,  $p = 0.331$  for treatment, Table 4). Control treatments had lower relative non-native species richness than either thin-only ( $p = 0.020$ ) or thin + chip ( $p = 0.017$ ) treatments at the 1000 m<sup>2</sup> scale, but there were not treatment differences at the 1 m<sup>2</sup> scale ( $p = 0.107$ , Table 4). Significantly greater relative non-native species richness was recorded in 2005 compared with 2006 at both spatial scales (1000 m<sup>2</sup>  $p < 0.001$ , 1 m<sup>2</sup>  $p = 0.002$ ). Forty-nine rhizomatous species and 80 non-rhizomatous species were

**Table 4**  
Mean values (standard error) of understory plant species richness compared separately at the plot scale (control vs. thin-only vs. thin + chip) and at the subplot scale within chipped stands (CHIP vs. FREE)

Understory species richness	Control	Thin-only	Thin + chip	CHIP	FREE
Total (# species/1000 m <sup>2</sup> )	53.0 (2.3)	47.4 (1.6)	48.0 (1.6)	–	–
Total (# species/1 m <sup>2</sup> )	4.3 (0.6)	4.9 (0.4)	4.1 (0.4)	3.6 (0.4) z	4.9 (0.5) y
Non-native (% of total) 1000 m <sup>2</sup>	14.0 (1.4) a	18.1 (0.9) b	18.2 (1.0) b	–	–
Non-native (% of total) 1 m <sup>2</sup>	8.9 (2.5)	14.3 (1.7)	9.5 (1.8)	9.2 (2.3)	13.1 (1.8)
Rhizomatous (% of total) 1000 m <sup>2</sup>	37.2 (1.5) a	36.3 (1.0) a	42.3 (1.1)b	–	–
Rhizomatous (% of total) 1 m <sup>2</sup>	50.1 (4.4)	44.2 (3.0)	45.9 (3.1)	47.9 (3.0)	40.1 (4.0)

No statistical comparisons are made between the two scales. Values within a row and treatment grouping that are significantly different ( $\alpha < 0.05$ ) are marked with different letters.

observed throughout the study site. Rhizomatous species represented a higher percentage of species present in the thin + chip treatment compared with control ( $p = 0.010$ ) or thin-only ( $p < 0.001$ ) treatments at the 1000 m<sup>2</sup> scale (Table 4), but there was no difference at the 1 m<sup>2</sup> scale between treatments ( $p = 0.465$ ).

Within the thin + chip treatment, CHIP areas had significantly lower total species richness than the FREE areas ( $p = 0.012$ , Table 4). There were no differences in the relative species richness between CHIP and FREE for non-native species ( $p = 0.185$ ) or species that reproduce vegetatively ( $p = 0.149$ , Table 4).

Increasing litter depth due to the addition of woodchips within the thin + chip subplots was associated with decreasing cover and species richness of understory vegetation (Fig. 2). Under median conditions (the 0.5 quantile), the quantile regression shows a significant decrease in understory cover with increasing litter depth (slope =  $-1.2$ ;  $p < 0.005$ ), until cover is completely suppressed at a depth of approximately 8.4 cm (Fig. 2a). However, the upper limit (0.95 quantile) of plant cover at each litter depth decreased more rapidly (slope =  $-4.6$ ;  $p < 0.0001$ ), with vegetation reduced to zero at about 11 cm (Fig. 2a). Even with very low litter depths, vegetation cover rarely exceeded 40%. Median species richness decreased with increasing litter depth (slope =  $-0.40$ ;  $p < 0.0001$ ). The linear 0.95 quantile line representing the upper limit to species richness under increasing litter depths decreased by about two species per three centimeters of litter depth ( $p < 0.0001$ ) until it approached a threshold litter depth near 8 cm, above which richness was reduced to near zero (Fig. 2b).

### 3.4. Understory plant communities

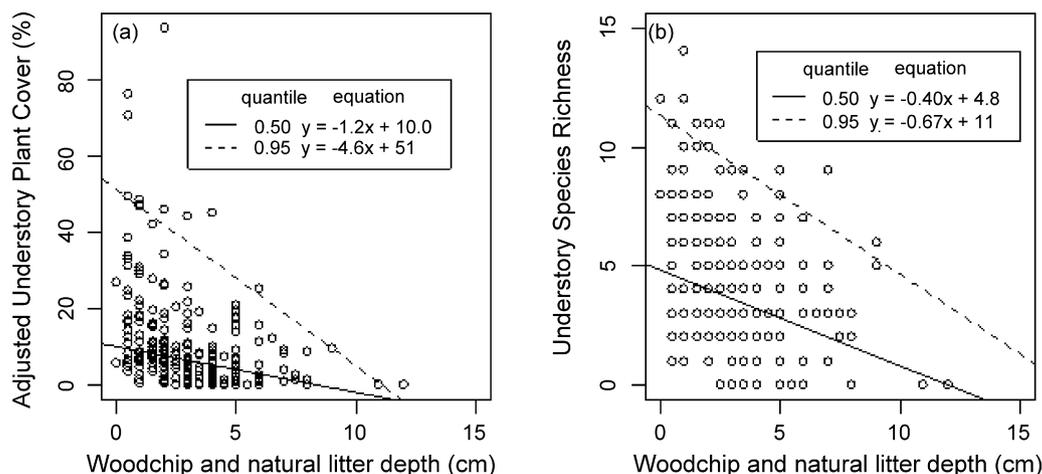
The understory plant community composition differed between treatments at the 1000 m<sup>2</sup> scale (MRPP test,  $A = 0.048$ ,

$p < 0.001$ ), but not between CHIP and FREE areas ( $A = 0.015$ ,  $p = 0.095$ ). Pairwise comparisons showed the thin + chip treatment differed from both control ( $A = 0.066$ ,  $p < 0.001$ ) and thin-only ( $A = 0.029$ ,  $p = 0.008$ ), but there was no difference in overall species composition between the thin-only and control treatments ( $A = 0.021$ ,  $p = 0.079$ ).

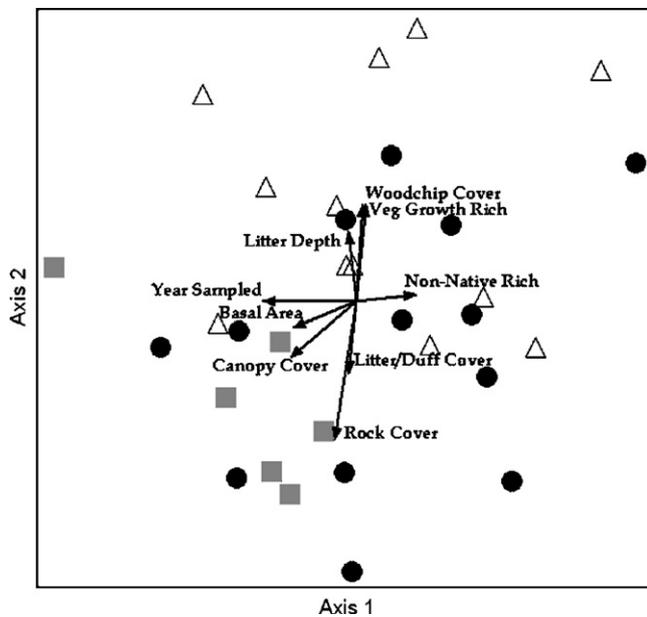
The NMS ordination used 85 iterations to produce a three-dimensional solution with a final stress of 15.2 and instability of 0.0050, which explained 80% of the variation in the original dataset. The joint plot of the two most important axes (24% and 34% respective variation explained) displays plots in the thin + chip treatment separated from control, with thin-only plots interspersed with both thin + chip and control treatments (Fig. 3).

Impacts of chipping (i.e. increased litter depth and cover of woodchips) were positively correlated with Axis 2 of the ordination (Fig. 3, Table 5) and inversely correlated with cover of rock and litter/duff (non-woodchip). Axis 2 therefore indicates a treatment effect from the addition of chipped material. Year sampled was strongly correlated to species composition of plots along Axis 1, in a direction orthogonal to the separation of the treatment groups. Rhizomatous species richness was positively correlated with woodchip cover, while non-native species richness was negatively related to plots in the control treatment (Fig. 3, Table 5).

Twelve species were significantly associated (indicator value  $> 30$  and  $\alpha < 0.1$ ) with one of the three treatments at the plot level (Table 6). These included five native indicators of the control treatment and four native indicators of thin + chip treatment, and two native indicators of the thin-only treatment. The only non-native species associated with a treatment was *Bromus tectorum* (cheatgrass), which indicated the thin-only treatment. Another particularly prevalent non-native species in



**Fig. 2.** Relationships between litter depth (chipped biomass + natural litter), and (a) total understory plant cover, and (b) species richness in the 1 m<sup>2</sup> thin + chip subplots. Total plant cover is relativized by the proportion of the subplot that could support vegetation (not rock, woody substrate, or coarse woody debris). Lines and equations represent quantile regressions between litter depth and understory cover at the median (0.5 quantile or 50th percentile) and the 0.95 quantile (95th percentile).



**Fig. 3.** Joint plot of a nonmetric multidimensional scaling ordination of species presence/absence values at the plot level. Scores for plots in the control (light grey square), thin + chip (black open triangle), and thin-only (dark grey circle) treatments are plotted on Axes 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 line vectors, where the vector length is proportional to the correlation strength. Abbreviations used for correlated variables are: Veg Growth Rich = relative (% of total) richness of rhizomatous understory plant species; Non-Native Rich = relative (% of total) richness of non-native understory plant species.

out plots, *Cirsium arvense* (Canada thistle), was most predominant in the thin + chip treatment. Canada thistle occurred in low abundance and therefore was not significant in our indicator species analysis that was conducted at the 1 m<sup>2</sup> scale. However, this species was present in 75% (nine of twelve) of thin + chip plots

**Table 5**

Coefficients of determination ( $r^2$ ) indicating percent of variation along each ordination axis explained by treatment, year sampled, forest structure, forest floor, and understory cover and richness attributes

		Axis 1	Axis 2	Axis 3
	Treatment	0.134	0.00	(-) 0.003
	Year sampled	(-) <b>0.322</b>	0.001	<b>0.389</b>
Forest structure	Canopy cover	(-) <b>0.224</b>	(-) 0.190	0.016
	Basal area	(-) <b>0.216</b>	(-) 0.088	(-) 0.010
	Tree density	(-) 0.153	(-) 0.079	0.007
Forest floor	Litter depth	(-) 0.027	0.240	0.012
	Duff depth	0.001	0.024	0.026
	Mineral soil cover (%)	(-) 0.010	(-) 0.109	(-) 0.004
	Rock cover (%)	(-) 0.077	(-) <b>0.477</b>	(-) 0.005
	Woodchip cover (%)	0.035	<b>0.335</b>	0.017
	Litter/duff cover (%)	(-) 0.025	(-) <b>0.247</b>	(-) 0.015
	Coarse woody debris (%)	(-) 0.005	(-) 0.001	0.00
	Woody plants (%)	0.029	0.018	0.041
Understory cover	Total (%)	0.041	0.122	(-) 0.043
	Non-native (% of total)	0.108	(-) 0.081	(-) <b>0.376</b>
	Rhizomatous (% of total)	0.057	(-) 0.054	0.061
Understory richness	Total 1000 m <sup>2</sup>	(-) 0.031	(-) 0.062	(-) 0.033
	Total 1 m <sup>2</sup>	0.057	(-) 0.053	(-) 0.028
	Non-Native (% of total) 1000 m <sup>2</sup>	<b>0.206</b>	0.022	(-) <b>0.391</b>
	Non-native (% of total) 1 m <sup>2</sup>	0.070	(-) 0.029	(-) <b>0.377</b>
	Rhizomatous (% of total) 1000 m <sup>2</sup>	0.024	<b>0.334</b>	0.137
	Rhizomatous (% of total) 1 m <sup>2</sup>	0.018	(-) 0.030	0.061

Variables with  $r^2 > 0.2$  are shown in bold, (-) indicates direction of correlation.

**Table 6**

Indicator species that are associated with treatments at the plot scale (1000 m<sup>2</sup>)

Treatment	Species	IV	p	Native	Vegetative spread
Control	<i>Ambrosia psilostachya</i>	33.0	0.043	Yes	Yes
	<i>Andropogon gerardii</i>	31.8	0.049	Yes	No
	<i>Elymus albicans</i>	38.1	0.052	Yes	Yes
	<i>Opuntia fragilis</i>	45.4	0.081	Yes	Yes
	<i>Schizachyrium scoparium</i>	37.5	0.076	Yes	No
Thin-only	<i>Bromus tectorum</i>	63.2	0.053	No	No
	<i>Harbouria trachypleura</i>	54.3	0.050	Yes	No
	<i>Phacelia heterophylla</i>	51.7	0.055	Yes	No
Thin + chip	<i>Agrostis scabra</i>	39.4	0.033	Yes	No
	<i>Danthonia spicata</i>	76.6	0.011	Yes	No
	<i>Luzula parviflora</i>	57.9	0.012	Yes	Yes
	<i>Scutellaria brittonii</i>	39.2	0.057	Yes	Yes

Species with  $p < 0.1$  and indicator value (IV)  $> 30$  are displayed. 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.

at the 1000 m<sup>2</sup> scale, 31% (four of thirteen) thin-only plots, and was absent from all six control plots.

## 4. Discussion and conclusions

### 4.1. *Ponderosa pine* regeneration

Our data suggest that successful conifer regeneration has not occurred since 2000 when thinning began at the HVR. *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). The higher germination rates observed in the chipped stands may suggest that the chips help to create a soil moisture environment favorable to germination in a year that was otherwise too dry. Because

germination did not lead to successful seedling establishment, we are reluctant to draw conclusions about the effects of thinning and chipping on *P. ponderosa* regeneration.

#### 4.2. Effects of chipped biomass depth and distribution on understory cover and richness

All treatments supported relatively low understory cover, suggesting that site characteristics other than chip depth, such as light, soils, litter accumulation, or moisture, are also likely to limit understory vegetation productivity. Chipping resulted in suppression of understory vegetation cover when litter depths were high. Organic litter has previously been shown to have a negative relationship with understory plant cover (Xiong and Nilsson, 1999), but such observations are confounded by the fact that 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 suppress understory vegetation (Covington and Moore, 1994; Naumburg and DeWald, 1999; Gildar et al., 2004). Because thinning with chipping increases litter depth in the presence of an open canopy, our observations support the notion that excessive litter layers *per se* can indeed suppress understory vegetation.

Contrary to our hypothesis and the observation that chip depth is negatively correlated with understory cover, the addition of chipped material to the forest floor in the thin + chip treatments did not suppress understory vegetation at the 1000 m<sup>2</sup> scale. Perhaps chip depths were low enough on average to permit understory vegetation recovery at similar abundances to thin-only treatments within three to five years following treatment. Alternately, the high understory cover values recorded in chip-free patches of the thin + chip treatment may have compensated in part for the suppression of understory vegetation in areas with heavy woodchip cover. Our experimental design unfortunately did not permit cross-scale statistical comparisons between the FREE subplots and the thin-only or control treatments, so we can only speculate that the enhanced understory cover observed in the chip-free areas at the subplot scale relative to all other treatments at the plot scale might be an important consequence of heterogeneous chip application. It is possible that some soil properties favorable for plant growth (e.g. increased soil moisture and moderated soil temperatures) that others have observed below layers of chipped biomass (Gower et al., 1992; Greenly and Rakow, 1995) might also be present beyond the edge of chipped areas (i.e. in FREE areas). The high understory cover in the FREE areas of the thin + chip treatment could then be interpreted as a consequence of favorable soil properties from the presence of chips but without the physical barrier of the chips themselves. Clearly there is a need to assess the effects of chipped biomass on soil properties and plant dispersal mechanisms at the operational scale of chipping treatments in order to confirm mechanisms for the increase in vegetation within chipping treatments.

While total species richness was not reduced in the thin + chip treatments relative to the thin-only treatments at the plot scale, CHIP areas had significantly fewer species per square meter compared with FREE areas. These observations suggest that species richness was not reduced uniformly throughout the thin + chip treatment, but rather that only a subset of species present in the FREE areas of the thin + chip stands could persist in the deeper areas of the CHIP subplots. Understory species richness declined with increasing depth of chipped biomass. We suspect that at low woodchip depths, most seeds can infiltrate the litter layer and germinate at the soil surface, then penetrate the chip layer and successfully establish. Where chipped biomass is deeper, the tightly packed physical structure of woodchips precludes soil-seed contact and successful germination for many species.

Differences in precipitation patterns are likely to explain the greater understory cover observed in 2005 relative to 2006. In 2005 there was moderate precipitation throughout the growing season. Winter and spring precipitation in 2006 was generally below average (November to June), although late summer (July) 2006 was especially wet (WRCC, 2008). Many non-native species at the HVR are winter annuals, such as annual *Bromus* sp., *Carduus nutans* (nodding plumeless thistle), and *Lactuca serriola* (prickly lettuce), which are dependent on winter and early spring moisture to establish. The dry winter and spring of 2006 likely reduced the ability of these non-natives to attain high cover, resulting in greater total non-native cover and richness in 2005.

#### 4.3. Effects of thinning and chipping on understory plant communities

Treatments that received no thinning or chipping manipulations had the lowest relative richness of non-native species (i.e. highest relative richness of native species) at the 1000 m<sup>2</sup> scale, indicating that some native species achieve competitive advantage in dense stands of ponderosa pine (Naumburg and DeWald, 1999), or that thinning reduces ecosystem resistance to invasion by non-native plants. Many of the native species that characterize control treatments generally occur with low abundance and frequency, explaining the low total cover and no more species observed per 1 m<sup>2</sup> subplot in control than on those of the other treatments. The species identified as indicators for control are all desirable, long-lived, native herbs that can persist under a dense canopy and increase in abundance locally within small patches of forest that provide suitable habitat.

While removing small-diameter *P. ponderosa* significantly changed the overstory forest structure compared to unthinned forests at the HVR, thinning by itself did not significantly alter the composition of the understory plant community compared to unthinned stands. However, thinning appears to provide the opportunity for non-native species to establish. At the stand scale, thin-only treatments supported a greater relative proportion of non-native species than did control treatments. Several distinct mechanisms may be contributing to this observed pattern. First, mechanical thinning equipment may have assisted the dispersal of non-native seeds into the treated stands. Second, the increased light to the understory may have helped to support populations of weedy species that were present as seeds but suppressed in the darker understory of the unthinned stands. Finally, disturbance to the forest floor and mineral soil exposure caused by thinning equipment and tree skidding might have helped to provide a more suitable seedbed for some non-native species. Although non-native species richness increased in thin-only treatments, it should be noted that native species still dominated the site, representing over 80% of relative understory species cover and richness in all treatments. The absence of a treatment effect on richness at the 1 m<sup>2</sup> scale indicates that many non-native species are found infrequently within all treatments; they are present at the 1000 m<sup>2</sup> plot scale but not frequent enough to contribute to richness of the average 1 m<sup>2</sup> subplot.

We observed that one species of management concern, *B. tectorum*, indicated thin-only treatments. We propose that thinning increased the occurrence of *B. tectorum* in all thinned stands (thin + chip and thin-only), but that the chipped biomass in the thin + chip stands reduced the abundance of this aggressive seed-dispersing understory plant species in that treatment. Wicks (1997) observed that *B. tectorum*, an aggressively colonizing seed plant, had reduced germination rates when seeds were sown in mulched agricultural fields or buried 2.5 cm in soil compared to seeds sown on the soil surface, lending further support to the idea that chips suppress *B. tectorum*.

The higher observed relative cover of non-native species in thin-only treatments does not extend to the thin + chip treatments, while the relative richness of non-native species does. The woodchip substrate may help to suppress populations of small seeded non-native plants such that, while they may occur in low abundance within thin + chip treatments, their ability to dominate the understory is kept in check by a shortage of suitable, chip-free germination sites.

Responses of understory plant communities to forest management practices are often driven by changes in the forest floor substrate (Gildar et al., 2004; Laughlin et al., 2004; Wienk et al., 2004; Wayman and North, 2007). Adding chipped biomass to the forest floor benefited some plant species more than others based on their reproductive strategy (vegetative vs. seed only), creating a unique understory plant community in the thin + chip treated areas. Chipping reduces the number of species that rely solely on seed reproduction by producing large areas that are unfavorable for establishment by seed. The small, uniform shape of woodchips and elevated levels of fungal decomposers found in chipped areas (Marchand et al., 2006) often creates a tight mat of litter. This compact physical structure of chipped biomass likely suppresses plant species which rely solely on seed reproduction by impeding soil-seed contact of new propagules, providing a substantial mechanical barrier to seedling emergence, and reducing light availability for new germinants (Facelli and Pickett, 1991). Xiong et al. (2001) found similar trends in riparian habitats, where a compact layer of silt favored species that reproduce vegetatively and loosely packed deep organic litter layers favored certain species based on seed size. Because most individual rhizomatous species are themselves patchily distributed, with each species relatively infrequent within a plot, the increase in richness of rhizomatous species we observed at the plot scale does not translate into a significant treatment effect at the 1 m<sup>2</sup> scale or an increase in the relative cover of rhizomatous species.

Most species capable of vegetative spread at our study site were desirable natives, such as the thin + chip indicators *Luzula parviflora* (smallflowered woodrush) and *Scutellaria brittonii* (Britton's skullcap). However, the non-native noxious weed *C. arvense* also seems to respond favorably to thinning and chipping treatments. Canada thistle was not detected as an indicator species, but data from the 1000 m<sup>2</sup> scale indicate that this species seems to do particularly well in the thin + chip treatment compared with thin-only and control treatments. Laubhan and Shaffer (2006) found that seeds of *C. arvense* had reduced germination rates similar to that of *B. tectorum* when buried 2–3 cm beneath the soil surface, but vigorous vegetative reproduction in *C. arvense* likely allowed this species to persist despite substantial depths of chipped biomass.

#### 4.4. Management implications

Our results suggest that understory plant cover and composition responds to both the depth and the distribution of chipped biomass. In large scale chipping operations similar to the ones described here, the common procedure is to drive the chipper through the forest to small piles or rows of previously harvested trees. This results in a fairly even distribution of chips in the areas near the piles. Local woodchip depths can be easily manipulated to meet management guidelines; this was accomplished at HVR using hand tools to spread woodchips below the maximum stipulated depth. In order to maximize desired effects on understory plants, future prescriptions could stipulate not only the maximum or average woodchip depth, but also the spatial distribution of woodchips. Manipulations of spatial pattern of chips could occur through more carefully guided yarding of harvested trees and/or

raking of chip piles. For example, stacking harvested material in windrows or closely spaced piles would produce a homogenous distribution of deep woodchip layers that would likely suppress the understory plant community, at least in the first several years following treatment. Alternatively, stacking harvested material in piles that are more widely spaced throughout the forest would produce uneven distributions of chipped biomass. Managers who would like to keep forest understory productivity low for wildfire prevention purposes may also find a deep, continuous chip layer to be beneficial, although the effects of the chips themselves on potential fire behavior and fire effects are poorly understood at this time (though see Glitzenstein et al., 2006). We found that chip depths of 11 cm were required to effectively block understory growth, though we recognize that the corresponding initial post-treatment chip depths may have been considerably higher than our depth measurements taken after three to five years of chip settling and decomposition. Results from related work indicate that chipped *P. ponderosa* biomass near the HVR settles at a rate of 18% after one year (Wolk and Rocca, unpublished data). Estimates of average and maximum woodchip depths reported here should be considered minimum depths when designing management plans, as original depths of chipped biomass at our site are unknown. We urge caution in extrapolating this exact depth into other forest types and locations, as local site conditions, species composition, and the size and shape of chips produced likely influence the ability of the understory community to grow through chipped biomass.

Guided, localized application of deep, homogenous woodchip layers could potentially prove helpful in reducing local infestations of short-lived, prolific seed-producing weeds, such as annual *Bromus* sp. Where whole tree chipping is implemented, careful consideration should also be made to monitor the occurrence of rhizomatous noxious weeds, such as *C. arvense*, which appear to benefit from chipping. Typical native understory vegetation in dry, low elevation Colorado Front Range *P. ponderosa* forests is dominated by perennial graminoids and low shrubs (Peet, 1981), while many problematic non-native species in our study site are short-lived, prolific seed producers. Although broadcast chipping reduced relative cover of non-native species at our study site, woodchip application could have detrimental effects on native species where short-lived aggressive seed colonizing plants are a dominant component of the native plant community or there is an abundance of vegetatively spreading non-native plants.

Efforts to increase thinning in ponderosa pine forests 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. Mechanical methods for *in situ* disposal of small-diameter trees can generally be classified into two categories based on the type of equipment used: (1) chipping, as described in this paper, and (2) mastication or chunking/shredding, which includes treatments completed using heavy machinery with a rotary blade, chain flail, chopper-roller, or similar mulching heads. These two methods are often lumped and called “chipping” or “mulching” 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 small, uniformly shaped pieces of wood typically deposited in piles or clumps, the location of which is determined by the forester. In contrast, mastication produces many size classes of woody debris (splinters to portions of intact tree boles) that are scattered throughout the treatment area at random (Stephens and Moghaddas, 2005). 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 even when woody debris depths are large. Additionally, mastication equipment shreds existing woody and herbaceous understory vegetation (Kobziar, 2007), while we observed that whole tree chipping buries the existing vegetation, which is otherwise largely undisturbed. We suspect that differences between chipping and mastication in the size and distribution of woody debris, soil mixing properties, and impacts to existing vegetation likely lead to different effects on understory plant community development and recovery following treatment. Future research should focus on describing ecological differences between forest management practices with careful consideration of the exact equipment and methods used.

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### Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at [doi:10.1016/j.foreco.2008.08.014](https://doi.org/10.1016/j.foreco.2008.08.014).

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