Project Title: A regional assessment of the ecological effects of chipping and mastication fuels reduction and forest restoration treatments.

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ABSTRACT:
Over the past several years, fire managers have increased their use of mastication treatments, the on-site disposal of shrubs and small-diameter trees through chipping and shredding. Mastication is a relatively untested management practice that alters the chemical and physical conditions of the forest floor and may influence vegetation regrowth and fuel development for years or decades. Eighteen sites were established across four ecosystems of the southern Rocky Mountains and the Colorado Plateau: lodgepole pine (*Pinus contorta*), mixed conifer (*Pinus ponderosa, Pseudotsuga menziesii*, *Pinus flexilis,* and *Pinus contorta),* ponderosa pine (*Pinus ponderosa*), 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 and first measured in 2007 or 2008.

The mechanical treatment added a substantial amount of 1-hr and 10-hr woody fuel (<2.54 cm in diameter) to the forest floor which resulted in a range of depths within each. The treatments provided a relatively large input of nitrogen (N) to the forest floor, but because of the high carbon (C) to N ratio of the added material (e.g. C:N of 125-175), the woody mulch is resistant to microbial decay and the added N is largely unavailable to plants. Slow mulch decomposition in arid and cold western forests may extend the consequences of this management treatment on plant germination, soil nutrient availability, and plant productivity for many years. We found that mastication had few short-term negative effects on plant communities and soil processes, but that responses to the treatment cannot be generalized across western conifer ecosystems. In some ecosystems, mulch additions had no significant impact on stand-level soil N availability, herbaceous cover, or tree seedling regeneration; in others, mastication decreased soil N availability and tree seedling regeneration and increased herbaceous cover. The depth of the added mulch also had consequences on plant cover and soil N availability. Specifically, above a thickness of 7.5 cm, mulching depressed herbaceous plant cover and soil N nutrition in lodgepole pine and pinyon-juniper ecosystems. Though the initial impacts of mastication were subtle, our findings indicate that responses will vary among ecosystems and justify further research to elucidate ecosystem-specific processes and long-term consequences of these treatments.

BACKGROUND AND PURPOSE
Many areas in the Rocky Mountain west are being thinned to reduce fire hazard and spread. Often the most economical solution for the disposal of the thinned trees is to chip or masticate them and leave the material on site. These treatments are assumed to reduce the ability of the forest to carry a crown fire, which is generally the primary objective. However, the effects of the added material (mostly wood and bark) on forest ecosystems are poorly known (Resh et al. 2008). Such treatments do not have natural analogues, because natural disturbances, such as fire, insect outbreaks, and blowdowns, leave woody material intact to decompose. Managers and the public are interested in understanding the impacts of the addition of this woody material on forest ecosystems so that they can evaluate the potential benefits and costs of these treatments.

Based on current ecological understanding, the addition of wood as chips or larger, masticated chunks can alter ecosystem function, but the issue has received very little study (Resh et al. 2008). A literature synthesis on the ecological effects of chipping and mastication in forested ecosystems uncovered many uncertainties and conflicting results preventing generalizations about treatment effects. First, most studies provided little information about the size of the added material and the distribution across the study site. Second, thinning itself can have a large effect on many ecosystem properties, because it reduces competition, leaf area, and interception of precipitation, and increases light to the forest floor (Oliver and Larson 1990). These changes in environment can cause changes in ecosystem function that act in the same or different direction as changes caused by the addition of wood chips or chunks. For example, reduced transpiration (from thinning) and reduced surface evaporation (from chip mulch) both increase soil moisture. However, decomposition of low nutrient wood can immobilize nutrients, while thinning can reduce plant uptake and increase nutrient availability. Third, most studies did not use common measurement protocols, nor did they replicate across sites. Furthermore, almost no information is available for the forest types found in the southern Rocky Mountains and the Colorado plateau.

The goal of our study was to understand the ecological effects of mulching (i.e. chipping and mastication) treatments in relation to the depth, density, and distribution of treated material in a broadly replicated study in four forest types for the southern Rocky Mountains and the Colorado Plateau: pinyon-juniper, ponderosa pine, mixed conifer, and subalpine. Our specific objectives were:

1. Estimate mulching treatment variability by measuring material depth, density, and distribution based on forest type.
2. Determine the ecological effects of the treatments in relation to forest type (pinyon-juniper, ponderosa pine, mixed conifer, and lodgepole pine).
    a. Understory vegetation (species richness, invasive species, cover and biomass)
    b. Tree recruitment
    c. Soil nitrogen availability
    d. Soil moisture and temperature
3. Assess the longevity of these treatments by establishing long-term monitoring plots for future study and by measuring tree recruitment for this study.
4. Determine the effects of chipping and mastication treatments on fuels, and assess how the rearrangement of woody biomass is likely to affect fireline intensity, rate of spread, crown fire behavior, smoldering of surface fuels, and heating.
5. Compare carbon balance and storage of mulching treatments with untreated areas.
6. Develop simple protocols that can be used by managers and citizen volunteers to help assess ongoing and future treatments.

STUDY DESCRIPTION AND LOCATION

Study sites and design

Eighteen sites were 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*), 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 and first measured in 2007 or 2008 (Table 1).

A total of five sites were located in the lodgepole pine ecosystem, with two sites on the western side and three sites on the eastern side of the continental divide. Lodgepole pine was the dominant (>95%) overstory tree species. Elevations for the sites ranged from 2600 to 2800 m (Table 1). Annual precipitation is 508 mm and falls as snow from September to May and rain in the summer months (WRCC, 2009). Average 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).

Three sites were established in the mixed conifer ecosystem. This ecosystem lies between the lower elevations where ponderosa pine forests dominate and the upper elevations where lodgepole pine or subalpine species dominate. Tree species dominance was mixture of lodgepole pine, limber pine, Douglas-fir, and ponderosa pine (Table 1). This dominance is often a function of site disturbance history, soils, and moisture (Whitfield, 1933; Peet, 1981). Elevations for our sites ranged between 2700 and 2900 m (Table 1). Annual precipitation for ranged between 610 to 660 inches and falls as snow from September to May and rain in the summer months (WRCC, 2009). Average maximum and minimum temperatures are approximately 10.5 and -5°C, respectively. Localized presettlement fire regime studies are still in progress, but other mixed conifer forests tend to have a mixed severity fire regime which varies in space and time (Arno, 2000).

Four sites were established in the ponderosa pine ecosystem. Ponderosa pine was the dominant trees overstory species with various amounts of Douglas-fir (Table 1). Elevations ranged from 2100 to 2360 m. Annual precipitation ranges between 406 and 560 mm and falls as snow from September to May and rain in the summer months (WRCC, 2009). Average 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).

Six sites were established in the pinyon pine/juniper ecosystem and they were distributed throughout central and western Colorado. Elevations ranged from 1915 to 2400 m. Juniper species dominated four of the six sites. Annual precipitation ranged between 254 and 483 mm, with snow falling from October to May and monsoonal rain in the summer (WRCC, 2009). Average 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 an untreated reference area. Untreated sites were located within 1 km of treated sites, on sites with similar aspect, elevation, soils, and forest type. Pre-treatment surveys and post-treatment stump measurements were used to verify similarities between untreated and mulched areas. In the summer of 2007 and 2008, we established three 50-m permanent transects in each of the treated and untreated areas of the 18 study sites. Transect orientation was selected using a randomly selected compass bearing.

Table 1: Site information for the 18 study sites.

Dominant Tree Species (>10 cm dbh)	Elevation (m)	Location	Site Name	Treatment Year	Measured
Pinus contorta (100%)	2800	Arapaho and Roosevelt National Forest, CO (USFS)	Columbine	2005	2007
Pinus contorta (100%)	2690	Arapaho and Roosevelt National Forest, CO (USFS)	Fraser Experimental Forest	2001	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 (68%), *Pseudotsuga menziesii* (32%)	2130	Foxton, CO (private)	Lower North Fork	2005	2007
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)	DaveWood	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.(88%), *Pinus edulis* (12%)	2200	Kremmling, CO (BLM)	Pumphouse	2006	2008
Juniperus sp. (78%), *Pinus edulis* (22%)	2170	Cortez, CO (BLM)	Summit	2005	2007

Field measurements

Ground cover and herbaceous plant cover and composition

Along each transect, 25 1-m^2 quadrats were established to measure ground cover and litter/duff depth. Ocular cover estimates were made for understory plant cover, exposed rock, mineral soil, litter and duff, living woody material (exposed roots, stems, and tree boles including fresh stumps), and dead woody material. Dead woody material was separated into three fuel particle sizes (1-hr + 10-hr = <2.54 cm diameter; 100-hr = 2.55 to 7.6 cm diameter; 1000-hr = >7.6 cm diameter) commonly inventoried using the planar intercept method (Brown *et al.*, 1982). Size class classification of each of the fine woody fuel particles (<100-hr) was made along the narrowest diameter (Kane *et al.*, 2009). Litter and duff depths were measured at the center and at each quadrat corner. Litter was defined as fresh and partially decomposed organic forest debris located above the mineral soil, while duff consisted of highly decomposed organic matter below the litter layer and above mineral soil. In the mulched areas, it was difficult to distinguish litter, duff, and fine woody material (<10-hr) layers due to the mixing of the forest floor caused by the equipment, so we combined our measure of these components (litter+duff+fine woody debris). We also established a 20 x 50 m plot with the permanent transect as the center axis to record presence/absence of plant species. Species composition was recorded in 16 of the 28 ground cover quadrats. Abundance of each species was quantified at the plot scale as the number of quadrats in which that species was present (range 0 to 16).

Fuel loads

Destructive plot-based sampling was used to estimate 1-hr, 10-hr, and 100-hr woody fuel loads (Hood and Wu, 2006; Kane *et al.*, 2009). Three 1-m^2 ‘calibration’ quadrats were established 5-m perpendicular to each transect at 10, 25, and 40 m. Cover estimates and depth measurements were similar to those conducted along the transect. Within untreated areas, all 1-hr + 10-hr and the 100-hr woody fuels inside the frame were collected. Litter and duff samples were collected from a 25 cm x 25 cm frame placed within the 1-m^2 quadrat. All of the 100-hr woody fuels in quadrats within mulched areas were collected from the entire 1-m^2 quadrat. The mulch mixture of litter, duff, and 1-hr + 10-hr fuels in quadrats within the mulched areas was collected from a 25 cm x 25 cm frame placed within the 1-m^2 quadrat. The total mulch depth was also measured within the smaller frame. Once the fuel was collected, each fuel type was weighed, bagged, and brought back to the lab. Due to logistics and space, only a ~200 g subsample of the 100-hr fuels was brought back to the lab. All fuels were oven-dried to a constant dry mass 60°C in a drying oven. Bulk density of each litter and duff sample from the untreated areas within each ecosystem were calculated and used to estimate litter and duff mass. Mulch bulk density (kg m^{-3}) for the mulch mixture was calculated by dividing the total fuel load estimates of (1-hr + 10-hr + litter + duff) by the total mulch depth (m). Once mulch bulk density was calculated, the mulch mixture was separated by fuel size class (1-hr, 10-hr, litter/duff) and each fuel component was reweighed to determine its proportion of the total weight. The 1-hr and 10-hr fuels collected from the untreated areas were also separated and reweighed.

Woody fuels >7.62 cm (1000-hr) loadings were measured along a 4 m x 50 m belt transect. The length, diameter at each end, and the decomposition class of each log encountered was recorded (Bate *et al.*, 2004). The volume of the 1000-hr fuels was calculated as a frustum of a paraboloid (Harmon and Sexton, 1996; Bate *et al.*, 2004) with specific gravity of sound (0.4) and rotten (0.3) wood (Brown and See, 1981).

Herbaceous fine fuel loads were also measured on the calibration quadrats. Ocular estimates of herbaceous cover (aerial coverage for live plants) at the peak of the growing season were estimated for all graminoids and forbs rooted inside the 1-m^2 quadrats. The herbaceous material was clipped within one centimeter of the surface and placed in a bag, oven dried at 60°C for 48 hours, and weighted to the nearest tenth of a gram.

Trees

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 areas and a width of 4-m within the untreated areas. Saplings were enumerated by size (0 to 5 cm and 5 to 10 cm), status, and species. Stump diameters were measured in the mulched areas on belt transects 10-m in width for stumps>10 cm in diameter and 4-m in width for stumps <10 cm in diameter. When possible tree species of stumps was recorded based on bark characteristics. Stumps < 10 cm in diameter were enumerated by size (0 to 5 cm and 5 to 10 cm). Tree height and crown length were measured on a subset of trees within each study area for modeling purposes. Canopy bulk density (CBD) and canopy base height (CBH) were calculated using the Fire and Fuels extension to the Forest Vegetation Simulator (FFE-FVS) (Reinhardt and Crookston, 2003).

Soil nitrogen availability
Soil nitrogen availability 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 2007 and removed in summer 2008.

Experiment to determine the effect of mulch depth on soil nutrients
At a subset of our study sites, adjacent to each transect in the mulched areas, we established three 2 x 2 m experimental plots by removing or adding mulch to establish 3 distinct depths to assess the effect of mulch depth on soil properties. Three replicates were established in the pinyon pine, four replicates in the ponderosa pine, three replicates in the mixed conifer, and six replicates in the lodgepole pine ecosystem. Depths for the pinyon pine were 0 cm, 2.5 cm, and 7.5 cm. Depths for the other ecosystems were 0 cm, 7.5 cm, and 15 cm.

In each plot, we assayed soil N availability using IER bags (n=4 bags per plot). We also continuously monitored soil moisture and temperature at one site in the pinyon pine, ponderosa pine, and lodgepole pine plots.

Treatment effects on carbon storage
To evaluate carbon stocks we used the Forest Vegetation Simulator (FVS) and its component model the Fire and Fuels Extension (FFE). To assess pretreatment carbon stores in the mulched stands, the stand density was reconstructed with the stump density data and surface fuel loads measured on the untreated reference stands were used. Posttreatment carbon stores were estimated with the residual trees and masticated surface fuel loads. Stands were simulated for 25 years to assess how carbon storage changes over time due to tree growth and wood decomposition.

Assessing fire behavior in mulched fuelbeds
We set out to develop custom fuel models for the mulch fuelbeds to assess the potential fire behavior under a variety of weather scenarios corresponding to local conditions during typical and severe fire seasons. With these custom fuel models, we used BehavePlus to predict and compare surface fire rate-of-spread and fireline intensity and the NEXUS software for crown fire behavior for each treatment. We also planned to use FOFEM to make predictions on soil heating, fuel consumption, smoke output, and tree mortality.

To assess the predictions from these models, we planned to install plots in mulched areas that would be broadcast burned according to prescriptions developed and implemented by our partner agencies. We planned to compare the actual fire behavior measurements to the behavior as predicted by our modeling efforts. Finding study sites that would be prescribed burned was problematic for our Colorado sites. However, we were able to set up some plots on the Lower North Fork site. In the mulched area, we set up six 10-m transects with five 1-m^2 quadrats to monitor fire behavior and fuel consumption. Prefire fuel loads were measured on each quadrat and a photograph was taken of each. To measure fuel consumption, a nail was inserted into the ground with the top of the nail flush with the top of the mulch. Unfortunately, the restrictive burn window along the Colorado Front Range has prevented this burn to happen. However, we hope the burn will take place the Spring or Fall of 2010.

Active crown fire risk was assessed for untreated and mulched stands within each ecosystem using FFE-FVS (Reinhardt and Crookston, 2003). Crowning index, the windspeed at 6.1 m above the tree canopy that is required to sustain an active crown fire (Scott *et al.*, 2001), was assessed for each stand in each ecosystem.

Data analysis
For each ecosystem and treatment, linear regression analysis was used to determine the relationship between surface fuel loadings and a predictor variable (fuelbed depth or %cover) from the destructive plot-based sampling estimates. Fuelbed depth was used to predict total mulch fuelbed load (litter + duff + 1-hr + 10-hr). Percent cover of 1-hr + 10-hr was used to predict 1-hr + 10-hr fuel load in untreated areas. Separate equations were developed to predict 100-hr fuel loads for the treated and untreated areas, but both used percent cover of 100-hr as the predictor variable.

Fuels loads were estimated based on transect-level means of substrate cover, mulch fuelbed depth, litter depth, or 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. With the calculated transect-level fuel loads, the contribution each fuel size category made to the total fuel load was calculated for the untreated and treated areas for each ecosystem. To determine changes in fuelbed properties, the ratio of needle litter loading to 1-hr fuel loading was also calculated for untreated and treated areas for each ecosystem.

For each ecosystem, a mixed-model ANOVA was used to detect treatment differences for each fuel loading category, plant cover, species richness, non-native cover and richness, tree density, and soil nitrogen availability. Location and location*treatment were designated as random variable and treatment as a fixed variable. Data that did not meet assumptions of equal variances or normality were log transformed. Differences among treatments were considered statistically significant when $P \leq 0.05$. Mixed effects ANOVAs were performed using PROC GLIMMIX (SAS, 2008). Proc FREQ and Proc SUMMARY were used to determine mulch fuelbed depth distribution and median fuelbed depth for each ecosystem, respectively.

Using data collected at the fine (1 m^2) scale from the treated plots only, we fit 0.9 quantile regressions on the relationship between mulch depth and herbaceous cover for each ecosystem. A significant 0.9 quantile (90th percentile) tells us that the upper limit on herbaceous cover depends on mulch depth or, in other words, that mulch depth becomes limiting when the other factors affecting herbaceous growth are permissive.

Crown fire risk
Proc FREQ (SAS, 2008) was used to determine the percentage of stands in each windspeed category for each treatment.

KEY FINDINGS

Mulched fuel loads can be estimated with fuelbed depth and/or coverage

In general, measurements of fuelbed depth and/or fuel coverage (%) were good predictors of fuel loadings in mulched treatments for each ecosystem (Table 2). Depth was a useful predictor to estimate mulch fuelbed loadings, which consisted of a mixture of litter, duff, 1-hr, and 10-hr fuels. The strongest relationships of depth and mulch fuelbed loadings occurred in the lodgepole pine, ponderosa pine, and pinyon pine/juniper ecosystems with coefficient of determination values (r^2) ranging between 0.84 to 0.90. The relationship in the mixed conifer ecosystem was moderate, with a r^2 of 0.58. Estimates of 100-hr coverage to predict mulched treatment 100-hr fuel loadings were also good with r^2 ranging from 0.75 to 0.82 (Table 2).

To be consistent, we chose to measure untreated fuel loads using the same methodology (% cover-fuel load) as we used in the mulched areas. Separate equations were developed for the untreated fuels because the fuel particle sizes and shapes were not altered. Predicting 1-hr and 10-hr fuel loads based on %cover was variable (Table 2). Pinyon pine/juniper, mixed conifer, and lodgepole pine ecosystems had moderate success with r^2 values ranging from 0.37 to 0.56. Ponderosa pine had a stronger relationship, with a r^2 value of 0.78 (Table 2). Percent cover of 100-hr fuels in untreated areas was a good predictor in untreated areas (r^2 values ranged between 0.84 to 0.97).

Table 2: Linear regression results between fuelbed depth (cm) or fuel cover (%) and litter/woody fuel load (kg/m^2) across mulched sites in four ecosystems in Colorado. Linear regression form: $y = b_o + b_1(x)$. Mulch fuelbed = Litter + Duff + 1h +10 h fuels. All equations were significant (P<0.001).

Lodgepole pine							
Fuel type	Treatment	b_o	b_1	Predictor variable	r^2	n	RMSE
Mulch fuelbed[a]	Mulched	-0.3858	1.5038	Depth (cm)	0.84	49	2.3
100 h	Mulched	0.0385	0.0927	% cover 100h	0.75	39	0.316
1 + 10 h	Untreated	0.0579	0.0410	% cover 1 + 10 h	0.56	41	0.123
100 h	Untreated	-0.0499	0.1663	% cover 100h	0.94	42	0.198
Mixed Conifer							
Fuel type	Treatment	b_o	b_1	Predictor variable	r^2	n	RMSE
Mulch fuelbed[a]	Mulched	-1.667	1.8076	Depth (cm)	0.58	26	3.34
100 h	Mulched	0.0892	0.115	% cover 100h	0.81	26	0.36
1 + 10 h	Untreated	0.0234	0.0207	% cover 1 + 10 h	0.41	27	0.057
100 h	Untreated	-0.0026	0.1005	% cover 100h	0.84	25	0.04
Ponderosa pine							

Fuel type	Treatment	b_o	b_1	Predictor variable	r^2	n	RMSE
Mulch fuelbed[a]	Mulched	-0.2559	1.4315	Depth (cm)	0.86	35	1.8
100 h	Mulched	0.0367	0.1144	% cover 100h	0.76	36	0.315
1 + 10 h	Untreated	-0.0004	0.0323	% cover 1 + 10 h	0.78	34	0.051
100 h	Untreated	-0.0159	0.1156	% cover 100h	0.93	35	0.07
Pinyon pine							
Fuel type	Treatment	b_o	b_1	Predictor variable	r^2	n	RMSE
Mulch fuelbed[a]	Mulched	-0.1050	1.5904	Depth (cm)	0.90	27	1.48
100 h	Mulched	0.1097	0.1395	% cover 100h	0.82	53	0.32
1 + 10 h	Untreated	0.0711	0.0245	% cover 1 + 10 h	0.37	49	0.124
100 h	Untreated	0.0005	0.1116	% cover 100h	0.97	48	0.014

[a]To break down the fuel loads into litter, duff, 1h and 10h fuel size classes apply these proportions to the predicted estimates of mulch fuelbed load (kg/m^2)
Lodgepole pine: litter = 0.18; duff = 0.20; 1h = 0.29, 10h = 0.33.
Mixed conifer: litter = 0.29; duff = 0.20; 1h = 0.25, 10h = 0.26.
Ponderosa pine/Douglas-fir: litter = 0.27; duff = 0.21; 1h = 0.16, 10h = 0.36.
Pinyon pine/Juniper: litter = 0.26; duff = 0.15; 1h = 0.23, 10h = 0.36.

Fuel loads increased substantially and fuel bed characteristics changed
Mulching substantially increased surface fuel loads in all of the ecosystems (Table 3). However, the magnitude of the total increase differed among the ecosystems (mixed conifer > lodgepole pine > ponderosa pine > pinyon-juniper). Average total woody fuel loads in the untreated areas ranged between 7 to 12 Mg ha^{-1} and increased to 27 to 63 Mg ha^{-1} in mulched areas (Table 3). Large diameter fuels (>7.62 cm; 1000-hr) represent about 35 to 69% of the total woody fuel load in the untreated areas, but only 8 to 14% in the mulched areas. The majority of woody fuels in mulched areas were 1-hr and 10-hr fuels (<2.54 cm in diameter), composing between 67 to 78% of total woody fuel loadings. One concern often expressed while discussing mulching treatments is the retention of coarse woody debris (1000-hr) for its ecological benefits (Brown *et al.,* 2003). We detected no significant difference in the amount of 1000-hr fuels among untreated and mulched stands in any of the ecosystems studied. Needle litter mass increased significantly in the mulched areas of the mixed conifer, ponderosa pine, and pinyon pine/juniper, but not in the lodgepole pine ecosystems (Table 3). Average herbaceous fuel loads increased significantly in the ponderosa pine and pinyon pine/juniper, but did not significantly increase in the lodgepole pine or mixed conifer ecosystems (Table 3). Median fuelbed bulk densities in mulched areas were similar among ecosystems, ranging between 137 and 150 kg m^{-3}. The increased surface woody fuel component in mulched areas corresponds to a shift from a needle fuelbed to a compact woody/needle fuel bed as indicated by the ratio of needle litter fuel load to 1-hr fuel loads (Table 4).

Table 3: Mean (and standard error) fuel loads for untreated and mulched areas (surface by timelag fuel moisture class, ground, and herbaceous) of four coniferous ecosystems in Colorado.

	Lodgepole pine		Mixed conifer		Ponderosa pine		Pinyon pine	
	Untreated	Mulched	Untreated	Mulched	Untreated	Mulched	Untreated	Mulched
	(Mg ha^{-1})							
Litter	12.0 (1.4)	10.2 (1.2)	13.2 a (0.37)	27.7 b (0.39)	10.5 a (2.9)	13.6 b (2.6)	6.0 a (1.1)	8.6 b (1.6)
Duff	14.2 (3.4)	11.5 (2.2)	12.8 (3.2)	19.15 (4.6)	8.7 (4.0)	10.5 (2.2)	4.2 (1.5)	4.9 (2.2)
1-hr	1.04 a (0.22)	16.9 b (5.9)	0.64 a (0.04)	23.03 b (8.79)	0.54 a (0.15)	8.0 b (1.9)	1.08 a (0.17)	7.81 b (2.2)
10-hr	0.83 a (0.04)	19.3 b (2.6)	0.80 a (0.09)	24.5 b (5.7)	0.72 a (0.20)	18.02 b (3.3)	1.09 a (0.21)	12.0 b (2.5)
100-hr	3.5 (1.3)	5.2 (0.9)	1.07 (0.25)	10.8 (4.1)	2.45 a (1.04)	7.4 b (1.0)	1.02 a (0.50)	4.15 b (0.6)
1000-hr	2.9 (0.65)	5.32 (2.02)	4.93 (1.47)	5.03 (2.25)	8.29 (3.49)	5.27 (0.67)	4.15 (2.2)	3.18 (1.24)
Total Woody	8.3 a (1.5)	46.7 b (8.6)	7.43 a (1.7)	63.4 b (12.2)	12.0 a (4.7)	38.7 b (5.0)	7.3 a (2.8)	27.2 b (3.6)
Herbs	0.08 (0.05)	0.16 (0.06)	0.06 (0.02)	0.11 (0.05)	0.11 a (0.03)	0.23 b (0.10)	0.26 (0.08)	0.39 (0.07)

Mean values in a row within an ecosystem followed by different letters are significantly different (P<0.05)

Table 4: Ratio of needle litter fuel loads to 1 hour fuel loads. Ratios were significantly different (P<0.0002) between untreated and mulched areas for lodgepole pine, mixed conifer, ponderosa pine, and pinyon pine.

Ecosystem	Untreated	Mulched
Lodgepole Pine	11.5	0.60
Mixed Conifer	20.6	1.2
Ponderosa pine	19.4	1.7
Pinyon pine	5.6	1.18

Mulch depth distribution varied across ecosystems (Fig. 1). Mulch depth in the pinyon pine/juniper stands ranged between 0 and 9 cm (Fig. 1) with a median depth of 1.4 cm. Both lodgepole pine and ponderosa pine stands had mulch depths that ranged between 0 and 13 cm (Fig. 1), with median mulch depths of 3.8 and 3.3, respectively. Mixed conifer stands had the deepest mulch. Mulch depths ranged from 0.5 to 15 cm, with a median mulch depth of 6.0 cm (Fig. 1).

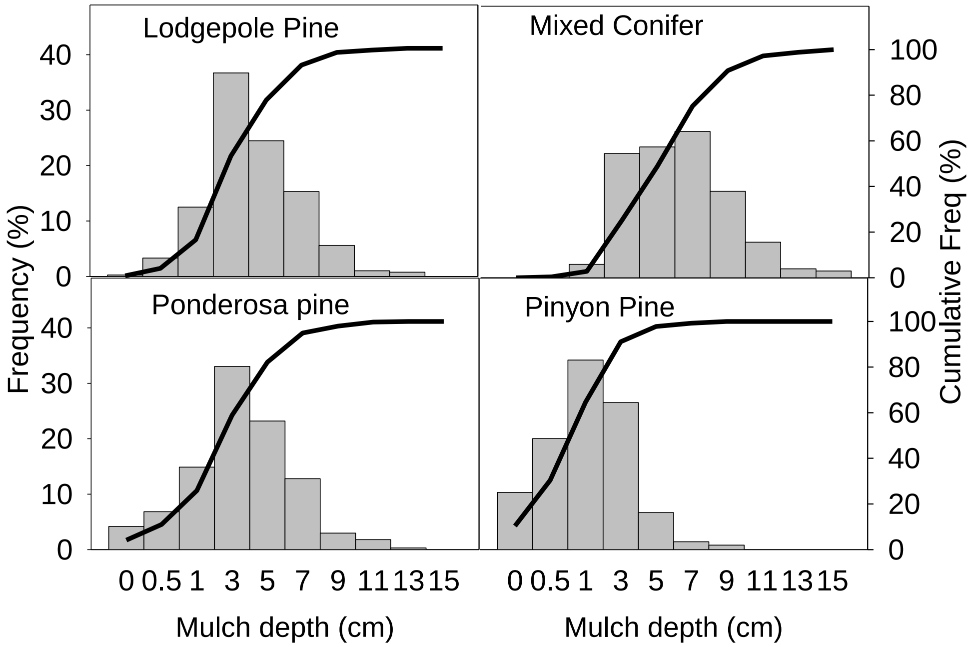

Figure 1: Frequency (bars) and cumulative frequency (line) distribution of mulch depth (cm) at the 1 m^2 scale for mulched study areas in lodgepole pine (n=5), mixed conifer (n=3), ponderosa pine/Douglas-fir (n=4), and pinyon pine/juniper (n=6).

Current surface fire behavior models are inadequate to predict fire behavior in mulched fuelbeds

Our surface fire modeling attempts with BehavePlus in mulched fuelbeds were unsuccessful. In most cases, BehavePlus predicted no fire spread or flame lengths. Knapp et al (2008) suggested that the Rothermel equations used in BehavePlus to predict fire spread may be overly sensitive to the fuelbed bulk density of the mulch fuelbeds and we agree. In addition, the surface area to volume ratio of the mulch fuelbed is likely much lower than the default values BehavePlus currently uses. Although we did not measure the surface area to volume ratio, it is obvious these numbers would be much lower in mulched fuelbeds due to the differences in the ratio of needle litter to 1 hour fuels (Table 4).

Unfortunately, we are still awaiting the fire behavior monitoring plots we installed in the mulched area to be burned. We are hopefully they will be burned in the Spring or Fall of 2010. Also, we are currently seeking opportunities on other sites to observe fire behavior. With these observations we will attempt to choose among the existing fire behavior models that best estimate the observed fire behavior (Knapp et al 2008). If our fire behavior monitoring plots do get burned, we will submit the results to Fire Management Today or another journal describing the comparison between observed fire behavior and modeled fire behavior.

Alteration of forest structure led to lower active crown fire potential

Untreated stands were dense with various amounts of live and dead trees, depending upon the ecosystem (Table 5). Total tree basal area in untreated stands of lodgepole pine and mixed conifer exceeded 30 m^2 ha^{-1}, with high densities of live and dead trees found in both tree size classes (>10 cm and <10 cm dbh). Total basal area in untreated stands of the ponderosa pine ecosystem was 24.3 m^2/ha with most of the tree density found in the <10 cm dbh size class. Pinyon pine/juniper stands had total basal area of 20.4 m^2 ha^{-1} with moderate densities of live and dead trees found in both tree size classes.

Mulching treatments reduced tree basal area and trees per hectare in each ecosystem compared to the untreated control (Table 5). Total tree basal area in the mulched treatments ranged between 4 and 11 m^2/ha, 47 to 89% lower than the untreated controls. Lodgepole pine and mixed conifer had the greatest absolute reduction in basal area, followed by ponderosa pine and pinyon pine/juniper. Total tree density was 69 to 97% lower in the mulched treatments, with low densities of standing dead material.

Table 5: Mean (and standard error) stand and canopy fuel characteristics. BA=Basal area, TPH=trees per hectare, QMD = quadratic mean diameter, CBH = canopy base height, and CBD = canopy bulk density.

	Lodgepole pine		Mixed conifer		Ponderosa pine		Pinyon pine	
	Untreated	Mulched	Untreated	Mulched	Untreated	Mulched	Untreated	Mulched
BA >10 cm dbh	35.3 a (3.3)	10.2 b (2.1)	36.6 a (4.3)	4.2 b (2.5)	24.3 a (3.7)	11.0 b (2.5)	20.4 a (4.3)	11.4 b (4.5)
BA <10 cm dbh	3.5 a (0.9)	0.3 b (0.2)	1.9 (0.5)	0.9 (0.6)	2.6 a (1.0)	0.15 b (0.1)	1.7 a (0.3)	0.4 b (0.2)
TPH >10 cm dbh	1691 a (132)	383 b (89)	1118 a (264)	55 b (28)	659 a (125)	180 b (79)	580 a (116)	203 b (54)
TPH < 10 cm dbh	1107 a (216)	89 b (54)	776 (211)	291 (184)	1598 a (495)	69 b (59)	671 a (118)	193 b (75)
QMD (cm)	5.7 a (0.3)	6.9 b (0.4)	7.0 a (0.5)	11.9 b (0.8)	6.0 a (0.9)	11.4 b (1.4)	6.2 (0.6)	7.9 (1.2)
CBH (m)	5.8 a (1.4)	7.7 b (1.1)	2.5 a (0.4)	5.1 b (0.6)	2.3 (0.7)	5.4 (1.1)	3.3 (0.6)	3.9 (0.8)
CBD (kg m^{-3})	0.15 a (0.03)	0.04 b (0.01)	0.14 a (0.01)	0.01 b (0.009)	0.12 a (0.02)	0.04 b (0.01)	0.02 (0.006)	0.007 (0.002)

Mulching treatments significantly increased quadratic mean diameter and canopy base height of residual trees, while reducing canopy bulk density for all ecosystems except pinyon-pine/juniper (Table 5). Windspeeds required to sustain an active crown fire decreased with mulching treatment for each ecosystem (Fig. 2). The majority of untreated stands within the lodgepole pine, mixed conifer, ponderosa pine ecosystems required windspeeds less than 16 km h^{-1} to sustain an active crown fire (Fig. 2). The mulching treatment reduced the active crown fire risk by increasing the required windspeeds to >56 km h^{-1} in the majority of mixed conifer and ponderosa pine (Fig. 2), however 50% of the lodgepole pine stands still had crowning index of 40 km h^{-1} (Fig. 2). The untreated stands within the pinyon pine/juniper ecosystem had crowning indices that exceeded 48 km h^{-1}, presenting a low to moderate active crown fire risk (Fig. 2). After the mulching treatment, 100% of the stands had crowning indices >72 km h^{-1} (Fig. 2).

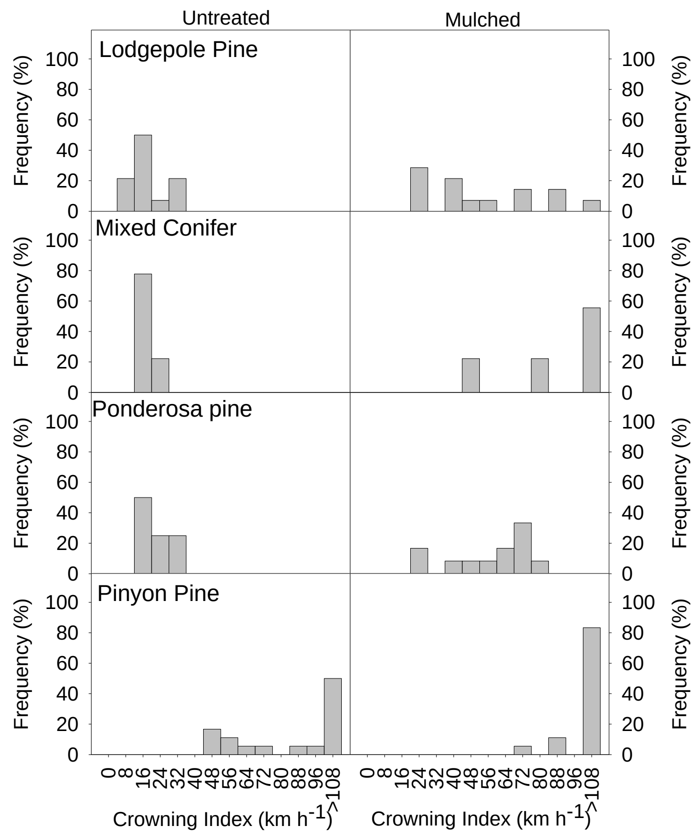

Figure 2: Frequency distribution for the windspeed at 6.1 m above the canopy that is required to sustain an active crown fire for each transect in the untreated and mulched study areas for lodgepole pine (n=5), mixed conifer (n=3), ponderosa pine/Douglas-fir (n=4), and pinyon pine/juniper (n=6).

Effect of mulching on vegetation response was variable

We examined how the combination of overstory thinning and broadcast mulching affected understory vegetation. First, we asked whether deep layers of mulch suppress understory vegetation. Pinyon-juniper and ponderosa had significant 0.9 quantiles (Figure 3), indicating that the upper limit on herbaceous cover depends on mulch depth in those ecosystems. Data from mixed-conifer and lodgepole seemed, at first glance, to demonstrate a similar relationship between mulch depth and herbaceous cover (Figure 3). However, the 0.9 quantiles were not significant; suggesting that something other than mulch limits understory herbs in these ecosystems.

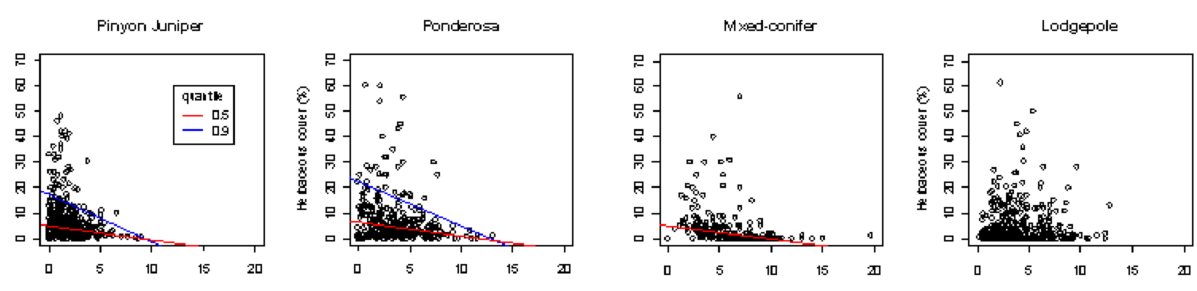

Figure 3. Quantile regressions between mulch depth and herbaceous cover at the plots scale. Only significant slopes for the 0.5 quantile (median) and 0.9 quantile (90th percentile) are shown.

At the operational (plot) scale, however, thinning and mulching does not suppress understory vegetation in any of the four ecosystems (Figure 4). Significant increases in herbaceous cover were observed in pinyon-juniper (p=0.03) and ponderosa (p = 0.002), and cover tended to be higher in treated mixed-conifer and lodgepole plots (Figure 4) but was not significantly different from controls. Shrub cover, and total cover (herb + shrub) were not different between treatments in any ecosystem.

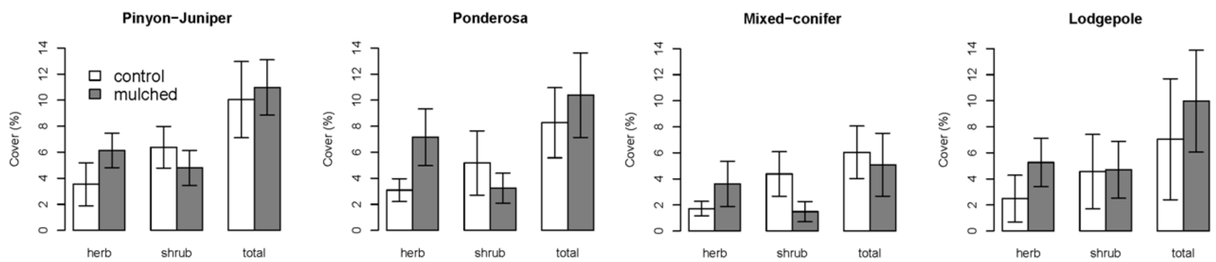

Figure 4. Herbaceous, shrub, and total understory plant cover in untreated and mulched areas.

These results suggest that increases in resources associated with canopy thinning outweigh the suppressive effects of deep mulch depths on herbaceous vegetation. Mulch is not evenly distributed across the forest floor; instead, the mulch is scattered in a mosaic, leaving some small patches with no mulch, and other patches with various depths of mulch (Figure #; mulch depths). In pinyon-juniper and ponderosa ecosystems, herbaceous cover in the areas with low or moderate chip depths is enhanced sufficiently to compensate for deep mulch areas where understory vegetation is suppressed. On the other hand, in mixed-conifer and lodgepole, we failed to detect an enhancement of understory vegetation in treated plots relative to untreated plots. We believe that, because the overstory of lodgepole and mixed conifer are naturally dense compared to ponderosa and pinyon-juniper, these ecosystems may lack sufficient understory flora and/or seed bank to respond rapidly to canopy reduction.

Shrub cover tends to be lower in mulched areas relative to untreated areas, as would be expected given that mulching machinery usually targets the shrub layer. However, our failure to detect a significant reduction in shrubs following treatment may indicate that some species of shrub, such as the oak species in the pinyon-juniper ecosystem, tend to resprout after they are masticated. For example, in two of the pinyon-juniper sites that had a component of gambel oak (*Quercus gambelii*), stem densities exceeded 24,000 stems/ha within 2 to 4 years post-treatment. In other ecosystems, the shrub component is relatively sparse and highly spatially variable, leading to large error bars and a difficulty in detecting significant reductions in shrub cover.

At the subplot (1 m^2) scale, no ecosystem showed differences in species richness between treatments. At the plot scale, only pinyon-juniper had higher richness in mulched (avg. 20 species per plot) than untreated (15 species per plot; p=0.04). In this ecosystem, release from competition from a dense overstory may have allowed new species to grow in treated areas.

No ecosystem showed differences in exotic plant cover between treatments. In the ponderosa ecosystem, however, non-native species richness was higher in treated stands (p=0.01). Even though we detected few statistical relationships between treatment and non-native abundance or richness, we suggest there are reasons to be concerned about possible longer-term problems with non-native plants. At the ecosystem level, exotic species were observed more often in mulched areas than in untreated areas (Table 6). These species were relatively infrequent, sometimes occurring on a small subset of sites and/or plots at a site, and occurred at low abundance on average. However, these species have the potential to increase in abundance with time.

To our surprise, while cheatgrass (*Bromus tectorum*) was present at three of the pinyon-juniper study sites, it occurred at the same abundance in mulched and untreated areas, and did not invade treated sites where it didn't occur in control sites. Canada thistle (*Cirsium arvense*) appears to be the most problematic post-treatment invader, occurring in mulched areas of all four ecosystems. This species is likely not responding to mulching treatments *per se*, but rather to the disturbance and increase in resources as a result of mechanical thinning.

Table 6. Exotic species observed across the four ecosystems

Ecosystem	Untreated	Mulched
Pinyon-Juniper	• 6 species • **cheatgrass** abundant at 3 sites	• 16 species • **cheatgrass** at essentially the same sites
Ponderosa	• essentially absent	• 11 species • **Canada thistle, prickly lettuce, mullein, dandelion** most common
Mixed-conifer	• none	• 4 species • **Canada thistle** most common
Lodgepole	• essentially absent	• 6 species • **Canada thistle** most common

Tree regeneration was variable

Tree regeneration in mulched pinyon pine, ponderosa pine, and mixed conifer stands was lower or similar to that found in the untreated stands, but lodgepole pine stands contained considerably more tree germinants in mulched areas than in the untreated controls (Fig 5). Assessment of tree regeneration at each site indicated that the regeneration response was variable (Fig 6). It is unclear whether the variability in seedling regeneration in the mulched areas was due to (1) lack of exposed mineral soil seedbed, (2) favorable microhabitat conditions created by the mulch, (3) variability in annual seed production, (4) climatic conditions since treatment or (5) an ecosystem specific response.

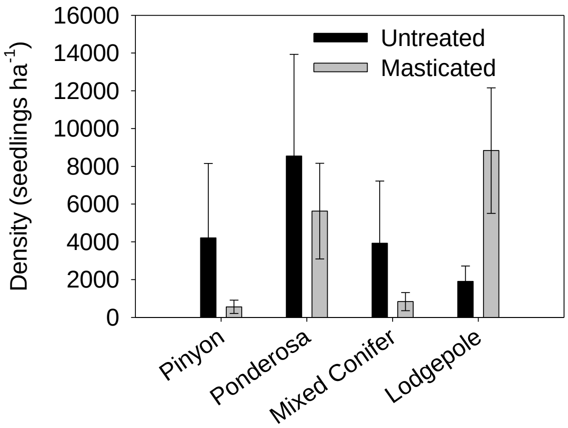

Figure 5: Average seedling regeneration densities 2 to 4 years post-mulching in lodgepole pine (n=5), mixed conifer (n=3), ponderosa pine/Douglas-fir (n=4), and pinyon pine/juniper (n=6).

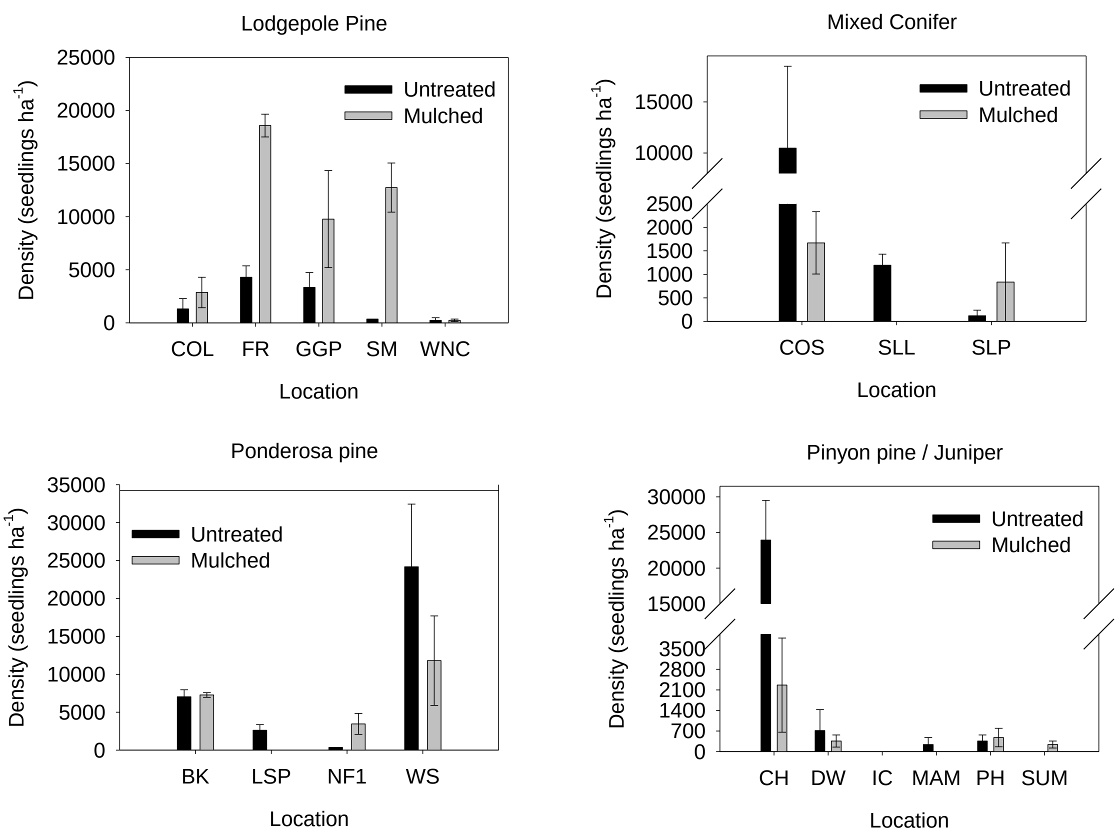

Figure 6: Average seedling regeneration densities 2 to 4 years post-mulching at each individual site.

Effect of Mulch Treatment on Soil Nitrogen are Mixed

At the operational scale, mulch had few negative effects on either ammonium (NH_4-N) or nitrate (NO_3-N) and in some cases increased these forms of plant available nitrogen. Mulching decreased IER N significantly at three sites, increased it at four sites, and had no effect on the remaining 56% of the sites. Positive effects of mulch occurred in ponderosa and mixed conifer ecosystems and negative effects occurred in pinyon-juniper and lodgepole ecosystems (Fig. 7).

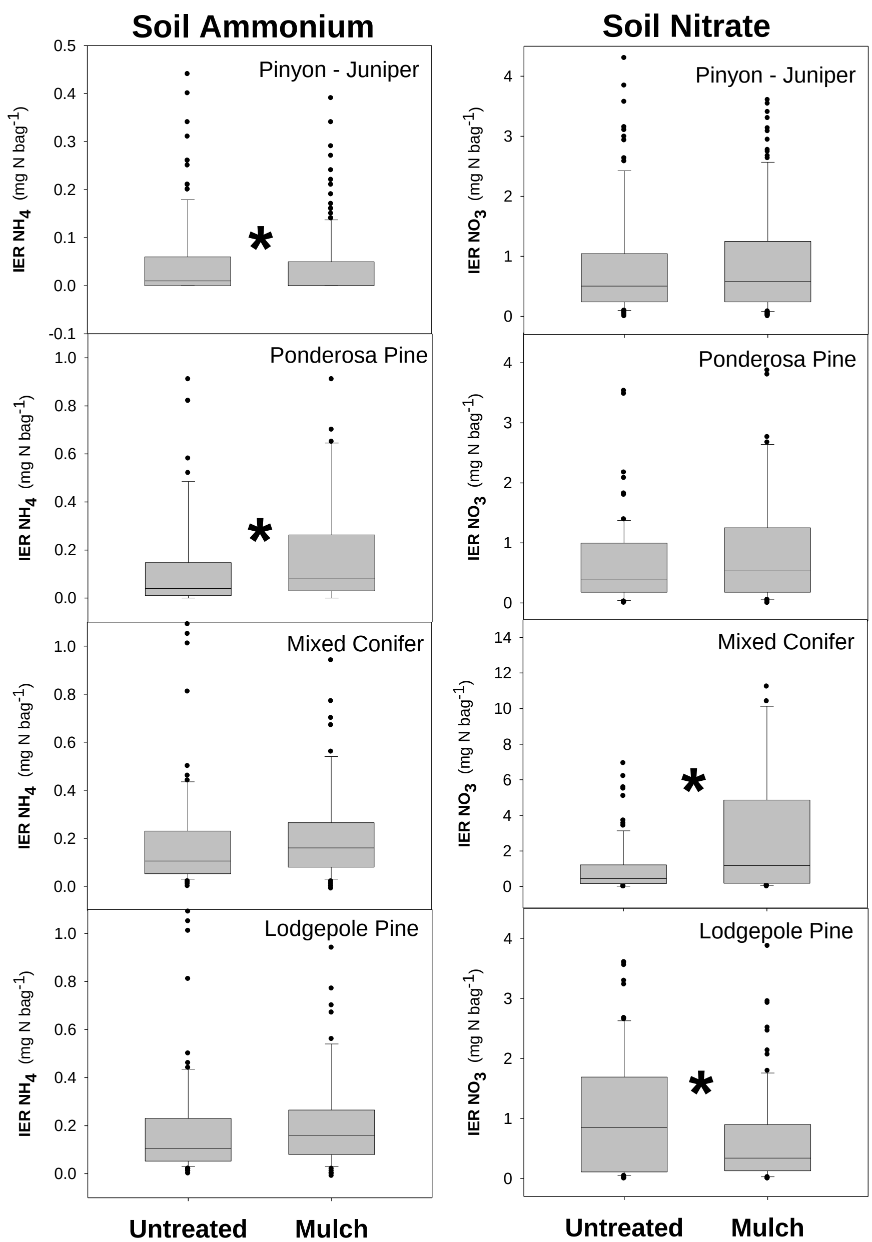

Fig. 7. Ion exchange resin ammonium and nitrate in untreated and mulched stands in four conifer ecosystems. Box plots show median, 25th and 75th percentiles (box), 10th and 90th percentiles (whiskers) and outliers (filled circles). Asterisks denote significant effect of mulch on IER N.

Effects on Soil N Increased with Mulch Depth

In contrast to the subtle effects of mulch applied operationally, deep mulch beds created for plot scale comparisons had larger effects on soil N (Fig. 8). Similar to the operational scale, negative mulch effects occurred in lodgepole and pinyon-juniper ecosystems, though differences were more sizeable. Beneath deep mulch beds IER ammonium and nitrate decreased by 52 and 67% in the lodgepole sites compared to untreated areas and by 94 and 79% in the pinyon-juniper sites, respectively. Deep mulch had no effect in ponderosa sites, but it increased IER nitrate nearly 3-fold in mixed conifer sites.

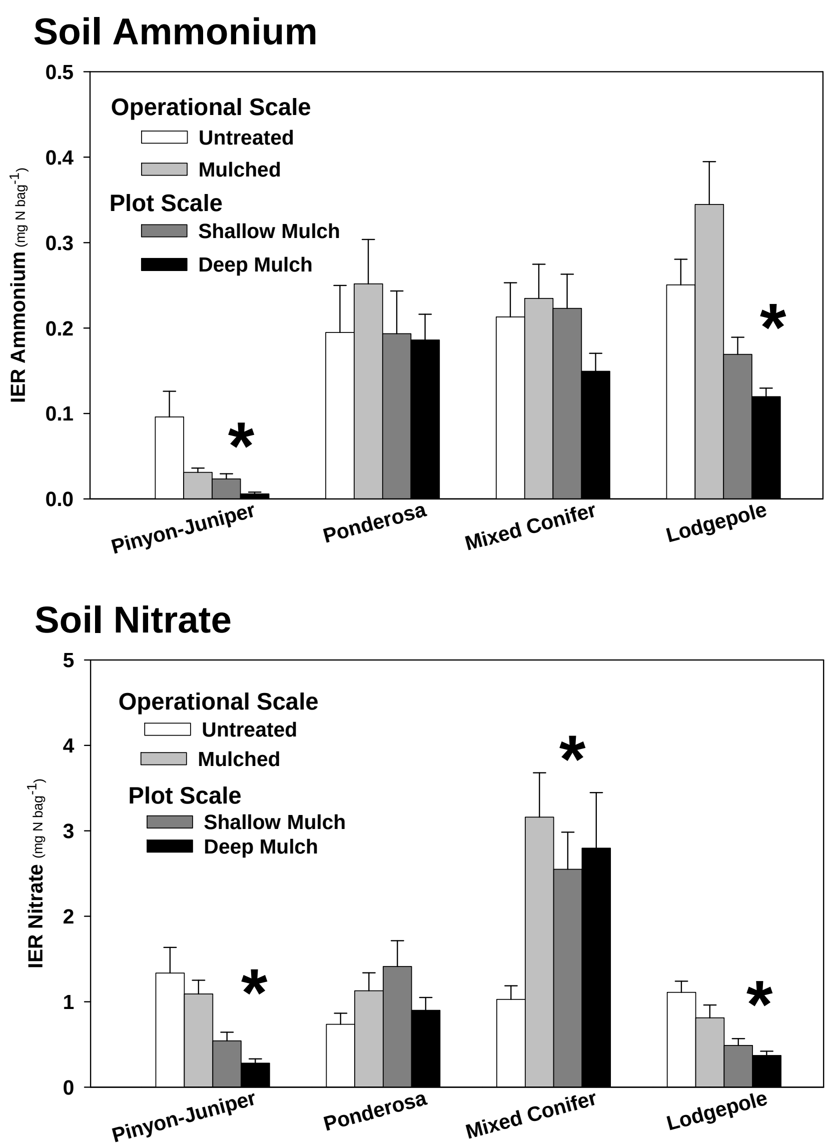

Fig. 8. Ion exchange resin ammonium and nitrate in untreated and operationally-mulched stands and experimentally-constructed mulch beds. Bars are means and SE. Asterisks denote significant effect of mulch depth on IER N.

The Nitrogen Added in Wood Mulch is Unavailable to Plants

Mulching increases the total mass of the forest floor up to 3-fold, and represents a ~10-fold increase in N contained in 1, 10 and 100 hr size material (Fig. 9). However, the N added by mulching is less than half that contained in the forest floor of untreated forests. The ratio of carbon to nitrogen increases from partially decomposed forest floor and litter of the untreated forest (C:N = 26 and 38, respectively) to the material in the 1,10 and 100 hr fuel size classes; the C:N of the newly applied woody mulch is about 1.5-fold higher than in comparable untreated size classes. Owing to its high C:N ratio, mulch is a source of carbon that stimulates microbial growth and uptake (immobilization) of soil N. Added mulch will remain a sink for N until its C:N ratio approximates that of the forest floor (i.e., 25- 30).

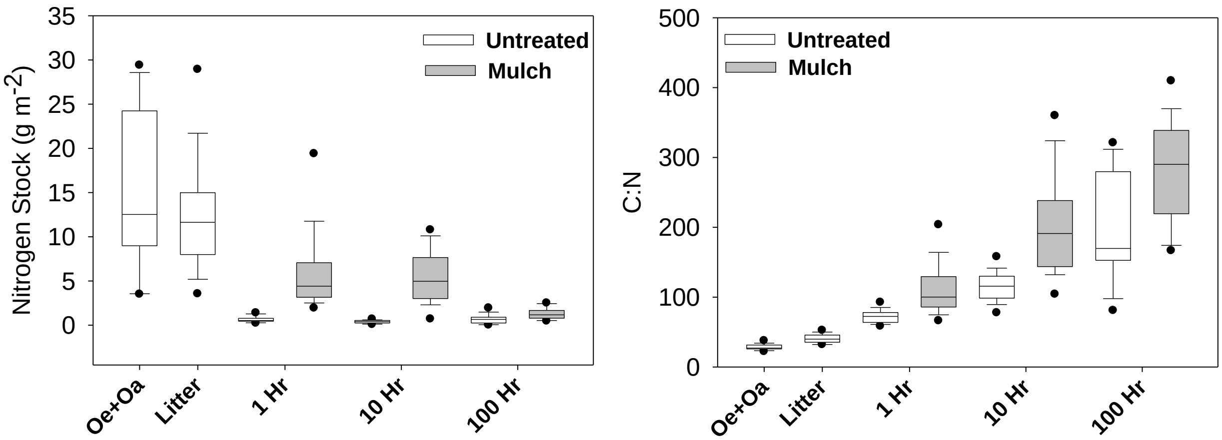

Fig. 9. Nitrogen content and C:N ratio of forest floor and woody fuel classes. Box plots show median, 25th and 75th percentiles (box), 10th and 90th percentiles (whiskers) and outliers (filled circles).

Soil microclimate is altered beneath mulch

Mulch additions reduced soil temperature (Table 7) in the summer months while increasing temperatures in the winter months (Table 7; Figure 10). The biggest mulch influence occurred during the summer months where soil temperatures did not fluctuate as much as the areas without mulch, especially in the pinyon pine/juniper ecosystem. Mulch also increased soil moisture (Fig. 10).

Table 7: Change in soil temperature (degrees C) compared to no mulch in pinyon pine-juniper (PJ) and Lodgepole pine (LPP) for shallow (2.5 cm PJ and 7.5 cm LPP) and deep (7.5 cm PJ and 15 cm LPP) mulch.

Summer Months

PJ	Shallow	Deep
Mean	-3.1	-4.6
Max	-16.5	-21.5
Min	-3.1	-4.2

LPP	Shallow	Deep
Mean	-1.0	-1.2
Max	-3.8	-5.3
Min	-4.1	-5.3

Winter Months

PJ	Shallow	Deep
Mean	+1.2	+1.9
Min	+2.7	+3.2

LPP	Shallow	Deep
Mean	+0.2	+0.4
Min	+0.0	+0.5

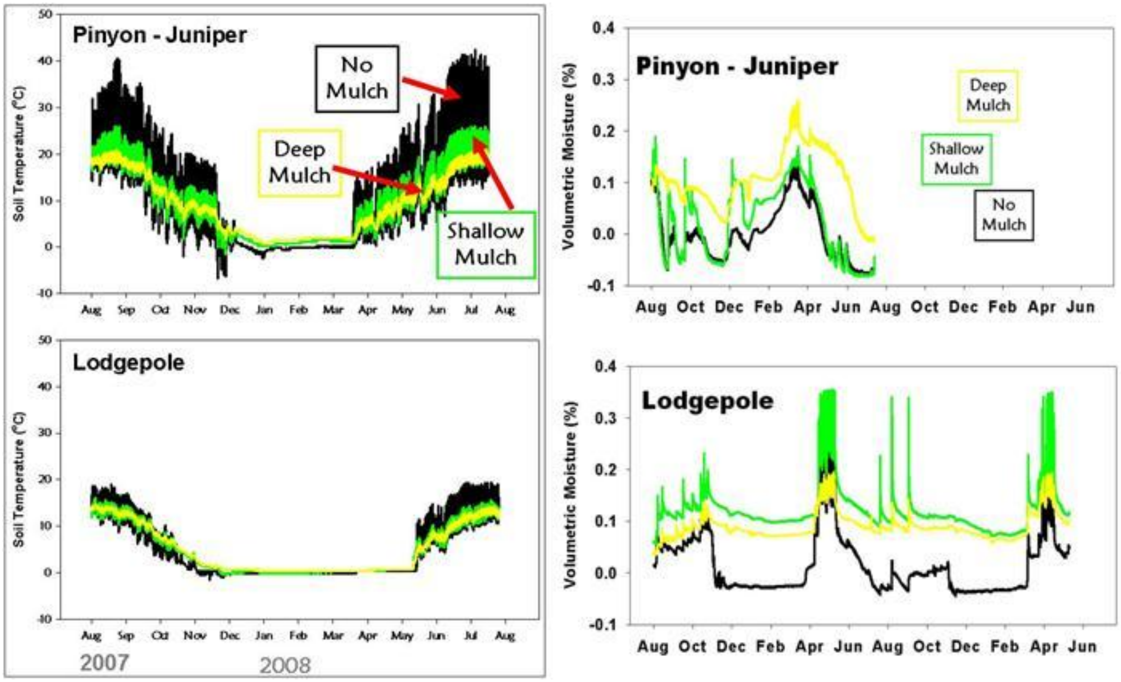

Figure 10: Soil temperature and soil moisture at 5 cm below the mineral soil for various mulch depths in pinyon pine-juniper and lodgepole pine ecosystems.

Carbon storage decreases over time

We expected that total stand carbon would be similar between untreated and mulched stands shortly after fuel reduction activities because the mechanical operations simply convert standing biomass into a surface layer of mulched material. Modeling results from FVS suggest that total stand carbon immediately after treatment was similar between the untreated and mulched stands in the ponderosa pine and pinyon pine ecosystems, but not the lodgepole pine or mixed conifer ecosystems (Fig 11). We are still exploring where the 'missing carbon' might be in the lodgepole pine and mixed conifer sites. We suspect that model inputs such as site index and tree heights specific for these sites are needed for these ecosystems to allow FVS to increase its predictive ability. Such inputs were available for the ponderosa pine and pinyon pine sites. We are currently working on obtaining these inputs for future analysis. In addition, we observed that the decomposition rates modeled with FVS for belowground biomass was similar regardless of ecosystem, which added some error to our estimates.

Nevertheless, an assessment of how carbon storage changes over the next 25 years is still warranted. Carbon continues to accumulate in the untreated stands as trees continue to grow. In contrast, carbon storage decreases through time in the mulched stands due to the decomposition of the woody fuel deposited on the forest floor and the low density of residual trees. The largest difference in the impact of the mulching treatment on carbon storage occurs in the lodgepole pine and mixed conifer sites which had the most trees removed (Table 5) and fuel deposited on the forest floor (Table 3).

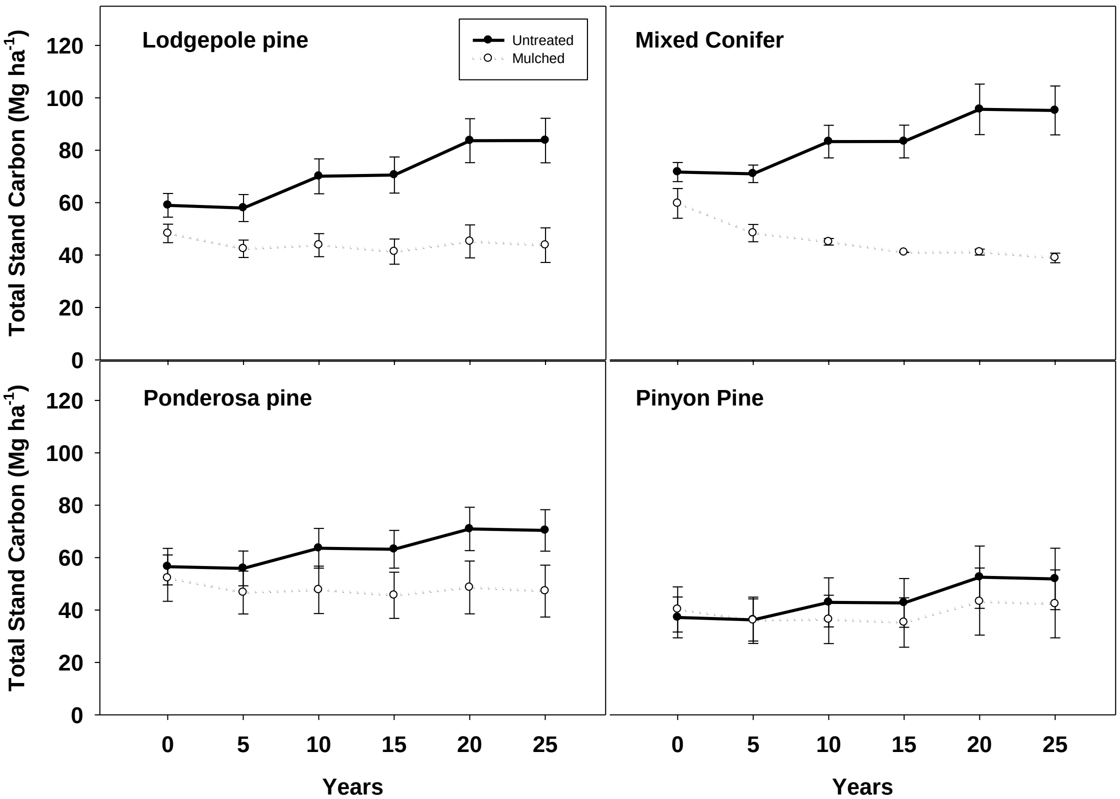

Figure 11: Total stand carbon (aboveground+belowground) in untreated stands compared to mulched stands.

MANAGEMENT IMPLICATIONS

Fuelbed depth or fuel coverage are good predictors of surface fuel loadings in mulched treatment areas.

These alternatives to planar transect sampling provide managers with a fast and easy technique to estimate total surface fuel loads. However, the ability to estimate the proportion of each fuel size class that contributes to the mulched fuel bed will take extra effort, including taking samples, drying and sorting by size class. This extra effort might be impractical for most managers due to constraints in resources. Applying the provided proportion values of each fuel size class associated with the mulch fuelbed equations (Table 2) should help with the estimation. However, the accuracy of the estimations will be subject to the variability in the masticator head used, duration of grinding action, and pretreatment surface fuel loads.

Deposition of large amounts of small material is altering the fuel bed and will likely impact potential surface fire behavior and effects

The majority of woody fuels in mulched areas were 1-hr and 10-hr fuels (<2.54 cm in diameter), composing between 67 to 78% of total woody fuel loadings. Addition of woody material to the needle litter resulted in a shift from a needle fuelbed with typical bulk densities below 100 kg m^{-3} in untreated areas (Brown and See, 1981; van Wagtendonk, 1998; Battaglia *et al.*, 2008) to a compact woody/needle fuelbed. These compact, small particles wood-laden fuelbeds would likely alter potential surface fire behavior and fire effects. The limited studies examining the fire behavior in these compacted mulched fuelbeds find that rate of spread and flame lengths are reduced, but flaming and smoldering duration is increased (Busse *et al.*, 2005a; Glitzenstein *et al.*, 2006; Knapp *et al.*, 2006; Kreye, 2008). Factors such as pretreatment surface fuels, standing biomass, and type of operation equipment could influence post-treatment fuel size and distribution (Harrod *et al.,* 2009; Kane *et al.,* 2009).

Mulching treatments altered stand structure which resulted in lower active crown fire risk by requiring higher windspeeds to sustain active crown fire behavior

Mulching reduced tree density by treating the majority of trees < 10 cm diameter and many of the overstory trees, resulting in a decrease in canopy bulk density, an increase in height to live crown, and an increase in average stand diameter for the lodgepole pine, mixed conifer, and ponderosa pine ecosystems.

Similar results were not observed in pinyon pine/juniper ecosystems. Although tree density was reduced, it did not result in a significant decrease in canopy bulk density or increase in canopy base height. The lack in significant reduction in these canopy characteristics is attributed to the nature of pinyon pine/juniper stand structure. Because untreated stands had low tree density and canopy bulk density the majority of stands already had a low active crown fire risk. Instead, pinyon pine/juniper stands would likely be susceptible to passive crown fire behavior due to its large gaps between trees (Evans, 1988), low canopy base height, and dense shrub layer.

Herbaceous cover is enhanced or unchanged following thinning and mulching

Understory plant response is important to management for at least two reasons. First, understory herbs and shrubs comprise a portion of the surface fuel stratum. For treatments designed to facilitate the reintroduction of a surface fire regime, a positive understory vegetation response might be considered desirable, as it provides flashy fuels that will help to carry a surface fire. In contrast, where surface fire prevention is a management objective, growth of the shrub and herb layer vegetation may be considered undesirable. Second, understory plants comprise the bulk of the floral diversity in most temperate forested ecosystems (Gilliam 2007), and in some ecosystems, particularly those that historically experienced a frequent, surface fire regime, the restoration of a diverse, weed-free herb layer is considered a restoration objective.

In some ecosystems, overstory thinning is not a typical characteristic of the natural disturbance regime. For such ecosystems, including the mixed-conifer and lodgepole from our study, herbaceous vegetation may not respond rapidly to thinning treatments. Enhancement of understory plants may not be an appropriate management objective in such forest types. Whether understory cover in such ecosystems increases with increasing time-since-treatment remains to be determined.

Mulch depth and distribution influence herbaceous vegetation response

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 9 cm in pinyon-juniper and 12.5 cm in ponderosa. (We cannot provide similar estimates for mixed-conifer and lodgepole 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.

Due to the heterogeneous distribution of mulch, herbaceous vegetation was not suppressed at the stand scale. Most pieces of mastication machinery scatter mulch randomly, with little control from the operator. However, with certain pieces of machinery, such as whole-tree chippers, manipulation of mulch depth and distribution by operators may be possible in order to meet management objectives.

Exotic species are infrequent and occur at low abundance in treated areas

Despite their low abundance, non-native species were observed more often in treated areas of all ecosystems. They may become more abundant with time and should be monitored.

Tree regeneration was variable

One key to determining treatment longevity is the rate of recruitment and vertical development of ladder fuels, which directly contribute to both passive and active crown fire risk. The woody residue that results from fuel reduction treatments may create a physical barrier or a nutrient sink that delays plant recruitment or slows growth, thus prolonging the effective treatment period. However, negative mulch effects may be short-lived and plant growth may be stimulated by changes in soil resources and site conditions as mulch layers age. In our current study, the variability in the tree regeneration immediately following the mulching treatment makes it difficult for us to determine fuel treatment longevity. Further study is needed to determine if the variability in response was due to climatic conditions, seed source, and mulch serving as a barrier and/or nutrient sink.

One result that was noticeable was the prolific resprouting (>24,000 stems/ha) of the gambel oak (*Quercus gambelii*) in areas where it was present before the mulching treatment. If these areas are not maintained with more mulching or prescribed fire, the high densities of gambel oak will continue to suppress the herbaceous understory and pinyon pine regeneration. Furthermore, if these oak densities are not managed, similar fuel structures observed in the untreated areas will occur within the next several decades.

The short-term consequences of mulch application on soil nitrogen are depth-dependent and they differ among ecosystems

Mulch treatments alter inputs of N and C and forest floor substrate quality, as well as the abiotic conditions that regulate microbially-mediated processes that control organic matter turnover and the availability of plant available forms of soil N. At the operational-scale, mulching treatments added between 1 and 3 cm of mulch on average (Fig. 1). Treatment application was patchy, so a significant extent of the treated areas received no appreciable mulch addition, especially in the pinyon-juniper and ponderosa pine ecosystems. Thus, it was not surprising that plant-available soil N responded little to mulching treatments at the operational scale. However, these treatments occasionally applied mulch in excess of 10-15 cm in depth, and in plots where we created uniform mulch layers a different story emerged. Plant-available ammonium, measured using ion exchange resins (IER), was significantly lower in mulch beds in all the conifer ecosystems we evaluated compared to untreated areas (i.e., >50% lower). In lodgepole and pinyon-juniper forests, greater mulch depths depressed IER-ammonium further (i.e., 94% lower in pinyon-juniper). We observed a similar decline in IER-nitrate in lodgepole and pinyon-juniper, but not the ponderosa pine ecosystem. We also found that in general mulch elevated soil moisture and depressed soil temperature during the growing season and blunted temperature extremes. As such, mulch maintained an environment more favorable for microbial activity during dry or hot summer periods. Mulch also increased soil temperature during colder months thus extending the period of microbial activity, decomposition, and nutrient turnover. The ecosystem-specific responses to mulching are the likely consequences of differing climatic or soil conditions.

The amount of carbon stored in mulched treatments is reduced

Mulching is an effective fuel management technique to reduce the risk of crown fire, but it also temporally reduces the amount of carbon stored on the landscape because the biomass is not utilized. Instead, the biomass is deposited on the forest floor and the stand loses carbon via decomposition. Although thinning might increase the growth of the residual trees, the overall stand-level growth is not enough to initially compensate for the reduced carbon until the trees re-occupy the site, which could take several decades.

An argument is often made that fuel treatments reduce the potential loss of carbon if a wildfire were to occur. If a crown fire burns through a forest that was thinned to a low density, the fire may move from a crown to a surface fire and many of the trees can often survive the fire. In contrast, many or all of the trees in an unthinned stand will be killed by a crown fire. This contrast in survival has led to the notion that fuel treatments offer a carbon benefit: removing some carbon from the forest may protect the remaining carbon. However, the science on carbon savings through fuel treatments is mixed. Evidence from a landscape-level modeling study suggests that fuels treatments in most forests will lose carbon, even if the thinned trees are used for biomass fuel. Only in sparse forests with very short fire return intervals did the carbon loss in thinning match the carbon savings from less intensive fires. More research will likely be needed to resolve these different conclusions.

RELATIONSHIP TO OTHER RECENT FINDINGS AND ONGOING WORK ON THIS TOPIC

Fuels

Our results combined with similar findings in other ecosystems across the Western U.S. (Hood and Wu, 2006; Kane *et al.*, 2009) suggest that fuelbed depth or fuel coverage are good predictors of surface fuel loadings in mulched treatment areas. As expected, total woody surface fuel loadings substantially increased in the mulched areas of each ecosystem, ranging from 27 to 63 Mg ha^{-1}, similar to other studies (Stephens and Moghaddas, 2005a; Hood and

Wu, 2006; Kane *et al.*, 2009). Our study also demonstrated that in each ecosystem 1-hr and 10-hr fuels contributed the most to the total fuel load in mulched areas, a finding also reported by Kane *et al.* (2009).

Deposition of large amounts of 1-hr and 10-hr fuels in the mulched areas created very different fuelbed characteristics from those observed in untreated areas. Mulch areas in each ecosystem had compact fuelbeds with median fuelbed bulk densities ranging between 137 and 150 kg m^{-3}. These values are similar to mulched fuelbed values reported for other ecosystems across the western U.S. (Busse *et al.*, 2005b; Hood and Wu, 2006; Kane *et al.*, 2009). Quantification of the mulched fuelbed characteristics in the four ecosystems in this study should aid in the modification of current fuel models (Anderson, 1982; Scott and Burgan, 2005) or in the creation of new fuel models. Further research is needed to quantify the potential effects burning in masticated fuelbeds will have on fire behavior, tree mortality, soil nutrient cycling, and other ecological processes.

Vegetation

Only a handful of published studies have investigated understory plant responses to mulching treatments (Collins et al. 2007, Perchemlides et al. 2008, Miller and Seastedt 2009, Wolk and Rocca 2009, Kane et al *in press*). Each of these studies focuses on one site, making it difficult to generalize trends within or among vegetation types. However, collectively the results from these studies support our observation that responses to mulching differ across ecosystems. Increases in understory cover and 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 *in press*), 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 fine scale.

One consistent trend across all studies is an increase in the cover and/or richness of non-native species (Collins et al 2007, Perchemlides et al 2008, Miller and Seastedt 2009, Wolk and Rocca 2009). It appears likely that exotic species increase as a result of thinning operations (soil disturbance, seed dispersal via equipment, increased light and nutrient resources) rather than of mulching *per se* (Perchemlides et al 2008, Miller and Seastedt 2009, Wolk and Rocca 2009).

Nitrogen

We found that for a handful of sites mulch decreased soil N available to plants. Surface application or incorporation of mulch, sawdust and sugar has been shown to reduce inorganic soil N by providing soil microbes a source of readily-available (labile) carbon that stimulates their growth and uptake (immobilization) of soil N (Morgan 1994; Zink and Allen 1998; Binkley et al. 2003; Blumenthal 2009). These forms of carbon addition are commonly used as an ecosystem restoration technique to 'reverse fertilization' and promote establishment of low Ndemanding native plants. Decline in soil N availability following mulching or C addition may effective for months to several years (Reever-Morghan and Seastedt 1999; Baer et al. 2003). While the direct effect of mulch on soil nutrients may be negligible or short-lived (Miller and Seastedt 2009), its influence on understory species establishment may have longer-term biogeochemical consequences. For example, where C additions favor establishment of species with low soil N demand and low tissue N content, plant-soil nutrient feedbacks may that sustain the reduction in soil N availability initiated by mulch addition (Chapin 1980; Tilman and Wedin 1991). Based on our findings it does not seem likely that mulching will have significant consequences on forest productivity in most sites. Nevertheless, since forest growth is commonly limited by N supply, a further reduction in N availability may substantially reduce tree growth in some sites.

FUTURE WORK NEEDED

Fuel beds and fire behavior

One of the objectives of mulching treatments is to reduce the risk of crown fire initiation (passive crown fire); however, we were unable to determine if this type of fire behavior was reduced. Modeling passive crown fire requires the user to choose a fire behavior fuel model (Anderson, 1982; Scott and Burgan, 2005) based on surface fuelbed characteristics. Kane *et al.* (2009) demonstrated that timber, brush, and slash-based fuelbeds (Anderson, 1982; Scott and Burgan, 2005) commonly used to model fire behavior differ substantially from mulched fuelbeds in sites dominated by *Arctostaphylos* and *Ceanothus* shrubs. Attempts to create custom fuel models based on measured mulch treatment fuel loads to model fire behavior and compare the outputs to actual observed fire behavior have been unsuccessful (Glitzenstein *et al.*, 2006). More information on parameters such as fuel loads, fuelbed bulk density, surface to area volume ratios, and fuel size class distribution are needed to develop fire behavior models appropriate for mulched fuelbeds.

There is little available data about the uncertainty associated with fire behavior prediction models in mulched fuel beds. The limited studies examining the fire behavior in these compacted masticated fuel beds find that rate of spread and flame lengths are reduced, but flaming and smoldering duration is increased (Busse *et al.*, 2005; Glitzenstein *et al.*, 2006; Knapp *et al.*, 2006; Kreye, 2008). Understanding the uncertainty related to a fire behavior prediction is essential information lacking within the scientific literature. The combination of custom fuel models along with information about uncertainty in model predictions can allow managers to better interpret modeled crown fire hazards, decide among alternative fuel treatments, and refine go-no-go decisions in prescribed burning operations.

We do not know how readily spot fires ignite from firebrands landing in mulched fuel beds, nor the ability of these fuel beds to generate firebrands. However, spot fires are an important element of fire spread and loss of houses when fires burn in wildland-urban interface areas. Knowledge of the susceptibility of these fuel beds to ignition, their ability to generate firebrands and general fire behavior within these stands needs to be investigated so that managers can make sound decisions about the benefits and consequences of using mulching as a fuels reduction method, especially in the wildland urban interface.

Vegetation

Mulching treatments can cause short-term increases in non-native plant species, but it is unknown whether they will increase or decrease in abundance with time. Managers are concerned about the introduction of non-native plants into masticated areas. Our initial findings indicate that their concern is valid. Although the cover of the non-native plants we found in the masticated areas was low, their presence and potential for expansion is a major concern especially as mulch decomposes and physical barriers are reduced. In the ponderosa pine, mixed conifer, and lodgepole pine ecosystems, we found little evidence of non-native plants in the untreated stands, but between 4 and 11 new species in the masticated areas. In the pinyon pine ecosystem, untreated stands had 6 non-native species, but that number increased to 16 species in the masticated areas. Informal visits to our study sites in 2009 suggest that abundance of non-native plants may be increasing.

Initial herbaceous species composition differed between treated and untreated areas in the ponderosa pine, pinyon pine, and lodgepole pine ecosystems. We expect these differences to become more pronounced through time and, perhaps, to begin to reflect different plant life history strategies. The physical barrier posed by mulch may affect plant regeneration in species with certain traits (e.g., annuals, small-seeded species), and the alterations to soil nutrient and moisture characteristics may favor some species over others. New species that were excluded from the understory due to low light conditions prior to treatment are more likely to disperse to and establish at a site with increasing time since treatment.

The mechanisms governing tree seedling establishment and growth in masticated areas remain unclear. Tree seedling germination can be enhanced after conventional mechanical treatments because the thinning activities typically scarify the forest floor, which provides a mineral soil seedbed (Oliver and Ryker 1990; Shepperd *et al.* 2006) and because thinning brings more light to the understory. However, evidence is equivocal on whether or not the subsequent deposition of a mulch layer due to mulching activities will suppress (Bensen 1982; Resh *et al. in press*) or enhance tree seedling germination (Dierauf and Apgar 1989; Resh *et al. in press*). Our initial assessment found that tree regeneration in masticated pinyon pine, ponderosa pine, and mixed conifer stands was lower or similar to that found in the untreated stands, but lodgepole pine stands contained considerably more tree seedlings in masticated areas than in the untreated controls. It is unclear whether the variability in seedling regeneration in the mulched areas was due to (1) lack of exposed mineral soil seedbed, (2) favorable microhabitat conditions created by the mulch, (3) variability in annual seed production, (4) climatic conditions since treatment or (5) an ecosystem specific response. Understanding the mechanisms that favor and discourage tree seedling germination in masticated areas will improve our understanding of the impacts of mulching treatments on future forest structure and treatment longevity.

Once a tree seedling germinates, its growth rate will influence the time it takes for it to reach a height that connects it to the overstory canopy. Tree seedling growth in masticated areas may be influenced by many factors, including ecosystem, mulch depth, and nutrient availability. For example, studies have observed that lodgepole pine seedling growth was negatively affected by wood chips (Benson 1982; Zabowski *et al.* 2000), whereas Douglas-fir seedlings showed mixed results (Zabowski *et al.* 2000; Roberts *et al.* 2005). Information is lacking for other tree species such as ponderosa pine and pinyon pine, as is site-specific information for all forest types in the southern Rocky Mountains and Colorado plateau. Assessing the impact that mulching treatments have on tree seedling growth will provide useful information for predicting future forest structure, which will inform fuel management decision making.

Nitrogen

The mulching treatment added a substantial amount of 1-hr and 10-hr woody fuel (<2.54 cm in diameter) to the forest floor which resulted in a range of depths within each ecosystem (Fig. 1). The treatments provided a relatively large input of nitrogen (N) to the forest floor, but because of the high carbon (C) to N ratio of the added material (e.g. C:N of 125-175), the woody mulch is resistant to microbial decay and the added N is largely unavailable to plants. Slow mulch decomposition in arid and cold western forests may extend the consequences of this management treatment on plant germination, soil nutrient availability, and plant productivity for many years.

We expect that the short-term consequences from mulching to be depth dependent and to differ among ecosystems. We hypothesize that in moist environments where mulch decomposes, plant N will become more available. In contrast, in places where decomposition and microbial processes are limited, plant N may change little over the time period studied.

The length of time that mulch serves as a nitrogen sink in mulching treatments may have impacts on site productivity, biomass production, and treatment longevity. An experiment that determines the growth response of planted seedlings (to mimic advanced regeneration) to nitrogen availability in mulched treatments could serve as a proxy to determine if nitrogen is still immobilized within the mulch layer as the mulch ages.

Carbon

We recommend that carbon research focus on a landscape scale because carbon loss in thinning needs to be placed in the context of the expected fire frequency and extent. We also recommend that long-term decomposition studies be installed to determine how fast mulched fuels will reduce carbon sequestered.

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DELIVERABLE CROSSWALK TABLE

Proposed	Accomplished/Status
Permanent plot installation	Completed
Sampling protocols	Completed
Management workshop	Planned for January 2010 as part of the 2010 Annual Rocky Mountain Area Fuels and Fire Use workshop, Denver, CO
Field tours	Completed
BMP recommendations	Due to the variability in results and the short-term nature of the project, best management practice recommendations would be premature.
Chipping effects in Rocky Mountain Region RMRS-GTR	In June 2009, we chaired a special session at the North American Forest Ecology Workshop in Logan, UT. Several researchers presented their data from sites across the Rocky Mountain West. We are currently working with these collaborators to synthesize all of our results.
Soil nutrient responses to fuel reduction (peer review)	The majority of the data has been processed and analyzed. The manuscript preparation and submission will occur in Spring 2010
Vegetation responses to fuel reduction (peer review)	Manuscript is in internal review and will be submitted in January 2010 to Forest Ecology and Management
Fuel reduction effects on C and water balance (peer review)	In preparation and will be submitted by Summer 2010

ADDITIONAL DELIVERABLES

Publications

Battaglia, M., Rocca, M.E., Rhoades, C., Ryan, M.G. (*internal review)*. Surface fuel loadings and potential crown fire behavior within mulching treatments in Colorado coniferous forests. To be submitted in January 2010 to Forest Ecology and Management.

Sharik, T.L., Adair, W., Battaglia, M., Baker, F., Comfort, E., D'Amato, A., Delong, C., DeRose, J., Ducey, M., Harmon, M., Levy, L., Logan, J., O'Brien, J., Palik, B., Roberts, S., Rogers, P., Shinneman, D., Spies, T., Taylor, S., Woodall, C., Youngblood, A. (***in press*** International Journal of Forestry Research). Emerging Themes in the Ecology and Management of North American Forests.

If our fire behavior monitoring plots do get burned, we will submit the results to Fire Management Today or another journal describing the comparison between observed fire behavior and modeled fire behavior.

Presentations

93rd Annual Ecological Society of America, August 2008, Milwaukee, WI.
Battaglia, M.A.; Rhoades, C.C., Rocca, M., Canova, N., Wolk, B., and Ryan, M.G. Mulching the forest: Mastication treatment effects on surface fuels and plant cover.

International Association of Wildland Fire Conference, September 2008. Jackson Hole, WY.
Rocca, M., Battaglia, M.A., Wolk, B., Canova, N., Rhoades, C., Ryan, M.G. The effects of chipping and mastication treatments on the forest understory in Colorado forests.

USFS Rocky Mountain Research Station, Forest and Woodland Ecosystem Seminar Series Spring 2009, Fort Collins, CO.
Battaglia, M., Rhoades, C., Rocca, M., Ryan, M.G. Initial changes on the forest floor in mulching fuel reduction treatments.

7th North American Forest Ecology Workshop, Logan, UT
Battaglia, M., Rhoades, C., Rocca, M., Ryan, M.G. Surface fuel loadings in mulching treatments in Colorado coniferous forests.

7th North American Forest Ecology Workshop, Logan, UT
Rhoades, C., Battaglia, M., Rocca, M., Ryan, M.G. Woody mulch effects on soil climate and nitrogen availability in mechanical fuel reduction treatments.

7th North American Forest Ecology Workshop, Logan, UT
Rocca, M., Battaglia, M., Rhoades, C., Ryan, M.G. The effects of mulching treatments in the forest herbaceous layer of Colorado coniferous forests.

94th Annual Ecological Society of America, August 2009, Albuquerque, NM
Battaglia, M., Rocca, M., Rhoades, C., Ryan, M.G. Herbaceous plant cover response to mulching treatments in Colorado coniferous forests: Lessons from a landscape sampling approach.

San Juan Public Lands Dolores Ranger District and Field Office, September 2009, Dolores, CO
Battaglia, M., Rocca, M., Rhoades, C., Ryan, M.G. Mulching treatments in Colorado's coniferous forests.

4th International Fire Ecology and Management Congress: Fire as a Global Process, December 2009, Savannah, GA.
Battaglia, M., Rocca, M., Rhoades, C., Ryan, M.G. Ecological responses to mulching treatments in Colorado coniferous forests: Lessons from landscape sampling.

Posters

International Association of Wildland Fire Conference, September 2008. Jackson Hole, WY.
Battaglia, M., Rhoades, C., Rocca, M., Ryan, M.G. Ecological effects of mastication fuels reduction treatments in Colorado.

94th Annual Ecological Society of America, August 2009, Albuquerque, NM
Rhoades, C., Battaglia, M., Rocca, M., Ryan, M.G. Woody mulch effects on soil climate, chip decomposition, and nitrogen availability in mechanical fuel reduction treatments.

4th International Fire Ecology and Management Congress: Fire as a Global Process, December 2009, Savannah, GA.
Battaglia, M., Rocca, M., Rhoades, C., Ryan, M.G. Surface fuel loadings in mulching treatments in Colorado coniferous forests (JFSP-06-3-2-26).

4th International Fire Ecology and Management Congress: Fire as a Global Process, December 2009, Savannah, GA.
Rhoades, C., Battaglia, M., Rocca, M., Ryan, M.G. Changes in nitrogen availability, soil microclimate, and chip decomposition in mulching treatments in Colorado coniferous forests (JFSP-06-3-2-26).

4th International Fire Ecology and Management Congress: Fire as a Global Process, December 2009, Savannah, GA.
Rocca, M.E., Battaglia, M., Rhoades, C., Ryan, M.G. The effects of mulching treatments on the forest herbaceous layer of Colorado coniferous forests.