Joint Fire Science Final Report: Mastication effects on fuels, plants, and soils in four western U.S. ecosystems: Trends with time-since-treatment

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Abstract
Mechanical fuel reduction mulching treatments have been implemented on millions of hectares of western North American forests in recent years. Mulching transfers woody biomass to the soil surface, creating a treatment with no ecological analogue. This relatively-new management practice may have lasting effects on forest regeneration, understory plant communities, fuel loads, and ecosystems nitrogen dynamics and forest productivity. Since 2007 we have compared fuel reduction mulching and adjacent untreated stands at conifer ecosystems distributed across Colorado and evaluated the effects of mulch depth both experimentally and in operational units. This project (JFSP-10-1-01-10) builds on research we conducted the first 5 years after treatment establishment (JFS- 06-3-2-26) and will provide land managers and researchers greater understanding of the lasting effects of mulching on community and ecosystem processes.

Woody surface fuel loads remained higher in mulched areas than in untreated stands during the 6 to 9 years since treatment, but mulch up to 50% of its mass. Mulching increased soil moisture during parts of the growing season in ponderosa and lodgepole pine/mixed conifer but had mixed effects in pinyon-juniper forests. In general, fuel reduction and mulching did not reduce soil N availability though we found that application of thick mulch (15 cm deep) can reduce soil N shortly after mulching and in pinyon-juniper forests. Seedlings frequently become established in mulch at depths < 5 cm, but were found at depths up to 15 cm. Tree seedlings planted into mulch generally grew better and had higher foliar N content than those planted in unmulched plots. Fuel reduction mulching increased herbaceous plant cover and species diversity. Exotic and noxious plant species were more prevalent in treated stands but their cover was low.

The effects of fuel reduction treatments and mulching will be lasting due to the slow decay of woody mulch and rates of forest growth in dry conifer forests. As such, additional study of woody and herbaceous plant and soil response to these treatments are justified. Our collective findings and the broad climatic, vegetation and soils gradient spanned by our sites provides a useful platform for future examination of forest mulching.

Background and purpose
Mechanical treatments intended to reduce crown fire risk typically focus on removing ladder and canopy fuels to interrupt the surface-canopy fuel continuum (Agee and Skinner 2005). Because the removed material (e.g. small diameter trees, shrubs, and dead trees) is usually non-merchantable, these fuels are increasingly being disposed of by mechanically masticating or chipping (hereby referred to collectively as 'mulching') the undesirable biomass and leaving it on site. Scientists and managers are beginning to understand the initial (1-4 years post treatment) ecological responses to mulching (Glitzenstein *et al*. 2006; Collins *et al*. 2007; Wolk and Rocca 2009; Kane *et al*. 2010; Battaglia *et al.* 2010, Rhoades *et al.* 2012), but we do not yet know the longer-term ecological impacts of mulching treatments nor how different ecosystems may vary in their response. These uncertainties are especially relevant for mulching fuel treatments in the Colorado wildland urban interface, where the use of prescribed fire as a follow-up treatment is not feasible and decomposition will determine the rate of surface fuel load reduction.

Our research utilized a previous JFSP-supported study of mulching (JFS- 06-3-2-26) that established a network of replicated sites in pinyon pine, ponderosa pine, and mixed conifer/

lodgepole pine ecosystems. Our initial study was designed to measure fuel loads, understory vegetation, and soil resources in recently implemented (2 to 4 year old) mulching treatments. Results from the initial assessment indicated that basal areas were reduced substantially, approximately 60 to 70%. The mulching treatments increased surface fuel loads in all ecosystems 3 to 6 times of that found in untreated stands. Most of the material was in the 1-hr and 10-hr woody fuel (<2.54 cm in diameter) size class (Battaglia *et al.* 2010) which resulted in a range of depths within each ecosystem. We observed a significant increase in herbaceous cover in the pinyon pine and ponderosa pine ecosystems and cover tended to be higher in the mixed conifer/lodgepole ecosystems. No ecosystem showed differences in exotic plant cover between untreated and mulched areas, however, exotic species were observed more often in mulched areas. Tree regeneration was variable and it was unclear if the variability was due to lack of favorable microsites, seed production, or climatic conditions. Mulching also had few negative effects on ammonium (NH_4-N) or nitrate (NO_3-N) and in some cases increased these forms of plant available nitrogen across the sites. In contrast to the subtle effects of mulching applied operationally, deep mastication beds created for plot scale comparisons had larger effects on soil N. Plant available soil N was reduced under deep-mulched experimental plots the year mulching occurred, but the effect did not persist for a second year. In contrast, 5-year-old material released N regardless of amount of material added. Three to five years after treatment, available N was 32% higher in mulched areas.

The current study (**JFSP-10-1-01-10)** expanded the scope of our previous JFSP-supported study of mulching, by assessing **the effects of mastication (mulching) treatments on plants and soils over longer time frames in the absence of prescribed fire.** Our approach combines multi-year observational studies, which identified temporal patterns in plant and soil responses to mulching treatments, with carefully designed manipulation experiments, which helped elucidate the mechanisms responsible for the trends observed. With a clearer understanding of the mechanisms responsible for the variability in ecological treatment effects observed within and among ecosystems, we can do a better job of generalizing the results found in Colorado to other ecosystems. Our specific objectives were:

Objective 1: Identify trends in soil productivity, understory plant composition and plant cover 6 to 9 years following mulching treatments.

Objective 2: Assess the potential for non-native plant species to invade and persist in mulched areas.

Objective 3: Characterize the trajectories in key fuel load components with time-since-treatment.

Objective 4: Describe microsite characteristics that favor and hinder tree seedling establishment relative to mulch depth.

Objective 5: Test the influence of mulch layer depth and nitrogen limitation on tree seedling germination and growth.

Study description and location

Study sites and design

Eighteen sites were originally established across four ecosystems of the southern Rocky Mountains and the Colorado Plateau: lodgepole pine (*Pinus contorta*), mixed conifer (*Pinus ponderosa, Pseduotsuga menziesii*, *Pinus flexilis,* and *Pinus contorta),* ponderosa pine (*Pinus ponderosa/Pseduotsuga*), and pinyon pine/juniper (*Pinus edulis/Juniperus sp.*). These sites were distributed across a wide geographic range throughout Colorado and represent treatments across several federal, state, and other land agencies. The sites were mulched between 2004 and 2006, first measured in 2007 or 2008, and remeasured between 2011 and 2013 (Table 1). Of the 18 original sites, we removed 2 from the analysis due to a salvage harvest (lodgepole pine/mixed conifer site) and vandalism (pinyon-pine/juniper). We also removed several transects within two ponderosa pine sites due to a wildfire and tornado impacting the some of the study sites. We combined the lodgepole pine and mixed conifer ecosystems into one ecosystem category type because they represented a gradient of species compositions and site characteristics rather than two distinct ecosytems.

A total of seven sites were located in the lodgepole pine/mixed conifer (LP/MC) ecosystem, with one site on the western side and six sites on the eastern side of the continental divide. Tree species dominance was either pure lodgepole pine, or a mix of lodgepole pine, limber pine, Douglas-fir, and ponderosa pine (Table 1; Fig. 1). Elevations for the sites ranged from 2600 to 2900 m (Table 1). Annual precipitation ranges from 508 to 660 inches and falls as snow from September to May and rain in the summer months (WRCC, 2009). Average annual maximum and minimum temperatures are approximately 11 and -8°C, respectively. Presettlement fires at these sites range between mixed severity to stand replacing events (Arno, 2000).

Four sites were established in the ponderosa pine ecosystem (PP; Fig. 1). These sites were all located on the east side of the continental divide. Ponderosa pine was the dominant overstory species with various amounts of Douglas-fir also present (Table 1). Elevations ranged from 2100 to 2360 m. Annual precipitation ranges from 406 and 560 mm and falls as snow from September to May and rain in the summer months (WRCC, 2009). Average annual maximum and minimum temperatures are approximately 14 to 17°C and -2 to 2°C, respectively. Presettlement fires at these sites range between surface to mixed severity events (Brown *et al.*, 1999).

Five sites were established in the pinyon pine/juniper ecosystem (PJ; Fig. 1) and they were distributed throughout central and western Colorado. Elevations ranged from 1915 to 2400 m. Pinyon pine dominated two of the five sites and juniper species dominated the remainder. Annual precipitation ranges from 254 and 483 mm, with snow falling from October to May and monsoonal rain in the summer (WRCC, 2009). Average annual maximum and minimum temperatures ranged between 13 to 18°C and -6 to 2°C, respectively. Presettlement fires in pinyon pine/juniper are thought to be infrequent, stand replacing events (Floyd *et al.*, 2000; 2004; Huffman *et al.*, 2008).

For each mulched study site, we identified untreated reference areas. Untreated areas were located within 1 km of treated sites, in areas with similar aspect, elevation, soils, and forest type.

Pre-treatment surveys and post-treatment stump measurements from the original study were used to verify similarities between untreated and mulched areas.

Sampling Design and Field Measurements

Objectives 1, 2, and 3: Assessing ecological impacts of mulching treatments 6 to 9 years post-treatment

In the summer of 2007 and 2008, we established three 50-m permanent transects in each of the mulched and untreated areas of the study sites. Along each transect, we established 25 1-m^2 quadrats where we measured a host of variables, including understory plant cover and composition, mulch depth, and ground cover (litter, duff, mineral soil, rock, and woody fuels). In summer 2011 we returned to our permanent transects and remeasured all previously measured variables on the same quadrats.

Tree dbh, species, and status (live or dead) were measured along the 50-m transects. Transect width varied with treatment and tree size. Trees >10 cm diameter at breast height (dbh) were measured on a belt transect width of 20-m within the mulched areas and a width of 10-m within the untreated areas. Trees <10 cm dbh (saplings) were measured on a 10-m belt transect within the mulched and untreated areas. Saplings were enumerated by size (0 to 5 cm and 5 to 10 cm), status, and species. Tree seedlings (<dbh) were separated into two groups: seedlings <15 cm tall (post-mulching) and seedlings >15 cm and <137 cm tall (pre- and post-mulching).

Woody fuel loads (<7.62 cm) were estimated from the cover data described above using equations developed from our initial study (JFSP-06-3-2-26; Battaglia *et al.* 2010). The length, diameter at each end, and decomposition class of fuels >7.62 cm was measured on a 4 m x 50 m belt transect to calculate coarse wood loadings.

Soil nitrogen availability on an operational level (transects) was assayed using ion exchange resin (IER) bags inserted in the surface mineral soil (5 cm depth). Ten bags were installed along the permanent transects at 5 m intervals in summer 2011 and removed in summer 2012. At these same locations, we measured the volumetric moisture content of the upper 10 cm mineral soil layer distributed along mulched and untreated transects with a hand-held probe (CS 620; Campbell Scientific Inc., Logan, UT); sampling was conducted three times throughout the growing season per site.

In the summer of 2007 and 2008, we established three replicates of 2 x 2 m experimental plots at 17 study sites by removing or adding mulch to establish three distinct depths (0, 2.5, and 7.5 cm in pinyon pine and 0 cm, 7.5 cm, and 15 cm for the other ecosystems). In summer 2011, we installed four ion exchange resin (IER) bags in the surface mineral soil (5 cm depth) of each mulch bed plot. These were also removed in summer 2012. At these same locations, we measured the volumetric moisture content of the upper 10 cm mineral soil layer in each replicate with a hand-held probe (CS 620; Campbell Scientific Inc., Logan, UT); sampling was conducted two times throughout the growing season per site.

In the summer of 2007 and 2008, within each mulch layer depth experimental replicate, we installed four decomposition bags filled with 2 ages of ponderosa pine chips: fresh and 5 years old. One decomposition bag of each age was collected in spring 2011 (4 years) on the replicates that were used for the seedling growth study (see below; objective 5). The final decomposition bags were collected in Fall 2014 (7 years). Decomposition bags were placed in a paper bag, dried at 65°C until a constant dry weight was obtained, and weighed to the nearest 0.01 g. Mulch was ground and analyzed for carbon and nitrogen.

Objective 4: Describe microsite characteristics that favor and hinder tree seedling establishment relative to mulch cover and depth.

Along each permanent transect, we did a tree seedling inventory on a 50 x 1 m belt transect. For each tree seedling found, we measured ground cover and mulch depth in the surrounding 0.25 x 0.25 m area to examine the influence of environmental factors on seedling occurrence at the microsite (Bonnet *et al.* 2005; Coop and Schoettle 2009). A comparison of the ground cover immediately around seedlings with the ground cover measured in the established 25 1-m^2 quadrats (mentioned above) will reveal whether the seedlings are located preferentially in particular conditions or randomly dispersed within the plots. We sampled a minimum of 30 seedlings at each transect.

Objective 5: Testing the influence of mulch layer depth and nitrogen limitation on tree seedling germination and growth.

In Spring 2011, we planted 12 containerized 2 year old seedlings (species specific to the study site) on 2 of the 3 replicates (randomly selected) of the experimental mulch depth plots. Care was taken to limit the disturbance to the mulch layer and to avoid existing vegetation. Seedlings were planted on a 0.4 m x 0.4 m spacing. One of the 2 replicates with seedlings was randomly chosen to be amended with nitrogen fertilizer (10 g N/m^2 as urea) at the time of planting. Seedling survival, height growth, basal caliper diameter, biomass, and foliar nitrogen were measured in fall 2014. To complement our seedling bioassay of soil N fertility, we placed IER bags in the rooting environment (5-10 cm mineral soil depth) below the mulch.

In Fall 2011, we sowed ten seeds (species specific to the study site) on top of the mulch or mineral soil (simulating wind-dispersed seed deposition) at 9 gridpoints on the mulch depth experimental plot that was not fertilized. A frame of hard wire mesh covered the seeds to limit predation. Sites were visited during the fall of 2012 and 2014 to record germinant emergence. Unfortunately, a severe drought during fall 2011 and 2012 limited germinant emergence success and the experiment was abandoned.

Data analysis:

Objectives 1, 2, and 3

Fuels loads were first estimated based on transect-level means of substrate cover, mulch fuelbed depth, litter depth, and duff depth. The proportion that each fuel category contributed to the estimated total derived from the plot-based sampling was applied to determine transect-level fuel loads for each fuel size category (Battaglia *et al.* 2010).

We examined the effects of mulching treatments on fuel loads, substrate, and understory plant attributes using a mixed-model repeated-measures analysis of variance (ANOVA) in SAS 9.4 (GLIMMIX procedure; SAS Institute Inc., Cary, North Carolina, USA). Significance in this and all analyses was assessed with a $P < 0.05$, unless otherwise noted. Analyses modeled the dependent attribute against treatment, time since treatment, ecosystem, and all interactions. The appropriate distribution for each attribute was defined in the model; beta-distributed attributes (e.g., cover data) were rescaled per Smithson and Verkuilen (2006), if necessary, to accommodate 0 and 1 values in the dataset. The nesting of study areas within ecosystems was included in the model as a random effect. Time since treatment was included in the model as a random effect with a first-order autoregressive covariance structure. Post-hoc pairwise differences between mulched and untreated stands, by ecosystem × time since treatment, were then examined using least squares means with a Tukey-Kramer adjustment.

Using data collected at the microsite (1 m^2 quadrat) scale in the mulched plots, we fit 0.9 quantile regressions of the relationship between forest floor depth (as measured in the initial survey period) and herbaceous (i.e., graminoid plus forb) cover (as measured in the later survey period) for each ecosystem. We chose to use the initial forest floor measurement to more closely represent the mulch depths that were created by the mulching treatment. Analyses were run using the QUANTREG package (Koenker 2013) in R 3.3.1 (R Core Team 2014). A significant 0.9 quantile regression suggests an upper limit on herbaceous cover based on mulch depth or, in other words, that mulch depth becomes limiting when the other factors affecting herbaceous growth are permissive.

We tested for differences in understory plant composition between treatments with Permutational Multivariate Analysis of Variance (PERMANOVA; Anderson 2001) using the VEGAN package (Oksanen *et al.* 2013) in R. We ran separate analyses for each survey period and ecosystem, and included study area as a blocking variable. Species with fewer than four occurrences were dropped, and Bray-Curtis was used as the distance metric. To discover which species are responsible for any observed treatment differences in understory plant composition, we ran indicator species analysis (ISA; Dufrene and Legendre 1997). Again, we ran separate analyses for each survey period and ecosystem. ISA was run in PC-Ord 6.17 (McCune and Grace 2002).

Nitrogen and experimental mulch depth

We used mixed-model ANOVA to evaluate differences between mulch treatments and forest types (IBM SPSS Inc., Armonk, NY, V. 19). Differences between no mulch, shallow, and deep mulch beds were analyzed with mulch depth and forest type as fixed effects and study site as a random effect. We tested assumptions of normality and equal variance (Levene's test, Keyes and Levy, 1997) and log-transformed values as needed, to correct for unequal variances. Where significant depth effects occurred in the plot-scale experiment, Tukey's comparisons were used to identify differences among means. Graphical comparisons were made for soil volumetric moisture measured along transects and in mulch depth plots

KEY FINDINGS

Forest floor cover was still dominated by woody fuels in mulching treatment areas.
Mulching treatments dramatically altered the composition of the forest floor (i.e., the cover of litter and duff and 1 and 10 hr fuels; Fig. 2). Litter and duff were the most abundant substrate in untreated stands, with average cover values ranging from 46 to 82% across all ecosystems and sampling periods. Mulching decreased the cover of litter and duff, with average cover values of 36 to 64% observed in stands subjected to this treatment. This decrease in litter and duff cover was driven primarily by an increase in the cover 1 and 10 hr fuels, which rose from an average of 4 to 7% in untreated stands to an average of 14 to 49% in mulched stands. The cover of 100 hr fuels also increased following mulching treatments, averaging 1 to 3% in untreated stands and 2 to 7% in mulched stands. In contrast, the cover of 1000 hr fuels generally did not differ between mulched and untreated stands, and averaged <3% for all ecosystems, treatments, and sampling periods.

Despite the considerable addition of 1 and 10 hr fuels due to mulching treatments, untreated and mulched stands varied little in the depth of the forest floor (Fig. 3). Only LP/MC stands had a deeper forest floor in mulched than untreated stands, but this increase was restricted to the initial sampling period.

Woody surface fuel loads are still greater in masticated areas but have generally decreased over time.
Mulching increased woody surface fuel loads substantially initially (2 to 4 years) after treatment in each of the ecosystems measured (Table 2). A longer-term (6 to 9 years) examination of woody fuel loads indicates that in general, woody fine fuels are starting to either decrease (LP/MC and PJ) or remain the same (PP) over time, but still remain substantially greater than in the untreated stands. Although fine fuels did not decrease in the PP ecosystem, the variability of fine fuel loads did increase. This result, combined with the increase in coarse fuels in the PP ecosystem, suggests that there may have been some tree mortality in these stands which transitioned from snags to downed wood. The majority of the woody fuel in mulched areas is still in the fine fuel load category. In contrast, the majority of the woody fuel in untreated stands is coarse fuels.

Tree regeneration is not negatively impacted by mulching treatments.
In each of the ecosystems, new seedling regeneration (<15 cm) has been prolific and exceeds the densities observed in the untreated forests (Table 3). In LP/MC, new seedling regeneration is dominated by lodgepole pine and ponderosa pine in mulched areas compared to lodgepole pine and Douglas-fir in untreated areas. In PP, new seedling regeneration is dominated by ponderosa pine with some Douglas-fir, but in untreated areas it is an even mix of ponderosa pine and Douglas-fir. In the PJ ecosystem, the extremely high seedling densities are due to the sprouting of Gambel Oak in two of the research sites, especially in the mulched areas. However, there are sufficient densities of pinyon pine regenerating in these areas as well. Similar trends in PJ are observed for the seedlings that established pre/post mulching (15 cm to 137 cm). In the LP/MC ecosystem, most of these seedlings are a mix of lodgepole pine and Douglas-fir, however in the mulched areas it is a mix of lodgepole pine and aspen. Seedling composition in PP ecosystems is dominated by Douglas-fir in both untreated and mulched

areas. Sapling densities are still lower in the mulched areas of LP-MC and PP due to the targeting of that size during the initial treatments. In contrast, sapling density in the mulched areas of PJ has reached densities observed in the untreated areas. Again, this result is due to the fast growth and sprouting of Gambel Oak in two of the study areas.

Tree seedlings were able to establish in mulch depths up to 15 cm but preferentially established in depths less than 4 to 5 cm deep.
Lodgepole pine, Douglas-fir, and ponderosa pine were found growing in mulch depths up to 9, 15, and 13 cm deep, respectively, but preferentially established in areas with 0 to 3 cm deep, as observed by the increased frequency at that depth than was available across the site (Fig. 4). Pinyon pine was found growing in mulch depths up to 7 cm, but preferred depths less than 5 cm.

Mulch did not reduce seedling growth and enhanced it in some cases.
Mulch did not reduce seedling growth and enhanced it in some cases (Fig.5). After 3 years of growth, the aboveground biomass of lodgepole and ponderosa pine planted in shallow and deep mulch was more than twice as high as unmulched trees. Nitrogen (N) fertilizer had no effect on seedling biomass except when it was added to deeply mulched lodgepole and Douglas-fir trees. Foliar N concentration was also generally higher for mulched compared to unmulched seedlings. Fertilization increased foliar N in lodgepole seedlings regardless of mulching, but had no influence on the other 2 species. As a whole, mulch does not appear to hinder tree growth or create nutrient limitations.

Herbaceous understory plant cover increased in mulched treatments compared to the untreated stands.
Mulching treatments generally stimulated graminoid and forb cover at the stand (i.e., transect) scale (Fig. 6). Elevated cover values in mulched stands were found during our initial 2 – 4 year post-treatment surveys in PP and LP/MC ecosystems, and during our 6 – 9 year post-treatment surveys, they were found in all three ecosystems. The increased cover values in mulched stands 6 – 9 years post-treatment were marked, with graminoid and forb cover each being 2 – 6 times higher than their untreated counterparts.

Mulch depth can suppress herbaceous cover, but those depths exceeded the typical depths created by treatments.
These trends occurred despite our observation that a deep forest floor layer (i.e., the combined depth of litter, duff, 1 hr fuels, and 10 hr fuels) can indeed suppress herbaceous cover (i.e., the combined cover of graminoids and forbs; Fig. 7). At the microsite (i.e., 1 m^2 quadrat) scale, significant 0.9 quantile regression lines indicated that the upper limit of herbaceous cover depended on forest floor depth. Mulch effectively eliminated graminoid and forb cover when depths exceeded 14 and 20 cm for PJ and PP ecosystems, respectively. LP/MC did not have a significant 0.9 quantile regression line, yet the upper limit of forest floor depth for herbs to grow seemed to be about 15 cm.

Shrub cover response to mulch treatments was minimal.
In contrast to graminoid and forb cover, mulching treatments generally had no consistent stimulatory effect on shrub cover at the stand scale (Fig. 6). Shrub cover was only elevated in mulched LP/MC stands, and only in the last survey period.

Mulching treatments altered understory plant composition by enabling the recruitment of new species.
Our PERMANOVA results showed that mulching treatments influenced understory plant species composition in all ecosystems and time-since-treatment (Table 4). Results of our Indicator Species Analysis (ISA) suggest that these changes in composition were driven by the recruitment of new species more than by the loss of pre-treatment species (Table 5). Indeed, total species richness was higher in mulched stands in each year and ecosystem, except for LP/MC stands in 2008 (Fig. 8). While some of the newly recruited species in mulched stands were exotics, most were natives that prefer open canopy environments (Table 5).

Exotic plants were present in mulching treatments, but at low levels.
The effects of mulching treatments on exotic plant cover and richness varied by ecosystem (Fig. 9). Exotics were generally not stimulated in PJ stands, with exotic cover averaging 0.8% across all treatments and years, and exotic richness averaging 1.4 species transect^{-1}. In contrast, exotics were clearly stimulated by mulching treatments in PP and LP/MC stands. However, exotic richness and cover values in mulched PP and LP/MC stands were low both 2 – 4 and 6 – 9 years post-treatment; average exotic cover never exceeded 2% and average exotic richness never exceeded 3 species transect^{-1}.

Of greater interest in PP and LP/MC stands, perhaps, is that many of the exotic species we encountered 6 – 9 years following mulching treatments are on Colorado's noxious weed list (Table 6). We found only one noxious weed species in untreated PP stands 6 – 9 years post-treatment, but we found five species in mulched stands. In LP/MC stands, we found one noxious weed species in untreated stands and two in mulched stands 6 – 9 years post-treatment. Canada thistle (*Cirsium arvense*) was the most aggressive of the noxious weeds in both PP and LP/MC stands. In PP, Canada thistle was encountered in 0% of untreated yet 50% of mulched stands, and in LP/MC, it was encountered in 5% of untreated yet 71% of mulched stands.

Overall, the depth of material deposited by mulching treatments did not have a negative impact on soil nitrogen 7 to 8 years post-treatment. However, deep experimental mulch beds showed a negative impact on soil nitrogen in PJ sites, but not in PP or LP/MC forests.
Mechanical fuel reduction and mulching had no significant effect on ion-exchange resin soil N availability measured at the operational scale 7 to 8 years after stand treatment across all sites. Soil N availability was not altered by shallow or deep mulch in PP or LP/MC forests. In PJ sites, mulching caused a depth-dependent reduction in IER nitrate and total IER-N (Fig. 10). Shallow and deep mulch decreased IER-nitrate by 40 and 60% compared to PJ unmulched plots, respectively.

Deposition of mulch within the forest benefits soil moisture in PP and LP/MC, forests, but results varied for PJ.
Mulch increased soil moisture throughout the growing season beneath experimental mulch beds (Figs. 11 and 12) and along stand-scale transects (Fig 13) in LP/MC and PP, but not PJ forests (Fig 13). On-average, soil moisture was higher beneath shallow and deep mulch beds and mulched transects during May and June in the LP/MC and PP forests. Deeper mulch had little

additional effect on soil moisture in those forests. Mulch had varying effects on soil moisture in PJ forests. Episodes of lower soil moisture in treated vs untreated PJ stands and plots suggests that the mulch layer may intercept rainfall and prevent it from wetting the mineral soil.

Mulch continues to decompose, but the rate differs by mulch age and ecosystem
Over the 7-year study, the mass of standard pine wood mulch declined by 50 to 80% (Fig. 14); mulch mass declined least in PJ and most in PP forests. In PP and LP/MC forests, mass loss was greatest for fresh as opposed to aged mulch and mulch buried in deep as opposed to shallow mulch beds. For example, mass loss was 1.8 and 1.7 times faster in the deeper mulch beds in the montane and subalpine ecosystems, respectively. Neither substrate age or mulch depth affected mass loss in the PJ forests.

The N concentration of the residual mulch material increased over the course of the study. Similar to mass loss, changes in N concentration were greater for the fresh mulch and deeper mulch beds and less for mulch applied in PJ ecosystems. For example the N concentration of fresh chips increased 3-fold on average after 7 years in the field. These N changes were reflected by declining C:N from 229:1 at the time of application to roughly 70:1 after 7 years in the field. In contrast, the N concentration of the older mulch increased by only about 25% and the final C:N of the old mulch (63:1) differed little from that of the fresh mulch.

MANAGEMENT IMPLICATIONS

Fuels
Surface woody fuel loads in mulched areas still exceed those found in untreated areas. However, our results indicate that they are decreasing due to decomposition as indicated by our field measurements and the decomposition bag study. Litter cover is starting to increase on the surface indicating a fuel profile of soil covered with a mix of woody and litter from the initial treatment and litter from needle cast post-treatment. This additional change in fuel profile should be something fire managers that intend to burn in these treatments should consider. As these fuels age and decompose, we suspect there is some settling occurring which is likely changing the bulk density and proportion that each particle contributes to the total load (Battaglia *et al.* 2010). This could lead to erroneous fine fuel load estimates and should be acknowledged.

Trees
Overstory density in mulching treatments are still low 6 to 9 years post-treatment and, as intended, stands will continue to have low active crown fire hazard . Trees <10 cm dbh are present in the mid-canopy strata and will contribute to a future overstory over the next several decades. Tree regeneration that was sparse pre-treatment and/or that established immediately after treatment is now at sufficient densities for the future forest. The concern for deep mulch impacting new tree regeneration is unwarranted for the mulch depths that are deposited operationally. Each of the tree species studied were able to establish and grow in mulch depths that are commonly deposited in Colorado coniferous forests. As noted in our experimental mulch depth experiment, tree seedling growth is not negatively impacted by mulch. Instead, the seedlings seem to grow better, likely due to the increase soil moisture availability and moderated soil temperature.

It should be noted however that both new tree regeneration and advanced regeneration densities are extremely high and will need to be treated mechanically or with prescribed fire in the future. If the dense regeneration is not treated, the effectiveness of these treatments to reduce the transfer of a surface fire to the overstory canopy will be reduced. In addition, the prolific sprouting and growth of the Gambel Oak in two of the PJ sites suggests that PJ ecsystems (and other ecosystems) containing Gambel Oak will need to be treated more often and sooner than those of other ecosystems.

Understory Vegetation

Undesirable consequences of mechanical mulching treatments on understory plants appear to be minimal in the three Colorado forest ecosystems studied here. While mulch suppressed herbaceous cover at the microsite scale, at the stand scale, understory plant cover was enhanced. This is because the mulch was not evenly distributed across the forest floor; instead, the mulch was scattered in a mosaic, leaving some small patches with no mulch, and other patches with various depths of mulch. The stimulation of understory growth due to an open canopy and/or reduced underground competition far outweighed the suppressive effects of mulch.

For treatments designed to facilitate the reintroduction of a surface fire regime, a positive understory vegetation response might be considered desirable, as it provides fine fuels that will help to sustain a surface fire. In contrast, where surface fire prevention is a management objective, growth of the herb layer may be considered undesirable. The enhancement of several native species that favor open canopy environments might be an additional benefit of mulching treatments, especially where open canopies are underrepresented on the landscape.

Our results suggest that managers may not need to be overly concerned about promoting long-term exotic species invasions in mulched PJ stands. While exotic richness in mulched stands was elevated relative to untreated stands 2 – 4 years post-treatment, neither it nor exotic cover were elevated 6 – 9 years post-treatment. Furthermore, only one Colorado noxious weed, cheatgrass (*Bromus tectorum*), was documented 6 – 9 years post-treatment, occurring with low abundance in 42% of both untreated and mulched stands.

Although exotic species were promoted by mulching treatments in PP and LP/MC ecosystems, exotic richness and cover remained low 6 – 9 years post-treatment, and so we also do not consider them to be a major management concern at present. Only in LP/MC did exotic cover and richness show an increasing trend between the two survey periods; in these ecosystems, continued monitoring is recommended to evaluate if exotics will warrant a greater level of concern in future years. Managers should continue to monitor Colorado noxious weeds, such as Canada thistle, in mulched stands. It is possible that these disturbance-loving, aggressive species will continue to benefit from mulching treatments in PP and LP/MC ecosystems.

Nitrogen

The ecological consequences of fuel reduction mulching treatments are the combined result of forest thinning and mulch deposition. By reducing tree nutrient and water demand and by altering the soil microclimate, forest harvesting often dramatically increases soil nutrient pools, leaching and nutrient losses in stream water (Hornbeck *et al.*, 1986; Prescott, 2002; Aber *et al.*, 2002). These changes often stimulate rapid regrowth and resource demand by herbaceous and woody vegetation that then modulate ecosystem responses to disturbance (Vitousek *et al.*, 1979; Swank *et al.*, 2001). In work we conducted within 3 to 5 years of treatment, higher soil N availability in mulched areas was the net outcome of decreased forest nutrient and water use balanced by increased N demand of understory plants plus N immobilization within added mulch as evidenced by increased N concentration in added mulch. Our recent work has shown that the post-treatment changes in soil N availability have abated generally. In PJ sites, we found lower soil N under experimental mulch beds, though the amount of biomass treated in fuel reduction operations does not generate enough material to create deep mulch layers at a stand scale.

Soil moisture was generally higher in mulched stands and experimental mulch beds in LP-MC and PP forests, but was periodically lower mulched PJ areas. Tree seedlings colonizing mulched stands were abundant; those planted into mulch beds grew at least as well as and often better than unmulched trees. Mulching does not appear to present significant biogeochemical limitations to tree growth.

RELATIONSHIP TO OTHER RECENT FINDINGS AND ONGOING WORK

Fuels

Elevated fuel loads observed in our mulched study sites are similar to woody fuel loads in other western coniferous forest types (Kane *et al.* 2009, Brewer *et al.* 2013, Hood and Wu 2006, Reiner *et al*. 2009, Kobziar *et al.* 2009, Stephens and Moghaddas 2005, Kreye *et al*. 2014, Young *et al.* 2015). These other studies also observed shallow, compact fuel beds consisting of fine woody fuels mixed with litter and duff. However, the variability in fuel bed composition (i.e. proportion of 1-hr, 10-hr, litter, duff, and 100-hr) that contributed to the total fine fuel load differed among each of the studies, as well as in our study. This makes it difficult to develop fire behavior models that would predict flame lengths, rate of spread, intensity, and subsequent fire effects.

The lack of longer-term studies in western coniferous forests that examine mulched fuels over time limits our ability to compare temporal fuel dynamics. In our study, we saw a decrease in fine fuel loads across our sites and experimentally with our decomposition mulch bags in the experimental fuel beds. In pine flatwoods in Florida, where decomposition rates are much greater than those of the arid west, Kreye *et al.* (2014) did observe a decrease in 1-hr fuels within a year of treatment. The lack of similar repeated measurements of fuels on a long time frame in drier forest types makes it difficult to speculate about treatment longevity.

Tree regeneration

It is difficult to ascertain if our values of tree regeneration are similar or different to those of other studies of mulched treatment areas for several reasons. Regeneration often takes several years to establish and the majority of studies on vegetative response to mulching are initial measurements (1 to 4 years post-treatment). For example, Wolk and Rocca (2009) reported no ponderosa pine regeneration present 3-4 years after treatment. Walker *et al.* (2012) found that white fir, red fir, and incense-cedar regenerated in areas with a masticated layer, but Jeffery and sugar pine did not do as well. In addition, vegetation studies often focus on herbaceous and shrub response, rather than tree regeneration. Nevertheless, the tree regeneration densities observed in our mulched study areas exceed minimum requirements for forest stocking and can be viewed as a positive for future overstory recruitment. The negative is that some of this regeneration is too dense and will need treatment via thinning or prescribed fire to maintain treatment longevity.

Understory Vegetation

Few other studies have examined understory plant responses to mulching treatments in conifer ecosystems, and most of these have focused in initial impacts. Increases in understory cover and/or species richness in thinned and mulched sites, relative to unthinned controls and/or pre-treatment data, were observed in a Georgia longleaf pine savanna (Brockway *et al.*, 2009) and a Colorado ponderosa pine forest (Wolk and Rocca, 2009), but not in a northern California ponderosa pine forest (Kane *et al.*, 2010), an Oregon shrubland community (Perchemlides *et al.*, 2008), or a Sierra Nevada mixed conifer forest (Collins *et al.*, 2007). Our results support the observations of Wolk and Rocca (2009) that, in Colorado ponderosa pine, an overall increase in herbaceous vegetation cover results from treatment at a stand scale, despite suppression of understory herbs by deep mulch layers at a microsite scale.

Managers often ask us, "How deep is too deep to leave the mulch?" Our results provide guidance on the depths at which understory vegetation is suppressed. According to x-intercept from the 0.9 quantile equations, understory vegetation is almost fully suppressed at a mulch depth of 14 cm in pinyon-juniper and 20 cm in ponderosa pine. (We cannot provide similar estimates for LP/MC because we failed to detect a significant relationship between the upper limit of herbaceous cover and depth.) It should be noted that these depths were measured two to four years after treatment, so initial post-treatment depths were likely somewhat higher, before the mulch was compacted.

While we found little effect of mulching treatments on exotic plants in PJ ecosystems, others have found that such treatments can stimulate exotics, especially the noxious weed cheatgrass. For example, Owen *et al.* (2009) found that mulching increased cheatgrass cover from <1% in untreated PJ stands to 5% in mulched stands in a southwestern Colorado study area. Likewise, Ross *et al.* (2012) found that cheatgrass was absent from untreated PJ stands in southeastern Utah study area, but accounted for 20% of total understory plant cover in mulched stands. Also working in southeastern Utah, Redmond *et al.* (2014) found that mulching increased the relative density of exotic species, although this increase was driven primarily by prickly lettuce (*Lactuca serriola*), prickly Russian thistle (*Salsola tragus*), and tall tumblemustard (*Sisymbrium altissimum*), rather than by cheatgrass.

We are only aware of two studies that detail exotic plant response in mulched PP and LP/MC stands, and their findings are generally in agreement with ours. Working in PP stands of the Sierra Nevada, Kane *et al.* (2010) found that exotic species increased slightly yet non-significantly four years following mulching treatments. Collins *et al.* (2007) documented a significant increase in exotic richness and cover following one year-old mulching treatments in a California MC forest, but also pointed out that the magnitude of the increase was small.

Nitrogen

Woody mulch application has the potential to change soil nitrogen (N) availability. Similar to sawdust or sucrose additions, mulch may provide soil microbes a labile carbon (C) source that stimulates their growth and N demand and depresses inorganic soil N levels (McLendon and

Redente, 1992; Gower *et al.*, 1992; Blumenthal *et al.*, 2003). In fact, labile C amendments are commonly used in ecosystem restoration to immobilize nitrogen and reduce the competitive advantage of high nutrient-demanding invasive plants (Zink and Allen, 1998; Baer *et al.*, 2003; Perry *et al.*, 2010). The addition of wood mulch has been shown to depress soil N availability both in eastern deciduous (Homyak *et al.*, 2008) and western conifer forests (Miller and Seastedt, 2009). The depressive effects of C additions on soil N availability may be transient (Reever-Morghan and Seastedt, 1999; Perry *et al.*, 2010), but owing to the slow decay of wood residue the added material is likely to create long-term physical changes to the O horizon.

Further Work Needed

Fuels

The lack of information about the temporal dynamics of fuel loads and decomposition rates of mulched material hinders our ability to understand the longevity of mulching treatments. While this research study provides information for several conifer forest types of Colorado, the climatic conditions, soils, and subsequent ecological processes that impact decomposition may differ for similar forest types in other regions of the West. Although this study has examined these processes for 6 to 9 year post-treatment, decomposition is still very slow and will require revisiting these areas in the future to develop a better understanding of fuel dynamics. Further work is needed to understand when surface fuel loads will return to pretreatment levels. Furthermore, understanding how fuel bulk density in mulched fuel beds changes over time with the addition of needles from litterfall combined with the settling of the mulch material and the decomposition of mulch material is needed. Finally, since fire is the ultimate decomposer in the Western United States, a better understanding of how these treatments might burn initially after treatment as well as a decade or more after treatment is needed.

Tree regeneration

This study identified that tree regeneration was not hampered by mulch material. The establishment of new trees is important for a future forest; however, too much regeneration can reduce the longevity of the mulched treatment. Quantifying growth rates of the newly established regeneration as well as for the residual trees is an important consideration. Furthermore, the impact of a prescribed fire or wildfire on the new regeneration and overstory growing within these mulched treated areas which have high fuel loads is currently not well studied.

Understory Vegetation

The trend of higher herbaceous understory cover and species richness in mulched stands, by 6 – 9 years post-treatment if not earlier, held across all three ecosystems. We feel that this largely resolves the question of whether mulching treatments will enhance or suppress understory vegetation, and we expect that these results will extend to other coniferous forest types. The observed increases in cover and richness took longer to develop in LP-MC than in the other ecosystems, suggesting that forest types that start with a denser canopy and/or a sparser understory flora may respond more slowly to canopy opening than types that have suppressed understory plants waiting to expand.

While the effects of mulching itself may now be fairly well understood, future work should examine what happens when mulched stands burn in prescribed burns and/or wildfires. Conceivably, intense forest floor heating from burning of a mulch layer could kill underground plant organs or propagules, leading to a different understory flora than that which would be expected following a fire in an untreated forest.

Nitrogen

Uncertainties regarding the potential for smoldering fires to cause severe soil effects during dry soil conditions and for more gradual changes in nutrient cycling as herbaceous and woody species respond to soil and microclimate changes following fuel reduction thinning operations and mulch application warrant further investigation. Owing to the slow decay of woody mulch in dry and high western forests, additional study of plant and soil microbial responses to mulch addition is justified. The broad climatic and soils gradient represented by our sites provides a research platform to test hypotheses regarding the abiotic conditions and biomass inputs on soil saprophyte and ectomycorrhizal fungal communities. Such work will help advance understanding of the balance between decay and plant symbiont fungal groups that help regulate forest production and C losses and storage.

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DELIVERABLES

Deliverable Type	**Description**	**Status**
Publication	Tree regeneration dynamics in masticated fuel treatments	Internal review
Publication	Soil nutrient responses to masticated fuel treatments	Fall 2013
Publication	Effects of mulch depth and distribution on herbs post-mastication	Fall 2013
Publication	Non-native species richness and abundance following mastication	Fall 2013
Mgmt workshop, field tours	Interactive workshop and periodic field trips	2012 – 2013 (see list below)
Presentations	Data will be presented at major conferences	2011-2013 (see list below)
Invited paper/presentation	Presented at Rocky Mountain Research Station Silviculture Seminar Series (teleconference/webinar)	Spring 2012 (see list below)
BMP recommendations	Technical recommendations based on study findings	Fall 2013 (see list below)
Website	Website describing project with summary data.	Fall 2013; In Beta version: http://www.fs.fed.us/rmrs-beta/projects/mastication-effects-fuels-plants-and-soils-four-western-us-ecosystems-long-term-

Other deliverables:

Research from this project has been presented at various venues including local U.S. Forest Service Ranger Districts, University classrooms, FS Regional Office staff, Colorado State Forest Service staff and regional, national, and international audiences. To wrap up this project, we have organized a special session on mastication at the 6th International Fire Ecology and Management Congress in November 2015. At this special session, we will present our work along with 13 other scientists working in the mastication research realm.

Presentations:

Battaglia, M.A. Mastication as a fuels treatment: Short-term ecological implications of mulching the forest.

a) *Colorado State University, Fire Ecology seminar, March 2012*
b) *Colorado State University, Dept. Forest and Rangeland Stewardship Timber Management Class, November 2013*
c) *Colorado State Forest Service Annual Meeting, May 2014*
d) *10th North American Forest Ecology Workshop. Sustainable Landscapes: From Boreal to Tropical Ecosystems. Veracruz, Mexico 14-18 June, 2015.*

Battaglia M.A., KL Cueno, PJ Fornwalt, CC Rhoades, and ME Rocca. Tree seedling germination and establishment in masticated forest stands, Colorado.

a) *Interior West Fire Ecology Conference, Snowbird Resort, UT, 14-17 November 2011*
b) *Southwestern Fire Ecology Conference, Santa Fe, NM, 27 February – 1 March 2012*
c) *Front Range Fuel Treatment Implementer's Meeting, Denver, CO, March 2012.*
d) *Front Range Roundtable Quarterly Meeting, Denver, CO, 14 November 2014.*

Fornwalt, PJ., Rocca, M., Battaglia, M.A., Rhoades, C., Faist, A. Understory plant response to mulching treatments in forested ecosystems of Colorado.

a) *Society of Restoration Ecology, Madison, WI, October 2013.*
b) *Front Range Roundtable Quarterly Meeting, Denver, CO, 14 November 2014.*
c) *XXIV IUFRO World Congress 2014. Sustaining Forests, Sustaining People: The Role of Research, Oct 5-11, Salt Lake City, UT.*
d) *10th North American Forest Ecology Workshop. Sustainable Landscapes: From Boreal to Tropical Ecosystems. Veracruz, Mexico 14-18 June, 2015.*

Rhoades C.C., Battaglia M.A., Pierson D., Rocca M.E., and Fornwalt P.J. 2014. Effects of Fuel Reduction Mulch Treatments on Soil Nitrogen in Colorado Conifer Forests. *Front Range Roundtable Quarterly Meeting, Denver, CO, 14 November 2014.*

Poster presentations:

Battaglia M.A., KL Cueno, CC Rhoades, PJ Fornwalt, and ME Rocca. 2012. Mastication effects on fuels, plants, and soils in four western U.S. ecosystems: trends with time-since-treatment. *Southwestern Fire Ecology Conference, Santa Fe, NM, 27 February – 1 March 2012*

Battaglia, M.A., Rhoades, C.C., Rocca, M.E., Fornwalt, P.J, Cueno, K., Ryan, M.G.. 2012. Short-term ecological effects of mastication fuels reduction treatments in Colorado. *5th International Fire Ecology and Management Congress, Portland, OR. December 2012.*

Fornwalt, P.J., Rocca, M.E., Battaglia, M.A., Rhoades, C.C. 2013. Understory plant response to mulching treatments in forested ecosystems of Colorado. *98th Ecological Society of America Annual Meeting, Minneapolis, MN, August 2013.*

Battaglia, M.A., Rhoades, C.R., Rocca, M., Fornwalt, P.J. 2014. Mastication effects on fuels, plants, and soils in four western U.S. ecosystems: trends with time-since-treatment *XXIV IUFRO World Congress 2014., Sustaining Forests, Sustaining People: The Role of Research, Oct 5-11, Salt Lake City, UT.*

Field tours:

75th Anniversary of the Manitou Experimental Forest, Woodland Park, CO. Discussed mastication research on Manitou Experimental Forest. May 2014.

Battaglia, M. (Organizer and Speaker). Forest Operations and Management in Northern Utah, In-Congress Field Trip. 2014. *XXIV IUFRO World Congress 2014, Sustaining Forests, Sustaining People: The Role of Research, Oct 5-11, Salt Lake City, UT.*

Table 1: Site information for the 15 study sites.

Dominant Tree Species (>10 cm dbh)	Elevation (m)	Location	Site Name	Treat ment Year	Meas ured
Pinus contorta (100%)	2800	Arapaho and Roosevelt National Forest, CO (USFS)	Columbine	2005	2007
Pinus contorta (98%)	2818	Golden Gate Canyon Park, CO (CSP)	Golden Gate Canyon Park	2005	2007
Pinus contorta (100%)	2657	Granby, CO (private)	Snow Mountain Ranch	2003	2007
Pinus contorta (96%)	2600	Arapaho and Roosevelt National Forest, CO (USFS)	Winiger Ridge	2003	2007
Pinus flexilis (44%), *Pinus ponderosa* (38%)	2900	Cascade, CO (Private)	Catamount	2005	2008
Pinus contorta (58%), *Pinus ponderosa* (30%), *Pseudotsuga menziesii* (12%)	2760	Arapaho and Roosevelt National Forest, CO (USFS)	Sugarloaf 1	2006	2008
Pinus contorta (78%), *Pinus ponderosa* (9%), *Pseudotsuga menziesii* (12%)	2700	Arapaho and Roosevelt National Forest, CO (USFS)	Sugarloaf 2	2006	2008
Pinus ponderosa (58%), *Pseudotsuga menziesii* (42%)	2300	Pike National Forest, CO (USFS)	Buck	2004	2007
Pinus ponderosa (50%), *Pseudotsuga menziesii* (50%)	2100	Lory State Park, CO (CSP)	Lory State Park	2006	2008
Pinus ponderosa (94%), *Pseudotsuga menziesii* (6%)	2360	Pike National Forest, CO (USFS)	Manitou Experimental Forest	2005	2007
Pinus edulis (89%), *Juniperus sp.* (10%)	2400	Salida, CO (BLM)	Cherokee Heights	2006	2008
Pinus edulis (65%), *Juniperus sp.* (35%)	2200	Montrose, CO (BLM)	Dave Wood	2005	2007
Juniperus sp.(84%), *Pinus edulis* (16%)	1915	Cortez, CO (BLM)	Indian Camp	2004	2008
Juniperus sp.(61%), *Pinus edulis (*39%)	2250	San Juan National Forest, CO (USFS)	May Canyon	2005	2007
Juniperus sp. (78%), *Pinus edulis* (22%)	2170	Cortez, CO (BLM)	Summit	2005	2007

Table 2: Mean (and standard error) fuel loads for untreated and mulched areas of three coniferous ecosystems in Colorado in 2007/8 (2 to 4 years post-treatment) and 2012 (6 to 9 years post-treatment).

		Forest floor (Mg/ha)		Fine fuels (Mg/ha)		Coarse fuels (Mg/ha)	
Ecosystem	Treatment	2007/8	2012	2007/8	2012	2007/8	2012
LP/MC (n=7)	Untreated	26.54 (2.91)	30.38 (4.00)	4.18 (1.09)	4.02 (0.94)	3.5 (0.67)	5.3 (1.7)
	Mulched	34.04 (5.13)	26.04 (3.73)	50.05 (7.04)	38.34 (4.47)	5.2 (1.0)	7.7 (2.1)
		2007/8	2012	2007/8	2012	2007/8	2012
PP (n=4)	Untreated	19.02 (6.88)	29.94 (9.46)	3.93 (1.27)	2.81 (0.37)	8.3 (3.5)	5.7 (2.9)
	Mulched	21.94 (5.31)	27.95 (10.56)	30.85 (4.50)	33.56 (7.39)	5.3 (0.67)	10.1 (2.6)
		2007/8	2012	2007/8	2012	2007/8	2012
PJ (n=5)	Untreated	10.45 (2.37)	8.13 (1.61)	3.52 (0.76)	3.91 (0.68)	4.5 (2.7)	7.0 (4.0)
	Mulched	13.67 (2.80)	9.75 (1.70)	24.65 (3.67)	18.24 (2.78)	3.7 (1.4)	4.3 (1.8)

Table 3: Average tree density for post-mulched seedlings (<15 cm tall), pre-and post-mulched seedlings (>15 and <137 cm tall), and sapling sized (>dbh to 10 cm dbh) trees in lodgepole pine/mixed conifer, ponderosa pine/Douglas-fir, and pinyon pine/juniper study sites 6 to 8 years after mastication.

		Saplings (up to 10 cm dbh)	Seedlings (15 cm to 137 cm tall)	Seedlings (< 15 cm tall)
Ecosystem	Treatment	Trees per ha	Trees per ha	Trees per ha
LP/MC (n=7)	Untreated	540 (75)	238 (85)	533 (200)
	Mulched	43 (19)	2725 (1573)	2417 (814)
		Trees per ha	Trees per ha	Trees per ha
PP (n=4)	Untreated	1301 (534)	2887 (1905)	1040 (338)
	Mulched	88 (51)	877 (665)	5093 (3971)
		Trees per ha	Trees per ha	Trees per ha
PJ (n=5)	Untreated	623 (161)	5790 (3406)	3520 (1613)
	Mulched	633 (504)	17247 (10850)	2400 (1162)

Table 4. PERMANOVA results (p-values) testing for mulching effects on understory plant community composition in stands, by ecosystem and time-since-treatment.

	2 to 4 years	6 to 9 years
Pinyon pine – juniper	0.001	0.003
Ponderosa pine	0.012	0.001
Lodgepole pine / mixed conifer	0.006	0.001

Table 5. Indicator species of untreated and mulched stands, by ecosystem and time since treatment. "*" indicates the species is exotic to the continental United States and "**" indicates the species is an exotic classified as noxious by the state of Colorado.

(a) Pinyon pine – juniper

2 – 4 years		6 – 9 years	
Untreated	Mulched	Untreated	Mulched
	Prickly lettuce (*Lactuca serriola*)*		Yellow salsify (*Tragopogon dubius*)*

(b) Ponderosa pine

2 – 4 years		6 – 9 years	
Untreated	Mulched	Untreated	Mulched
Spearleaf stonecrop (*Sedum lanceolatum*)	Rough bentgrass (*Agrostis scabra*)	Common juniper (*Juniperus communis*)	Common yarrow (*Achillea millefolium*)
	Canadian horseweed (*Conyza canadensis*)		Canada thistle (*Cirsium arvense*)**
	Dandelion (*Taraxacum officinale*)*		Squirreltail (*Elymus elymoides*)

(c) Lodgepole pine / mixed conifer

2 – 4 years		6 – 9 years	
Untreated	Mulched	Untreated	Mulched
Common juniper (*Juniperus communis*)	Canada thistle (*Cirsium arvense*)**	Common juniper (*Juniperus communis*)	Canada thistle (*Cirsium arvense*)**
Greenflowered wintergreen (*Pyrola chlorantha*)	*American raspberry* (*Rubus ideaus*)	Greenflowered wintergreen (*Pyrola chlorantha*)	Front Range penstemon (*Penstemon virens*)
			Bigflower cinquefoil (*Potentilla fissa*)
			American raspberry (*Rubus ideaus*)
			Goldenrod spp. (*Solidago* spp.)
			Dandelion (*Taraxacum officinale*)*
			Thermopsis divaricarpa

Table 6. Exotic species documented during the two sampling periods, by ecosystem, and the percent of untreated and treated stands containing each. An * indicates the species is classified as noxious by the state of Colorado.

Species	**2 – 4 years post-treatment**		**6 – 9 years post-treatment**	
	Untreated	**Mulched**	**Untreated**	**Mulched**
Pinyon pine – juniper				
Alyssum (*Alyssum simplex*)	25	0	33	8
Field brome (*Bromus arvensis*)	0	17	0	0
Smooth brome (*Bromus inermis*)	0	0	0	8
Cheatgrass (*Bromus tectorum*)*	50	42	42	42
Musk thistle (*Carduus nutans*)*	0	25	0	0
Bur buttercup (*Ceratocephala testiculata*)	17	8	25	17
Prickly lettuce (*Lactuca serriola*)	0	50	8	0
European stickseed (*Lappula squarrosa*)	8	8	25	25
Tall tumblemustard (*Sisymbrium altissimum*)	0	17	0	0
Yellow salsify (*Tragopogon dubius*)	0	25	0	33
Ponderosa pine				
Smooth brome (*Bromus inermis*)	13	10	0	10
Musk thistle (*Carduus nutans*)*	0	20	0	20
Canada thistle (*Cirsium arvense*)*	0	20	0	50
Bull thistle (*Cirsium vulgare*)*	0	0	0	10
Prickly lettuce (*Lactuca serriola*)	0	20	0	0
Yellow toadflax (*Linaria vulgaris*)*	0	30	0	30
Kentucky bluegrass (*Poa pratensis*)	0	20	0	20
Dandelion (*Taraxacum officinale*)	13	70	0	30
Yellow salsify (*Tragopogon dubius*)	0	10	0	30
Mullein (*Verbascum thapsus*)*	0	30	13	10
Lodgepole pine / mixed conifer				
Smooth brome (*Bromus inermis*)	0	5	0	0
Canada thistle (*Cirsium arvense*)*	0	38	5	71
Bull thistle (*Cirsium vulgare*)*	0	5	0	5
Prickly lettuce (*Lactuca serriola*)	0	0	0	5
Marsh cudweed (*Gnaphalium uliginosum*)	5	0	0	0
Timothy (*Phleum pratense*)	0	0	0	5
Canada bluegrass (*Poa compressa*)	0	0	0	10
Kentucky bluegrass (*Poa pratensis*)	0	0	0	5
Dandelion (*Taraxacum officinale*)	5	24	10	48
Yellow salsify (*Tragopogon dubius*)	0	14	0	5
Mullein (*Verbascum thapsus*)*	0	10	0	0

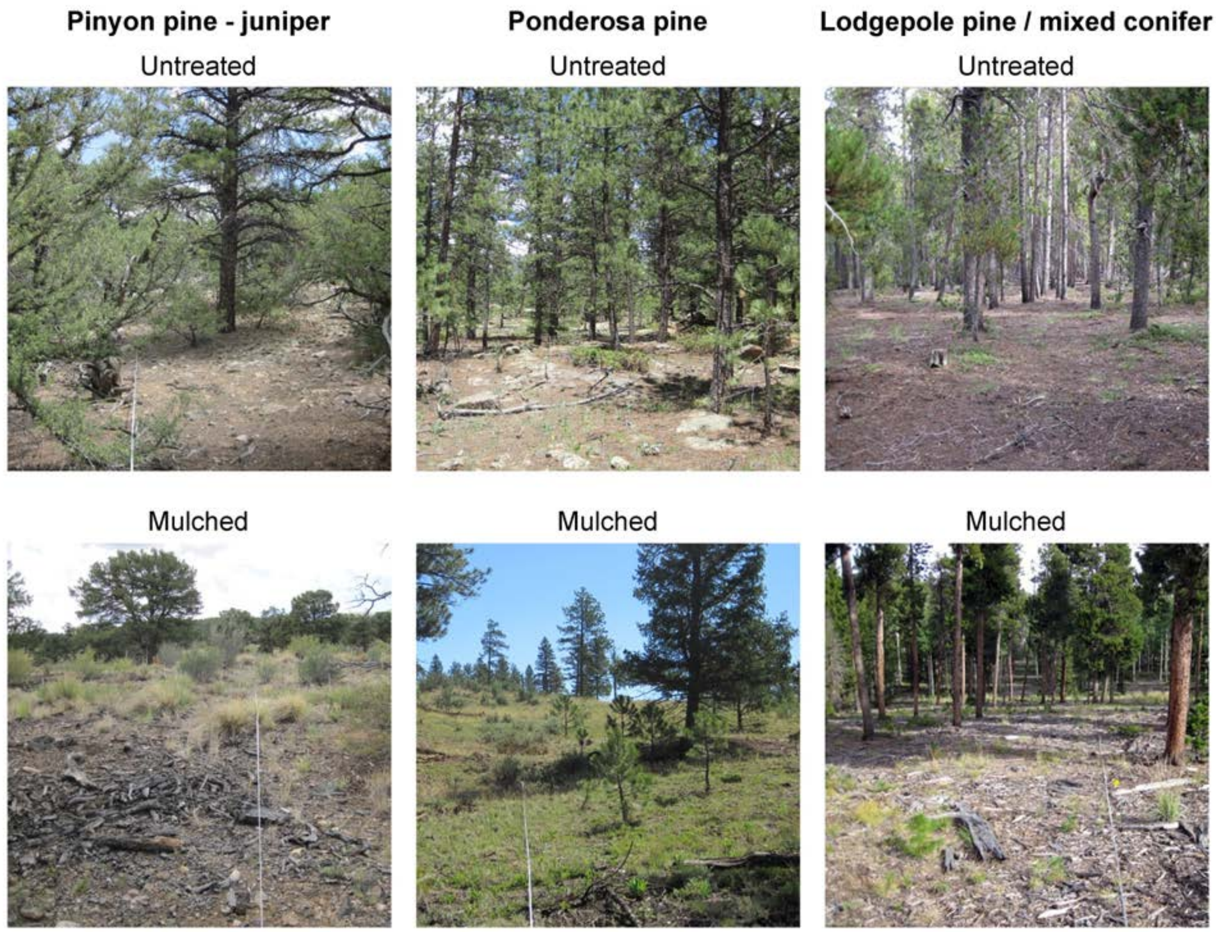


Figure 1: Representative conditions of untreated (top) and mulched (bottom) stands 6 – 9 years post-treatment, by ecosystem. The study areas depicted are CH (pinyon pine – juniper (PJ)), WS (ponderosa pine (PP)), and GGP (lodgepole pine – mixed conifer (LP/MC)).

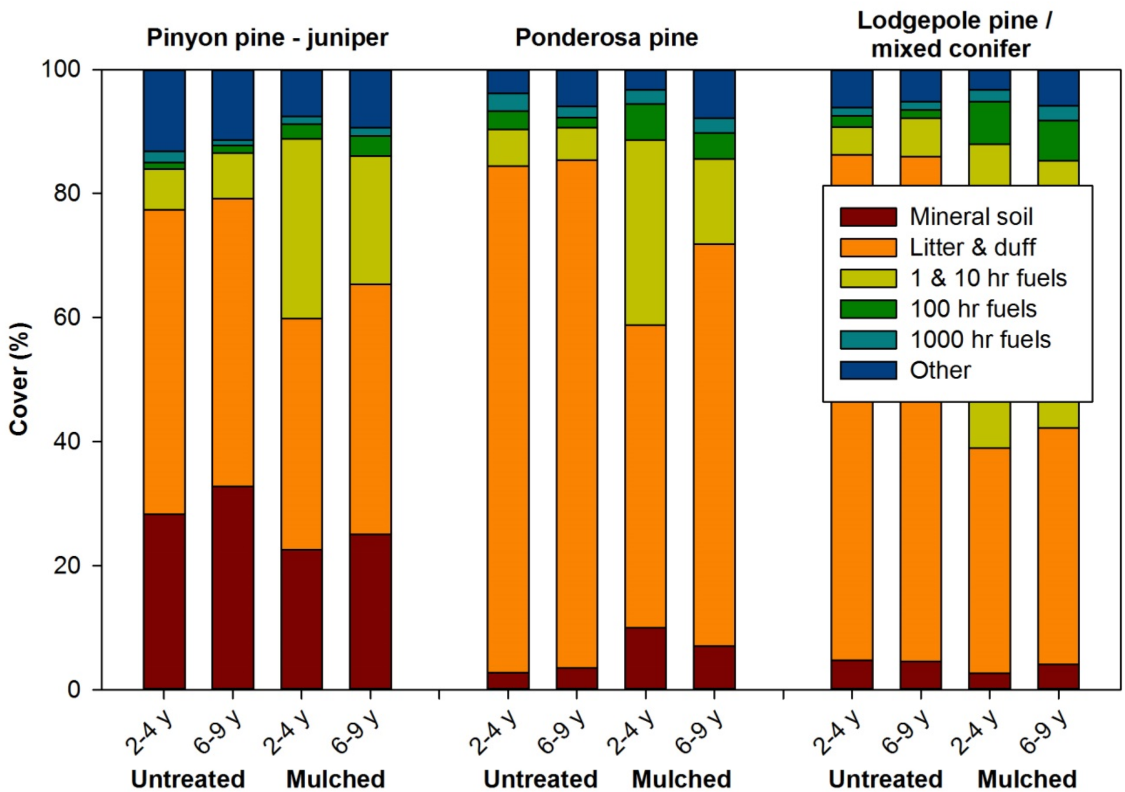


Figure 2. Stand scale substrate cover in each ecosystem, by treatment and time since treatment.

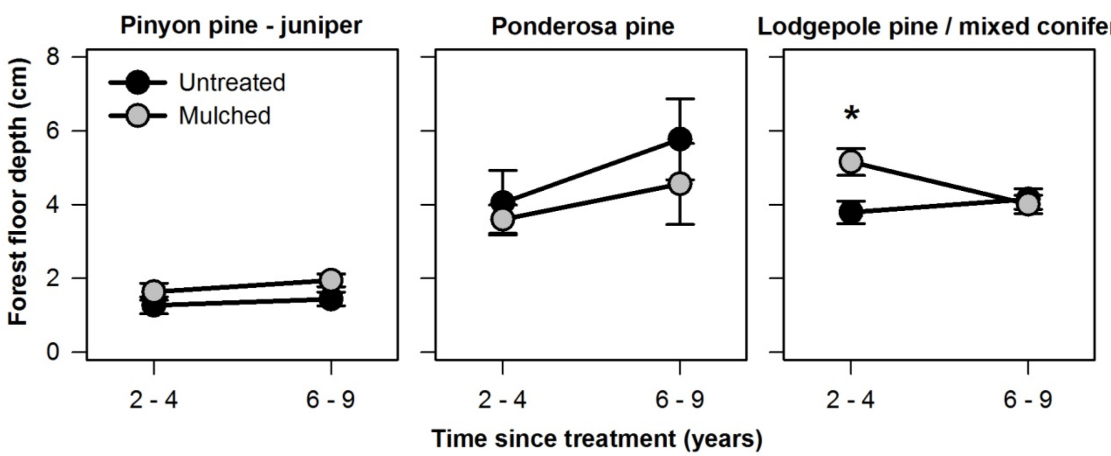


Figure 3. Stand scale forest floor depth in each ecosystem, by treatment and time since treatment. In untreated stands, forest floor depth is the cumulative depth of litter and duff. In mulched stands, it is the cumulative depth of litter, duff, 1 hr fuels, and 10 hr fuels; these woody fuels were often incorporated with the litter and duff during mulching operations. “*” indicates significant differences ($P < 0.05$) between treatments for that ecosystem and year.

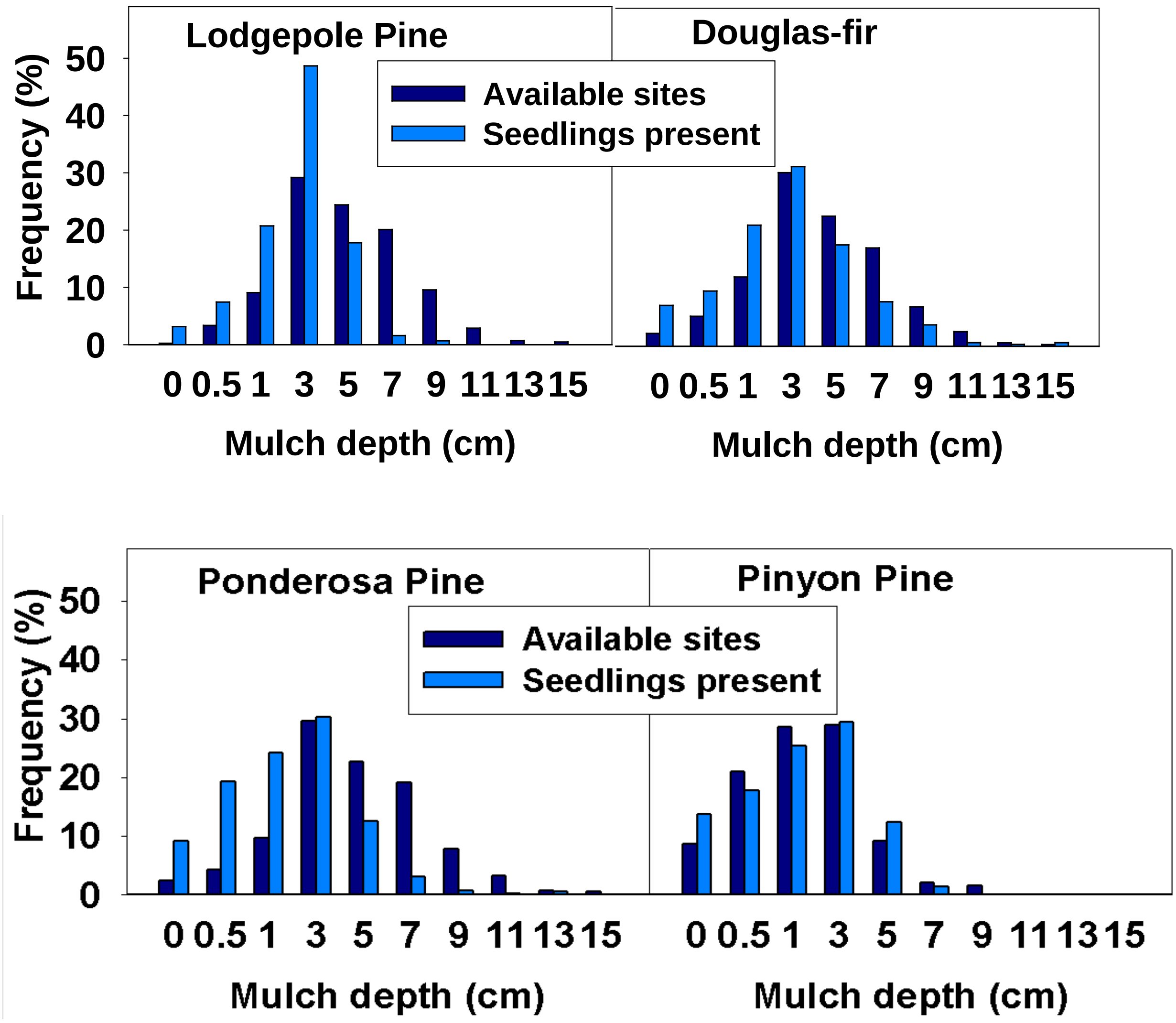


Figure 4: Frequency of seedling regeneration found at different mulch depths in comparison to the available mulch depths along a 50 x 1 m belt transtect about 7 years post-mulching.

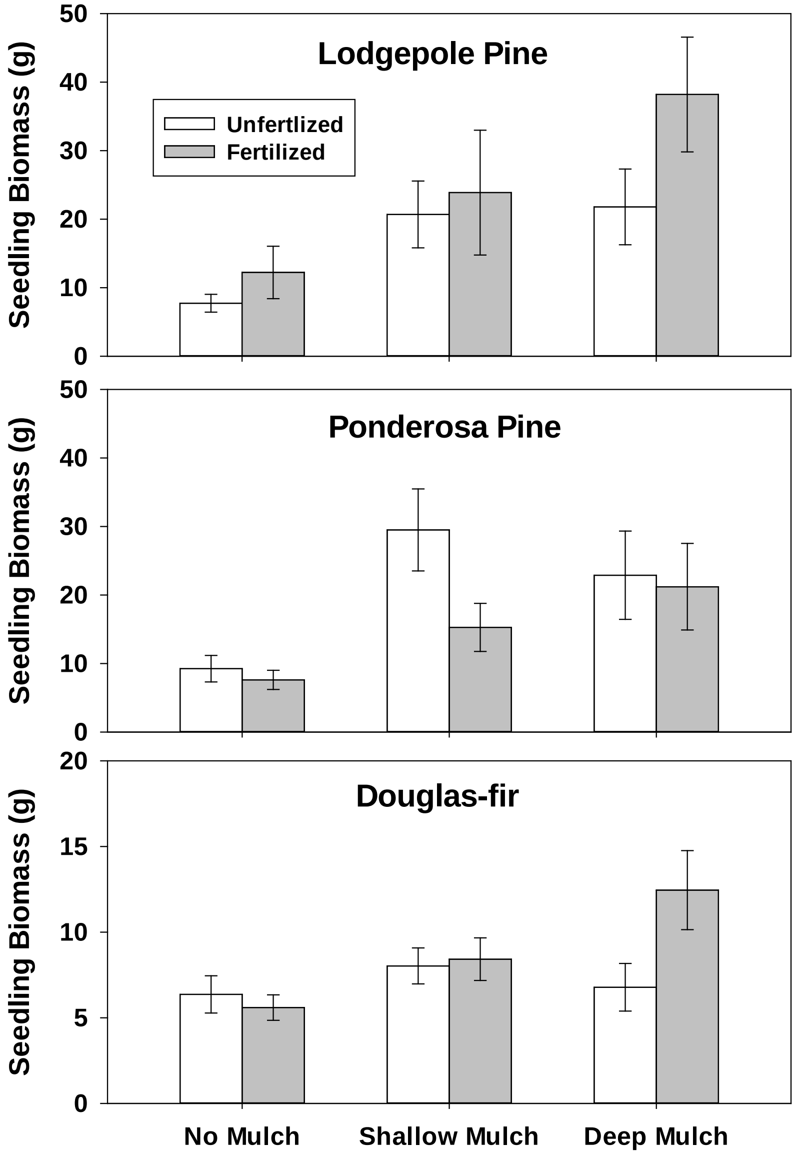


Figure 5. Aboveground biomass of lodgepole pine, ponderosa pine and Douglas-fir seedlings planted in mulched and umulched plots. Bars display mean and SE of final biomass after 3 years in the field.

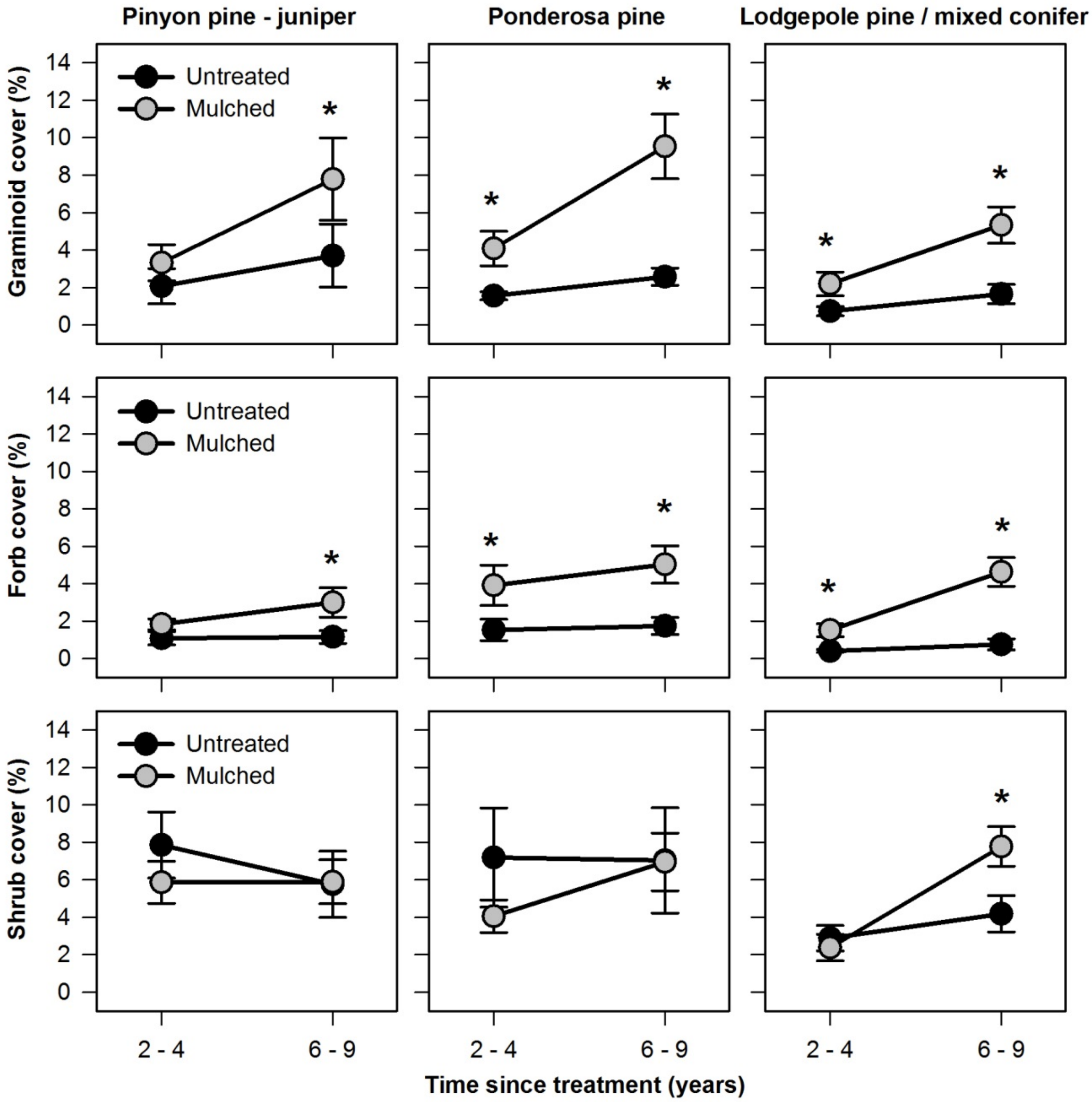


Figure 6. Stand scale graminoid, forb, and shrub cover in each ecosystem, by treatment and time since treatment. "*" indicates significant differences (P < 0.05) between treatments for that ecosystem and year.

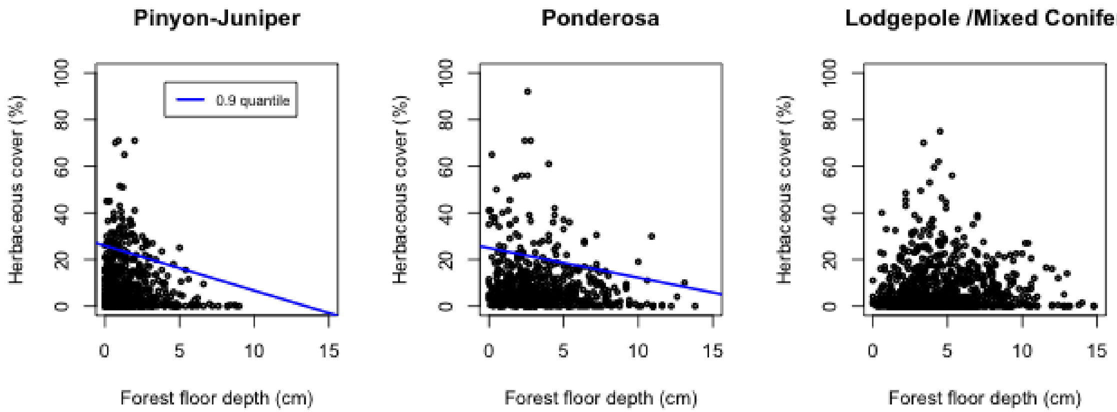


Figure 7. Quantile regressions between forest floor depth and herbaceous cover at the microsite (i.e., 1m^2 quadrat) scale, by ecosystem. Data are from mulched stands 6 – 9 years post-treatment. The regression was not significant (P ≥ 0.05) for lodgepole pine / mixed conifer.

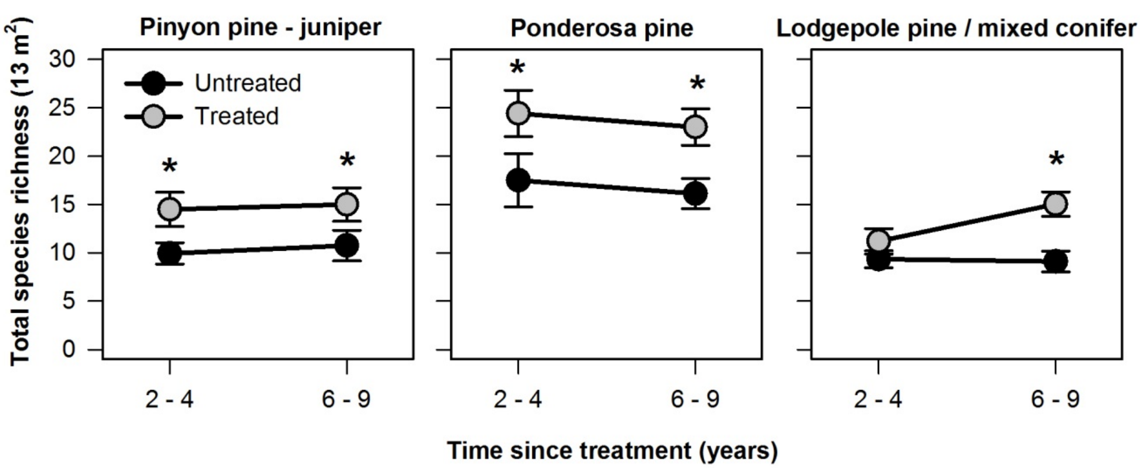


Figure 8. Stand scale total understory species richness in each ecosystem, by treatment and time since treatment. “*” indicates significant differences (P < 0.05) between treatments for that ecosystem and year.

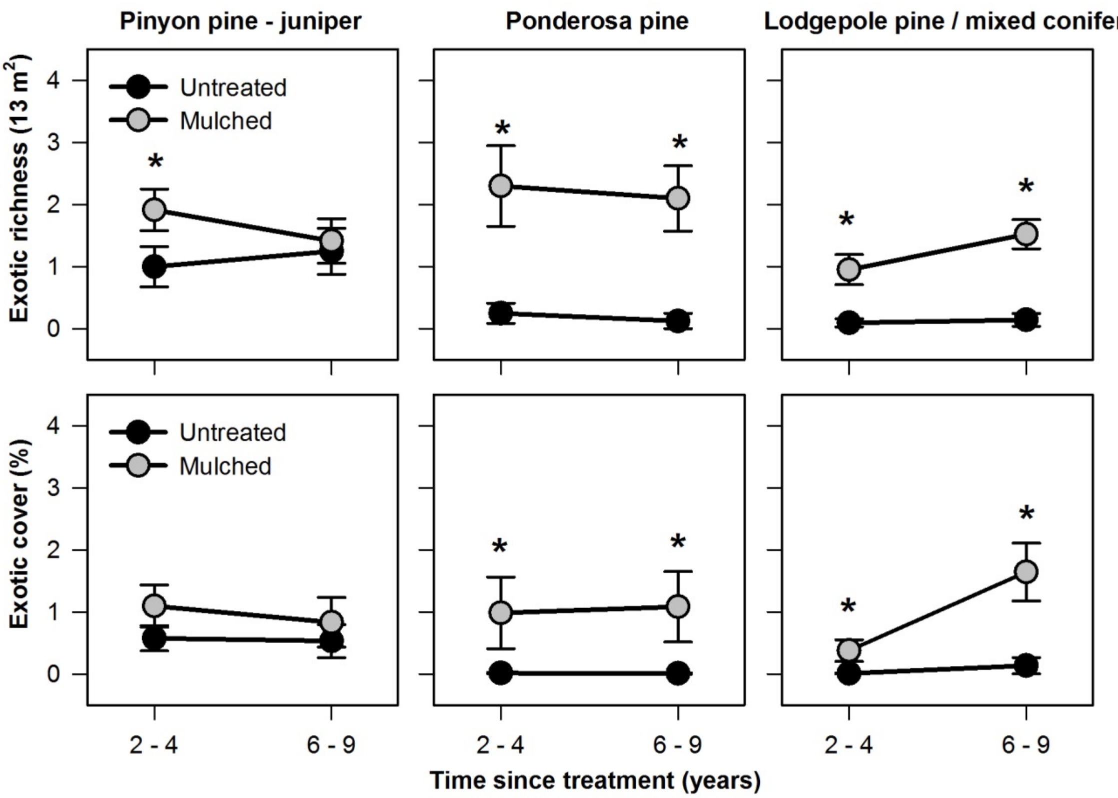


Figure 9. Stand scale exotic species cover and richness in each ecosystem, by treatment and time since treatment. Exotic species are those that are not native to the continental US (USDA NRCS 2015). “*” indicates significant differences ($P < 0.05$) between treatments for that ecosystem and year.

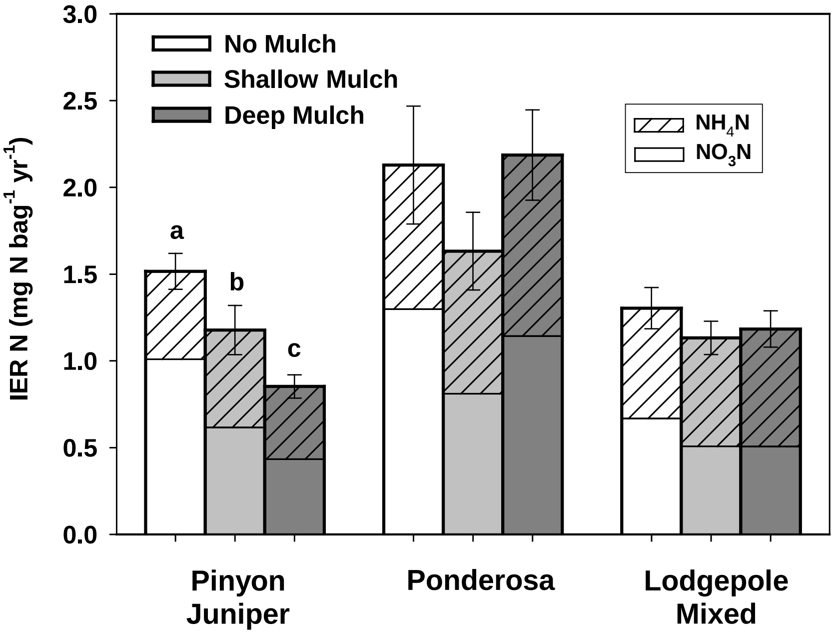


Figure. 10. Soil nitrogen availability measured by ion exchange resin (IER) bags experimental mulched and unmulched plots at 10 conifer forest sites (10 cm depth). The stacks display means of each IER N form. The SE bar and means test letters are for the sum of IER nitrate and ammonium.

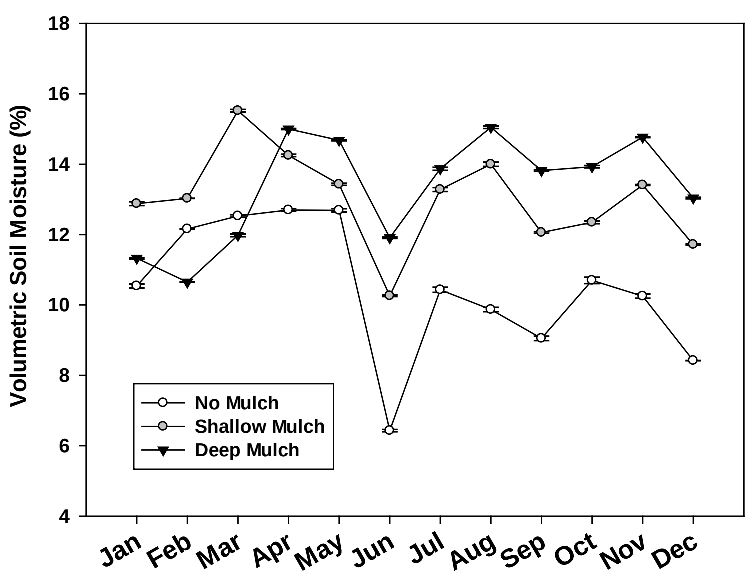


Figure 11. Volumetric soil moisture at a Colorado LP-MC forest site (10 cm depth). Symbols display monthly means and SE bars calculated from 10-minute interval measurements.

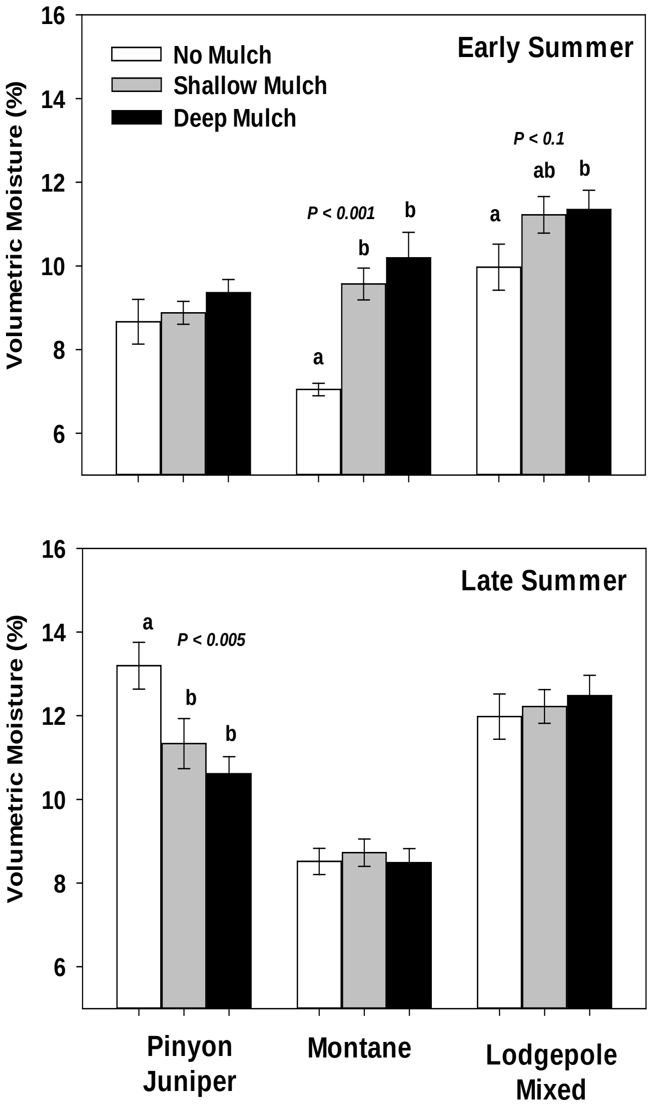
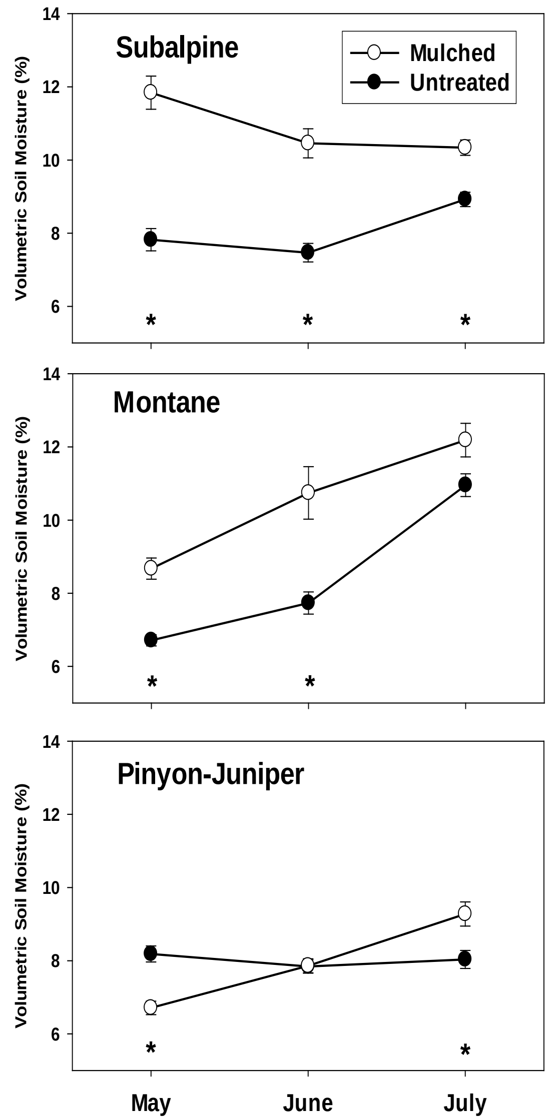


Figure 12. Volumetric soil moisture (0-10 cm depth) measured with a hand-held probe in experimental mulched and unmulched plots at 10 Colorado conifer forests during early (May-June) and late summer (Aug-Sept). Differing letters signify that means vary among mulch depths.

Figure 13. Volumetric soil moisture (0-10 cm depth) measured with a hand-held probe in fuel reduction mulch treatments and untreated stands at 15 Colorado conifer forests sites. Stars denote significant differences between monthly mean moisture content.



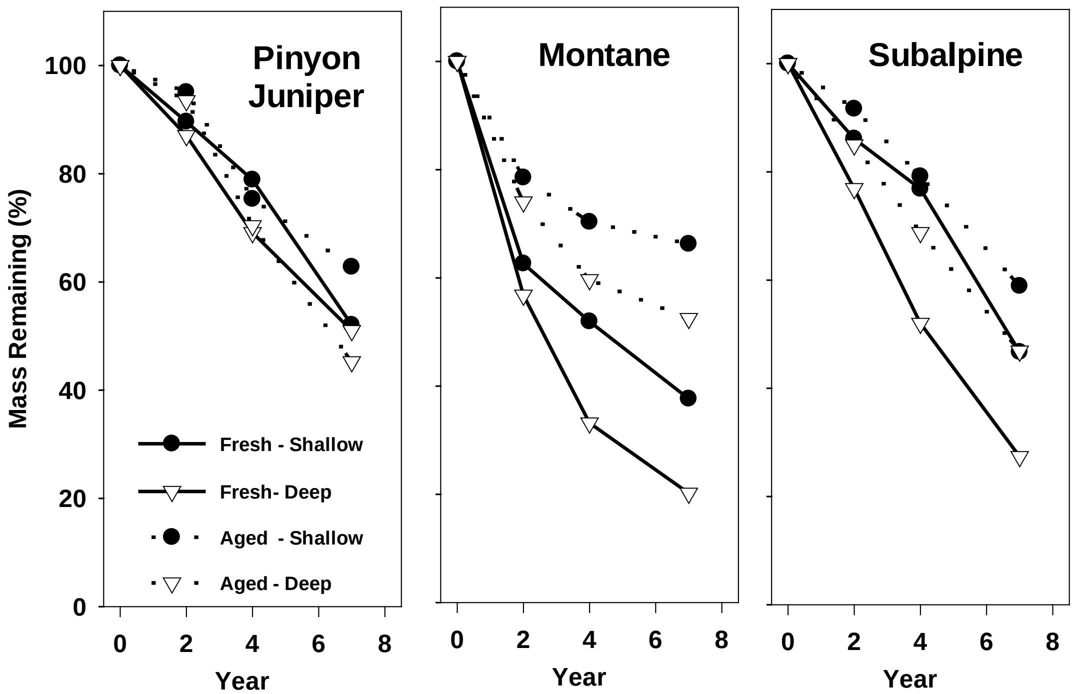


Fig. 14. Decline in the mass of ponderosa pine mulch over 7 years. Fresh and 5-year-old wood chips were compared in various mulch bed depths to gauge how mass and N changes vary with site conditions and initial substrate quality; fresh mulch N and C:N were 0.23% and 229:1 and old mulch N and CN were 0.6% and 93:1, respectively.