Final Report: Piñon and juniper tree mastication effects in the Great Basin and Colorado Plateau

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Abstract

Land managers are implementing fuel reduction treatments to maintain sagebrush communities where pinyon and juniper trees have encroached and are infilling. We determined effects of tree shredding or mechanical mastication on vegetation, fuels, and soils on 44 sites in Utah over a range of pretreatment tree cover percentages. The majority of sites in the Great Basin were encroachment sites (26 of 29), while the majority of sites in the Colorado Plateau were tree climax sites (12 of 15). We also measured effects of shredding and subsequent wildfire on fuels on 5 sites. Total shrub and sagebrush cover decreased with increasing tree cover across untreated, shredded, and shredded and seeded treatments, with shrub cover approaching zero between 70-80% tree cover. Shredding did not decrease shrub cover, but because of sagebrush loss associated with encroachment and infilling, maintenance of sagebrush cover of >10% requires that trees be reduced before tree cover exceeds 20%. Shredding where tree cover is >50% and shrub cover is lost produced a perennial herbaceous-dominated community. Shredding with and without seeding increased tall grass cover on both encroachment and tree climax sites. After shredding or shredding and seeding, perennial herbaceous understory cover increased to >20%, across most sites. When shredding was implemented at mid to high ranges of pretreatment tree cover (15-90%), this increase resulted in perennial herbaceous cover that equaled or exceeded that at early phases of encroachment where tree cover was <10%. Shredding on encroached sites increased cheatgrass cover compared to no treatment at 25-90% pretreatment tree cover. Sites with high cheatgrass cover on untreated plots had high cheatgrass cover after tree shredding or shredding and seeding.

Shredding converted canopy fuels to 1 and 10-hour woody fuels and increased herbaceous fine fuels. Shredded surface 1-hour woody fuels decreased substantially after 5 to 6 years since treatment, and this loss increased with increasing pretreatment tree cover. Although shredding did not decrease live shrub fuel loading, shredding + subsequent burning greatly reduced shrub and woody debris loading, but increased herbaceous fine fuel loading.

Microsites regulated the effects of shredded material on soil nutrient availability. Addition of shredded materials depressed N mineralization in soils at the edge of the tree canopy but enhanced N mineralization in interspace soils. Nutrient and microbial analyses suggest that shredded debris was a recalcitrant C source that was high in lignin and therefore difficult for microbes to decompose. This depressed N microbial mineralization and therefore inorganic N availability at the canopy edge as microbes scoured soils for N. However, shredding increased perennial native grass frequencies in interspace soils, thereby supplying microbes with more labile C substrates to stimulate interspace N mineralization. We suspect this was associated with a longer period of available water in shredded areas which increased grass growth in interspaces and added more labile C to the soil from grass root turnover and litter. Shredded material increased the availability of P with the greatest percent increase occurring in canopy (36%), followed by canopy edge (26%), and finally interspace (17%) soils. In shredded areas, frequency of cheatgrass was associated with lower P availability in surface soils, squirreltail grass was associated with greater N mineralization, and bluebunch wheatgrass was associated with greater P availability.

Shredding to reduce canopy fuels should increase ecosystem resilience by increasing perennial grass cover, which is critical to resisting weed dominance and erosion. Sites that have high cheatgrass cover and low perennial grass cover before shredding should be seeded in conjunction with shredding.

Background and Purpose:

Sagebrush (*Artemisia* L.) steppe offers a multitude of ecosystem services including wildlife habitat, soil stability, forage production, and biodiversity (Bestelmeyer and Briske 2012; Chambers et al. 2013). Encroachment into sagebrush communities and subsequent infilling of pinyon (*Pinus* spp.*)* and juniper (*Juniperus* spp.) trees increase risk of fast-spreading crown fire as woody fuel loads increase (Gruell 1999; Miller et al. 2014; Young et al. 2014a). High intensity fire may cause the system to pass a biotic threshold into an alternative state of weed dominance and recurrent fire (D'Antonio and Vitousek 1992; Miller and Tausch 2001). According to Miller and Tausch (2001), 33% of current pinyon and juniper woodlands are in closed canopy conditions, and such conditions will likely double in the next 40 to 50 years. With an average of 342,700 ha of piñon and juniper woodlands in the Great Basin burned by wildfire annually (Balch et al. 2013), managers are implementing fuel reduction treatments to maintain sagebrush communities (Page et al. 2013). These efforts are supported by concern for critical sagebrush-obligate wildlife species like greater sage grouse (*Centrocercus urophasianus*) (Connelly et al. 2004) and pygmy rabbit (*Brachylagus idahoensis*) (Wilson et al. 2011), as well as an emphasis on vegetation management that increases carbon sequestration by avoiding weed dominance (Prater et al. 2006; Rau et al. 2011, 2012).

Ideally, managers seek to use fuel treatments to increase the ability of the system to regain (ecological resilience) or retain (ecological resistance) fundamental structure and processes in the face of stressors and disturbances, such as fire, grazing, and invasive species (Chambers et al. 2014). Soil water and nutrient resources made available by tree reduction may increase growth and cover of both desirable vegetation and weeds (Roundy et al. 2014a,b). High resilience after treatment is indicated by a return to similar shrub and perennial herbaceous cover as was present before or during early phases of encroachment and infilling. High resistance is indicated by lack of transition to weed dominance after treatment. Perennial grasses are especially important to both biotic and abiotic resilience of these systems. Perennial grasses help avoid the crossing of a biotic threshold by resisting weed invasion and dominance (Chambers et al. 2007, 2014; Roundy et al. 2014a). In addition, they help avoid an abiotic threshold by increasing infiltration and decreasing erosion in interspaces (Pierson et al. 2010, 2013; Williams et al. 2013). Selective mechanical treatments which leave desirable understory species to use resources and resist weed invasion should be expected to best maintain resilience of these systems.

Land managers must decide what fuel-reduction treatments to apply, as well as where and when to apply them. Mechanical tree control is easier to implement than prescribed fire and can be selectively applied (e.g. thinning, clear-cutting, or mosaics) almost any time of year when the soil surface is dry. Although shredding can be expensive compared to prescribed fire (up to $300/acre compared to $175/acre- see SageSTEP 2011), it is obviously much less risky to implement. Unlike cutting, shredding trees with a large, toothed drum (Cline et al. 2010) converts canopy fuels to small 1 and 10-hour fuels, which can greatly reduce fire spread and facilitate containment (Young et al. 2014a). Debris from shredding also increases infiltration and reduces sediment production on interspace microsites (Cline et al. 2010). Along with tree reduction, shredded debris increases soil temperature and available water for seedling establishment (Young et al. 2013a,b), and may increase nutrient availability (Bates et al. 2002; Young et al. 2014b).

In this study, we determined effects of tree shredding in relation to where (ecological site type), and when (extent of tree encroachment and infilling at time of treatment) for 44 sites in Utah. Ecological site types were either sagebrush steppe sites encroached by trees (wooded shrublands), or tree climax sites (persistent woodlands; Romme et al. 2009). Treatments were either shredding (mastication) or shredding and seeding. We determined effects of treatments on vegetation response, fuels, and soil nutrients and microbes. We also determined the effects of time-since-treatment and post-treatment fire on fuels.

Study Description and Location

Study sites were located across Utah in the Great Basin or Colorado Plateau physiographic provinces (Figure 1). The majority of sites in the Great Basin were encroachment sites (26 of 29), while the majority of sites in the Colorado Plateau were tree sites (12 of 15). Encroachment sites, "Rangeland Ecological Sites" (NRCS 1997), or wooded shrublands (Romme et al. 2009) typically have trees <150 years old and are associated with deeper, less rocky soils. Tree climax sites, "Forest Land Ecological Sites" (NRCS 1997), or persistent woodlands (Romme et al. 2009) typically have shallow (<0.5-m deep), rocky soils, trees >150 years old, and support infrequent fire. Each site had an untreated control area and either a shredded only or a shredded-seeded treatment. Seed was aerial broadcast prior to treatment according to specifications of the individual agency.

We made field visits and checked soil surveys to select untreated and treated sample plots or areas on the same ecological site types on each of the 44 study sites. Within these areas at each study site, we randomly selected 30 by 33-m potential subplots for sampling to represent a range of untreated or an apparent range of pretreatment tree cover (TC). We then used object-based image analysis software (Feature Extraction ENVI 4.5) and pretreatment NAIP imagery (1-m pixel resolution) to determine untreated and pretreatment TC (Hulet et al. 2014) on the potential subplots. We randomly selected three subplots each on untreated and treated areas from the potential subplot population for each of three TC categories: low (<15%), intermediate (15-45%), and high (>45%). Not all study sites had all TC categories, so the number of subplots ranged from a minimum of six (1 TC category x 2 TRT-untreated and shredded x 3 subplots = 6) to 18 (3 TC categories x 2 TRT x 3 subplots = 18). The only exception to this sampling scheme was for three sites originally treated and measured in a previous study known as SageSTEP (Sagebrush Steppe Treatment and Evaluation Project, McIver et al. 2010). On those sites, 15 subplots were measured on untreated and shredded areas across a range of initial tree covers. Because sites were either shredded and left unseeded or shredded and seeded, these treatments occurred on different sites. We also measured subplots on a few sites that were shredded and then subsequently burned in wildfires. Effects of time-since-treatment on fuels were determined on three SageSTEP sites because those sites had pretreatment fuel measurements.

Vegetation and fuels were measured and soil samples collected in the summers of 2011 and 2012. Vegetation cover, density, and biomass were measured on each 30 by 33-m subplot according to the protocol of McIver et al. (2010) and Miller et al. (2014). Fuels were measured according to the protocols of Tausch (2009), McIver et al. (2010) and Young et al. (2014a). For treated areas, woody debris loading of 1-, 10-, and 100-hr fuels was quantified by collecting fuels within 25 x 25-cm quadrats to be oven-dried, separated by species and fuel size class, then weighed. Quadrats were placed each 3rd meter on 5, 30 m transects (50 total quadrats) for subplots with tree cover <15%, and on 3, 30 m transects (30 total quadrats) for subplots with >15% tree cover. At each site, soil samples were taken in untreated and treated plots. For each subplot, three soil samples were collected relative to one tree per subplot: 1) at 1/3rd of the distance from the tree bole to the canopy, 2) at the canopy edge, and 3) in the interspace between trees. Soils were collected with a 5-cm diameter by 10-cm long soil corer. Samples were collected at 0-2 and 15-17 cm; samples were bulked by depth across 3 subplots for each treatment and tree canopy cover class. Nitrogen transformation rates and phosphorus availability were measured, as well as soil respiration and microbial biomass. Nitrogen mineralization and phosphorus availability were evaluated in soil microcosms incubated for ten days at constant gravimetric water content (0.3 g H_2O g dry soil^{-1}).

We used mixed model analysis of covariance (Proc Glimmix, SAS v9.3, SAS Institute, Inc., Cary, NC) to compare vegetation responses of functional groups. These cover groups included total shrubs, sagebrush, tall, short, and total perennial grass (tall, short, and rhizomatous grasses), cheatgrass (*Bromus tectorum* L., considered separately due to concern for its dominance), perennial forbs, sage grouse food (Connelly et al. 2004; Nelle et al. 2000; Pyle and Crawford 1996; Rhodes et al. 2010), annual forb, total perennial herbaceous (total perennial grass plus perennial forb), lichen, biotic crust, and bare ground. Sandberg bluegrass (*Poa secunda* J. Presl) was considered the only short grass, while all other bunchgrasses were considered tall grasses. Analysis included ecological site type (ES) and treatment (TRT) as fixed factors. Untreated and pretreatment TC, estimated from NAIP imagery was analyzed as a covariate (Roundy et al. 2014a). Site was considered random and subplots (557 total across 44 sites) were treated as subsamples within TRT and ES. The Tukey-Kramer test was used to determine significant differences among ecological site type and treatment combinations for each 5% increment of untreated and pretreatment TC (Roundy et al. 2014a). We adjusted for false positives from multiple comparisons by using a $P < 0.01$. Vegetation response cover data were normalized by the arcsin square root transformation, while density data were normalized by the square root transformation. Tree cover covariate data were not transformed. Observation of residual plots indicated that assumptions were met for analysis of covariance.
Fuels were similarly analyzed using mixed model analysis for all sites, and using mixed model repeated measures analysis for SageSTEP sites only. Fuels data were square-root transformed. The SageSTEP sites were the only sites that had fuel measurements over time (1-6 years since treatment for herbaceous fuels and 1, 5 or 6 years since treatment for coarse woody debris). Soil data were analyzed by ANOVA and ecological causal network analysis (CNA), which is a systems-oriented methodology to analyze causation and interactions among ecosystem characteristics. To test the impacts of shredding on soil characteristics and mediating effects of these characteristics on the frequency of four grass species, we modeled plant responses as a function of time since treatment and five soil variables (N mineralization, C mineralization, P availability, dissolved organic C, and microbial biomass) each measured to two depths (0-2 cm, 15-17 cm) across the three microsites. These models included site location as a random factor. Comparing the coefficient estimates of this model with those that do not include the soil factors allowed us to estimate the degree to which changes in the plant responses can be attributed to changes in the soil variables caused by the shredding.

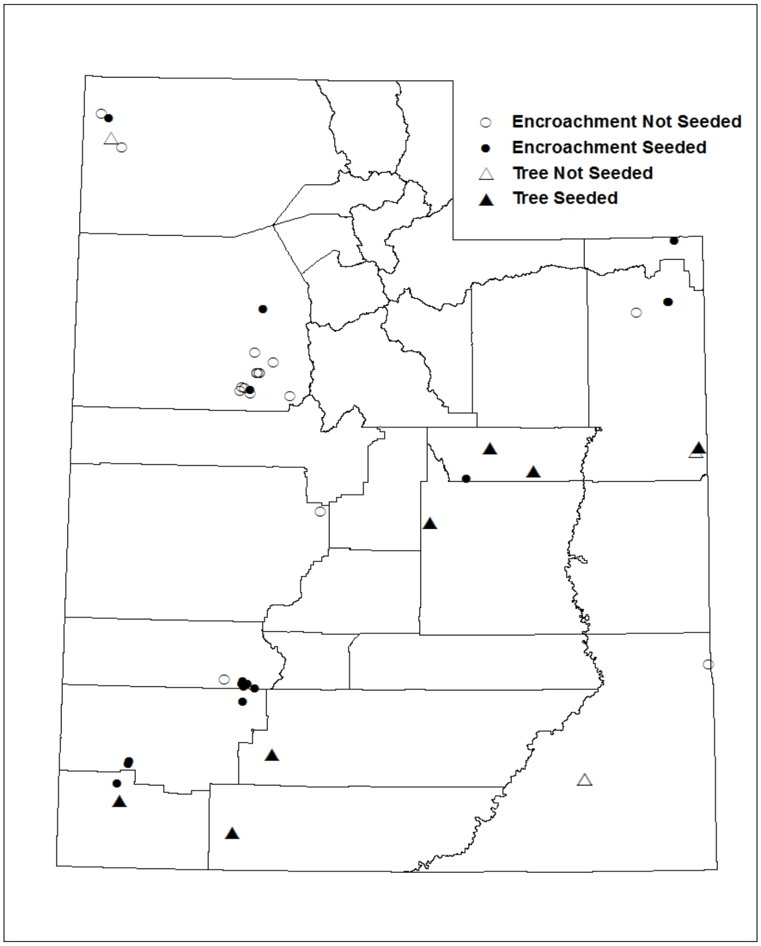

Figure 1. Location of study sites.

Key Findings

Vegetation response: Encroachment sites had 0.3% higher sagebrush cover, 5% higher cover of Sandberg bluegrass, 4.5% higher cover of tall perennial grass, and greater cheatgrass density (45 vs 13 plants m^{-2}) than tree sites. ***Total shrub and sagebrush cover decreased with increasing TC across all treatments, with shrub cover approaching zero between 70-80% TC (Figure 2).*** Sagebrush cover was 67% of total shrub cover across all sites and treatments. Encroachment sites had higher total shrub cover (23-30%) and sagebrush cover (18-23%) at low TC than tree sites (total shrub cover= 20% and sagebrush cover= 11-18%).

However, shrub and sagebrush cover persisted until TC reached 80% on tree sites and only until it reached 70% on encroachment sites. ***Shredding did not decrease shrub cover, but because of sagebrush loss associated with infilling, maintenance of sagebrush cover of >10% requires that trees be reduced before tree cover exceeds 20% (Figure 2).*** We observed sagebrush seedlings on 61% of the 44 study sites. For these sites, the number of seedlings m^{-2} was 0.073 on untreated plots, 0.43 on shredded not- seeded plots, and 0.65 on shredded-seeded plots.

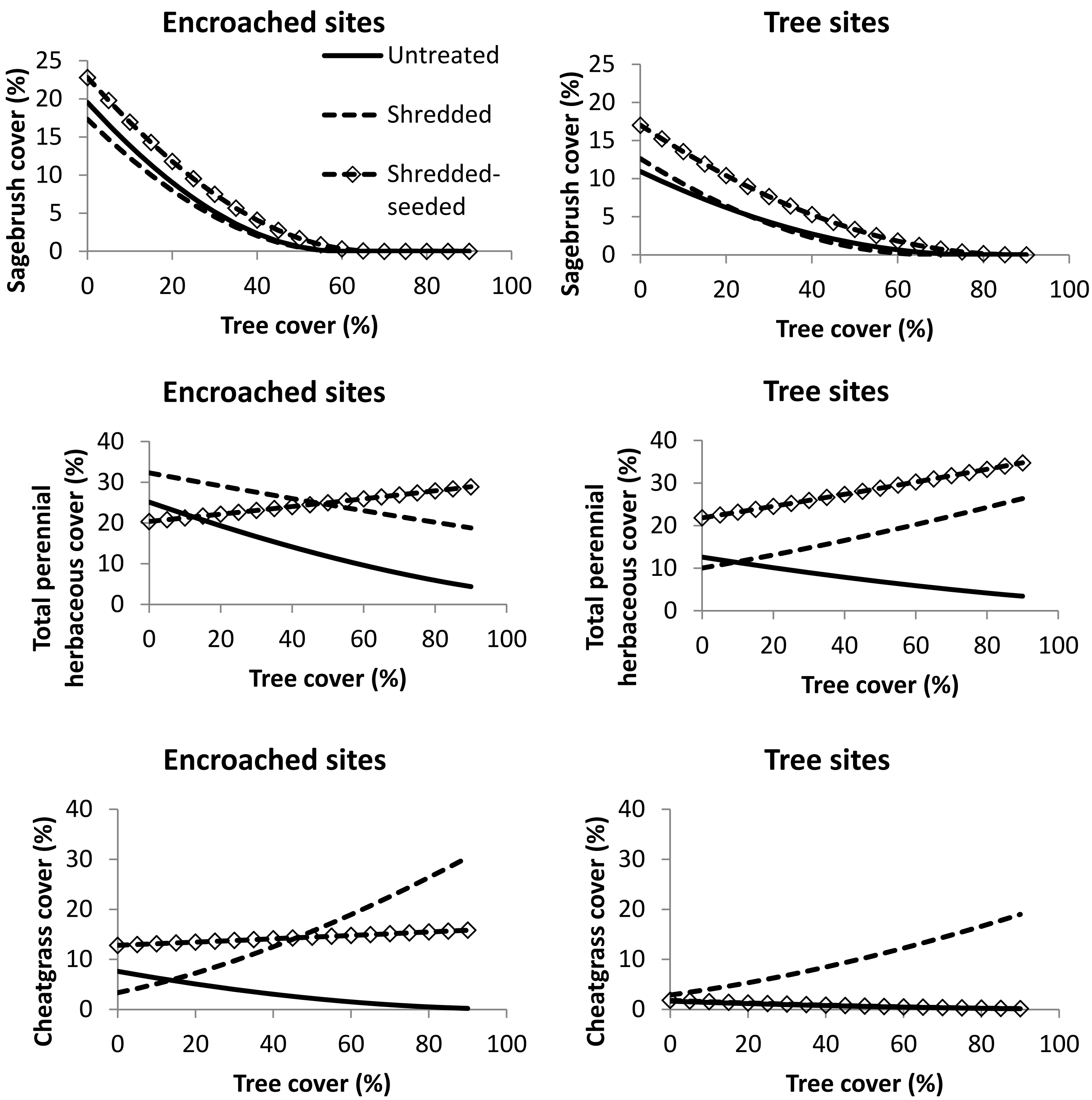

Figure 2. Vegetation cover in relation to pretreatment tree cover on sagebrush steppe sites encroached by pinyon and juniper trees and on tree climax sites.

Shredding where tree cover is >50% and shrub cover is lost produced a perennial herbaceous-dominated community (Figure 2). Across all sites and treatments, the percentage of total perennial herbaceous cover was 47 for tall grass, 34 for short grass, and 87 for total perennial grass. Perennial and sage grouse forb cover were low for all treatments (<6.2 and <8%) and not related to TC. Shredding alone did not increase perennial forb cover, but seeding after shredding increased perennial forb cover by 2.4% compared to no treatment across both ecological site types and across the range of TC. Sage grouse forb cover was low (<2%) on tree sites and did not vary by treatment. On encroachment sites, there was a trend for increasing sage grouse forb cover on shredded and seeded plots compared to untreated or shredded only plots as pretreatment tree cover increased, although differences were not statistically significant. Treatments did not significantly affect short grass cover, which decreased slightly with increasing TC. ***Shredding with and without seeding increased tall grass cover on both encroachment and tree sites.*** On encroachment sites, shredding increased tall and total perennial grass cover across the range of TC. While shredding increased tall and total perennial grass cover most at high TC, perennial grass cover still decreased with increasing TC. In contrast, on tree sites, shredding had little effect at low initial TC, but increased tall and total perennial grass cover with increasing pretreatment TC. Shredding and seeding showed a similar pattern by increasing tall and perennial grass cover as pretreatment TC increased for both encroachment and tree site types. However, shredding and seeding increased tall and total perennial grass cover most on tree sites and shredding and seeding was most effective at high pretreatment TC. ***After shredding or shredding and seeding, perennial herbaceous understory cover increased (generally to >20%) to equal or exceed that at early phases of infilling (<10% tree cover), at mid (15-35%) to high (90%) ranges of pretreatment tree cover (Figure 2).***

Cheatgrass also increased in cover after tree shredding or shredding and seeding. Shredding increased cheatgrass cover with increasing pretreatment TC for both ecological site types. ***For encroachment sites, shredding without seeding increased cheatgrass cover compared to no treatment at 35-90% pretreatment TC while shredding and seeding increased cheatgrass cover at 25-90% pretreatment TC.*** However, this difference may have been because managers seeded sites that had more cheatgrass potential. Cheatgrass cover varied widely across the study sites. A few sites had >18% cheatgrass cover (6 of 44 sites for untreated plots; 9 of 44 sites for shredded or shredded-seeded plots). Sites with > 35% perennial herbaceous cover had <10% cheatgrass cover. ***Sites with high cheatgrass cover on untreated plots had high cheatgrass cover after tree shredding or shredding and seeding.***

<u>Fuel response:</u> As expected, ***shredding converted large canopy fuels to 1 and 10-hour fuels (Figure 3).*** As untreated or pretreatment tree cover increased, so did untreated standing tree 1 hour and 100+1000-hour fuels. After shredding, 1 and 10-hour fuels especially increased with increasing pretreatment tree cover (Figure 4). Although herbaceous fuels increased with tree shredding and with higher pretreatment tree cover, the majority of surface fuels were woody debris from tree shredding. Shredding places canopy fuels on the ground, which can facilitate wildfire containment. ***Surface 1-hour fuels decreased substantially after 5 to 6 years since treatment, and this loss increased with pretreatment tree cover (Figure 4). Although shredding did not decrease live shrub fuel loading, shredding + subsequent burning greatly reduced shrub and woody debris loading, but increased herbaceous fine fuel loading.*** Models have not been developed or tested to appropriately model fire behavior with shredded fuels (Battaglia et al. 2010). However, this project reports detailed fuel data that can be used as models are developed (Shakespear 2014).

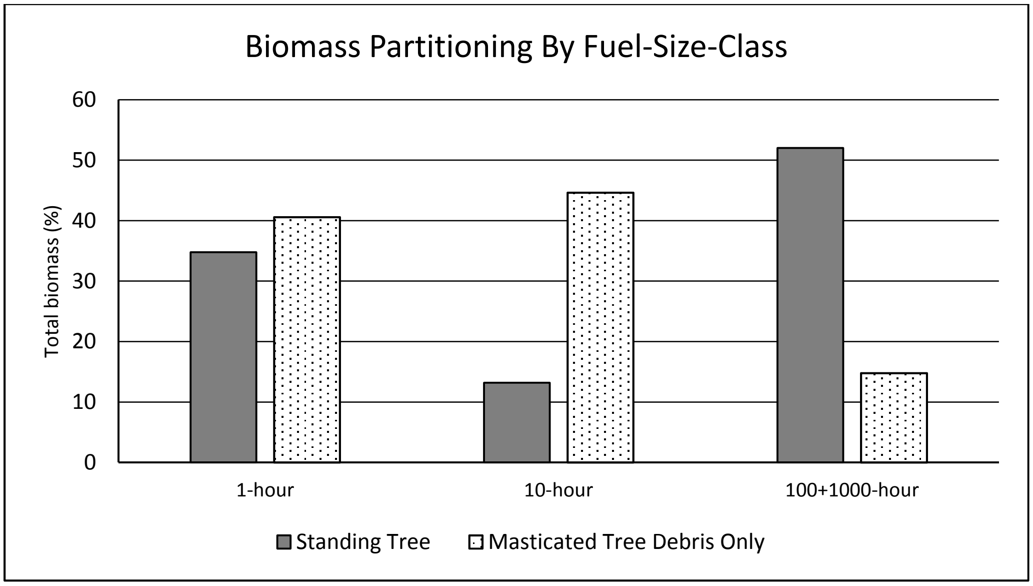

Figure 3. Biomass partitioning by fuel size class for untreated standing trees and shredded or masticated trees. Masticated tree debris represents only debris additions to the pretreatment fuelbed as a result of mastication and does not include pretreatment woody debris.

<u>Soil response:</u> ***We found that microsites regulated the effects of shredded material with the addition of shredded materials depressing N mineralization in soils at the edge of canopy but enhancing N mineralization in interspace soils (Figure 5).*** N mineralization rates describe the amount of available inorganic N produced in soils for colonizing plant species (Rigby 2013). We found that N mineralization decreased with depth, was not influenced by tree species, (*J. osteosperma* versus *P. edulis*), and decreased with increasing tree density. Further, shredding additions also created differential responses based on microsite. The addition of coarse woody debris decreased N mineralization and nitrification rates in tree-island edge soils but increased N mineralization in interspace soils (Figure 4). We found no effects of shredding on N mineralization in deeper soils. ***Our results suggest that inorganic N availability was directly depressed by shredded material as microbes potentially coped with the addition of more woody recalcitrant C sources by scouring soils for N, but also indirectly enhanced as higher grass frequencies in interspace soils supplied microbes with more labile C substrates stimulating N mineralization.***

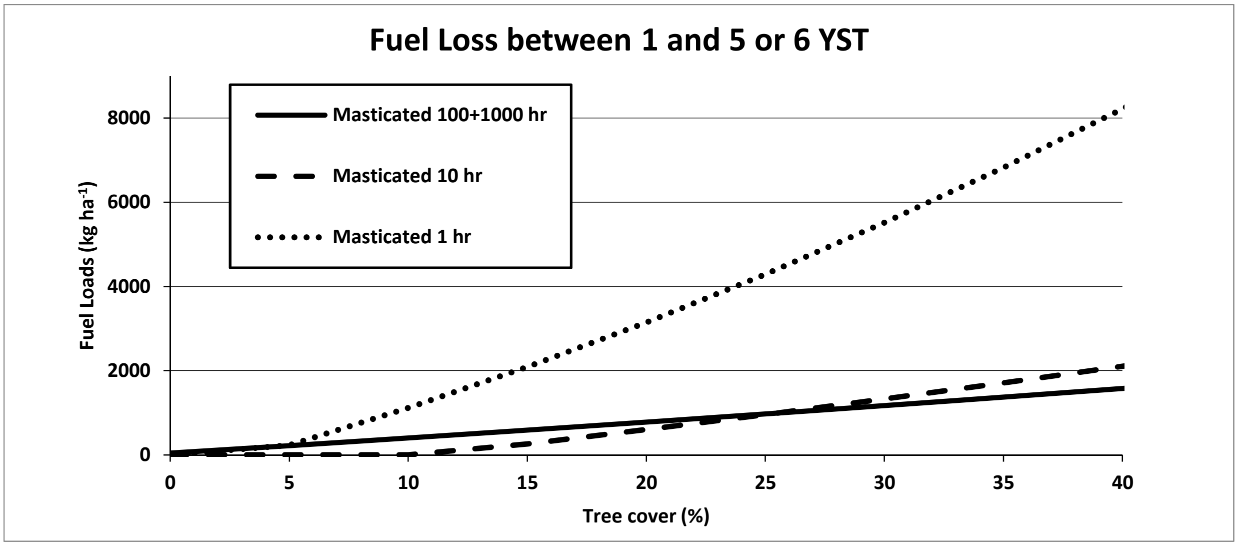

Figure 4. Woody debris fuel loss 5 or 6 years after tree shredding or mastication in relation to pretreatment tree cover.

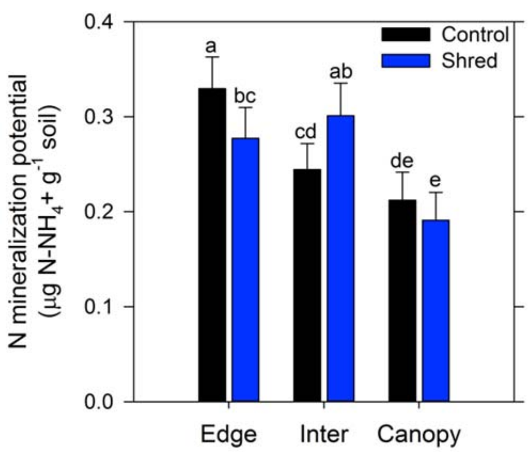

Figure 5. Effects of shredding addition on N mineralization in surface (0-2 cm deep) soils within the tree canopy mound, edge, and interspace between trees. Values are means (n=40) ± one standard error with letters indicating differences (P< 0.05).

We also found that shredded material increased the availability of P with the greatest percent increase occurring in canopy (36%), followed by canopy edge (26%), and finally interspace (17%) soils (Figure 6). We found that P availability decreased with depth, was more available under *P. edulis* than *J. osteosperma*, increased with increasing tree density, was highest in soils underneath tree canopies, and increased following shredding.

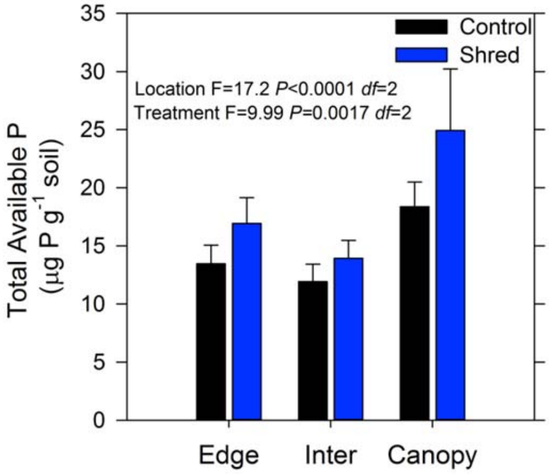

Figure 6. Effects of shredding addition on P availability in surface soils (0-2 cm depth) for tree canopy mound, edge, and interspace between trees. Values are means (n=40) ± one standard error.

Nitrogen and phosphorus availability influenced grass species. In woodlands with more intermediate cover of *P. edulis* than *J. osteosperma*, the frequency of perennial native grasses, especially *Elymus elymoides* (ELEL) and *Psuedoroegneria spicata* (PSSP), was at least 70% higher following the addition of shredded materials (***Figure 7***). Also, the frequency of perennial exotic grasses and *Bromus tectorum* (BRTE) was higher under shredded additions in interspace and edge soils. We found no evidence for changes in *Poa secunda* (POSE) frequency following shredding.

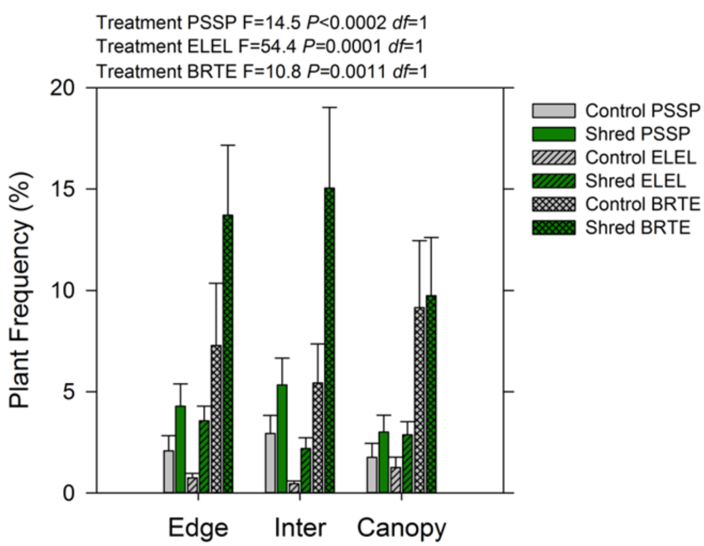

Figure 7. Effects of shredding on frequency of grasses relative to tree mid-canopy, edge, and interspace microsites.

Based on our CNA modeling efforts linking these plant responses to shredding-induced changes in soil characteristics, we found that N mineralization in deeper soils and P availability in surface soils influenced several grass species (***Figure 8***). The model used differencing data (shredded minus control treatment) by site of plant and soil responses from all three microsites. From these differences (Δ), **we found that as N mineralization increased in deeper soils (15-17 cm) the frequency of PSSP (coefficient = 1.37) and BRTE (coefficient = 1.54) also increased. These coefficients specify that for every 1% change in N mineralization the frequency of PSSP and BRTE increased from 1.37 and 1.54% respectively, suggesting that both natives and exotics are stimulated by soil containing higher N mineralization rates.** This model accounted for soil environmental differences across sites and among microsites. The relatively large increases that we measured in N mineralization in shallow soils following shredding did not influence any plant species and may not be as important as these deeper reservoirs of N that are potentially accessed by plants as they establish. As for P, **shredding-induced increases of P in shallow soils only negatively impacted POSE (coefficient = -0.54), demonstrating that as P availability increased across sites the frequency of this native plant declined.** We found no direct evidence linking the increase in P occurring in shallow soils to shifts in plant frequency after shredding. Other soil variables besides nutrients did vary following shredding and influenced changes in plant frequency such as C mineralization in deep soils (CMin_deep), microbial biomass (Micro) in shallow and deep soils, and all of these soil characteristics influenced PSSP only. This perennial grass species seemed to be the most sensitive to changes in microbial mediated processes centering on C dynamics. Last, years since shredding also influenced the frequency of all species, with POSE (coefficient = 0.18) and PSSP (coefficient = 0.18) increasing, and BRTE (coefficient = -0.13) and ELEL (coefficient = -0.15) decreasing as time progressed.

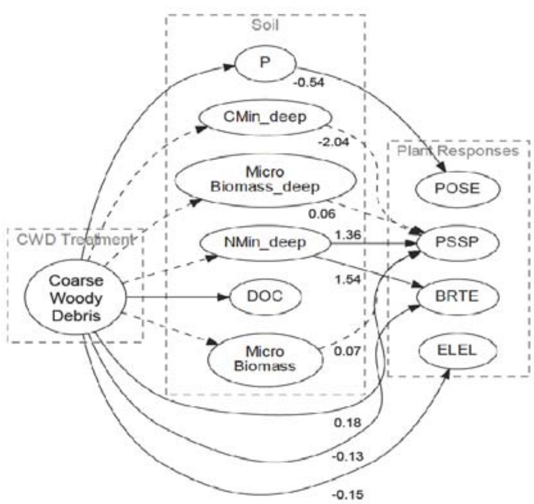

Figure 8. Casual network model identifying the impact of shredding (coarse woody debris) on soil characteristics and grass species frequency. All solid arrows indicate significant relationships ($P < 0.05$) and dashed arrows indicate marginally significant relationships ($P = 0.06 – 0.08$) between shredding and soils characteristics, soil characteristics and plant frequencies, or shredding and plant frequencies. Values are coefficients for the model describing the direction and magnitude of relationship between soil variables and plant frequencies.

Management Implications

Shredding pinyon and juniper trees changes canopy fuels to surface fuels which should allow better control and suppression of wildfires. The smallest surface woody fuels (1-hour fuels) decreased substantially 5 to 6 years after shredding. Shredding trees at low tree cover maintained a shrub/grass community while shredding at intermediate to higher tree cover resulted in a herbaceous-dominated plant community. Perennial grasses especially increased after shredding, and did so at high pretreatment tree cover. Seeding after shredding increased the perennial herbaceous response, especially on tree climax sites. Sites that are dominated by cheatgrass before shredding have high probability of cheatgrass dominance after shredding. Where residual or seeded perennial herbaceous species amounted to > 35% cover, cheatgrass cover was generally limited to < 10%. Seeding should accompany shredding where perennial herbaceous cover is limited by infilling and pretreatment cheatgrass cover is a concern. Tree reduction and shredding in particular increased water and nutrient availability. Increases in N mineralization and the resulting increase in N availability may stimulate growth and establishment of desirable perennial grasses, as well as cheatgrass. Additional analysis will attempt to better relate site environmental characteristics that associate with desirable or undesirable vegetation dominance.

Relationship to related work

Vegetation response: The ideal goal for fuel reduction treatments is to reduce the probability of severe wildfire and improve potential suppression, while supporting ecosystem resilience, wildlife habitat, and forage values. Perennial herbaceous cover is key to sagebrush steppe resilience by helping to avoid biotic and abiotic thresholds through resisting weed dominance and interspace erosion (Roundy et al. 2014a). Shrub (especially sagebrush) cover is key to supporting wildlife habitat. Therefore effects of fuel treatments and the timing of their implementation should be judged relative to sagebrush and perennial herbaceous versus weed cover. In our study, shredding trees maintained shrub cover and increased perennial herbaceous cover. These results are similar to effects of tree reduction by cutting (Bates et al. 2007; Miller et al. 2014; Roundy et al. 2014a). Perennial herbaceous cover recovers a few years after fire or chaining (Tausch and Tueller 1977; Bates et al. 2000; Miller et al. 2014; Roundy et al. 2014a). Shrub cover may take 5 or more years to recover after chaining (Tausch and Tueller 1977). Mountain big sagebrush recovers 15 to 100 years after fire, depending on post-fire cool season precipitation (Nelsen et al. 2013), while Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis* Beetle & Young) recovery is limited (Miller et al. 2013). Chambers et al. (2014b) found higher shrub cover 3 to 4 years after prescribed fire and tree cut and drop treatments on Wyoming and mountain big sagebrush sites encroached by trees. They attributed this increase to sagebrush recruitment and growth of resprouting shrubs (Miller et al. 2014). Mechanical treatments and especially fire can increase weed cover on warmer sites (Bates et al. 2013; Chambers et al. 2014; Miller et al. 2014; Roundy et al. 2014a,b). Although increasing perennial cover may reduce weed cover over time after treatment (Bates et al. 2007), seeding is usually recommended where pretreatment tree cover is high and perennial herbaceous cover is low (Chambers et al. 2014; Roundy et al. 2014a). Our results support that recommendation. Although we did not find statistically significant differences between shredding only and shredding plus seeding, at the highest pretreatment tree covers, seeding produced numerically higher estimates of perennial herbaceous cover and lower estimates of cheatgrass cover than shredding without seeding (Figure 2).

It is generally expected that tree reduction by fire or mechanical methods at higher tree cover could result in a weed-dominated community because the seedbank and residual shrub and perennial herbaceous plants are limited (Miller et al. 2005; Bates et al. 2013). Our study and that of Roundy et al. (2014a) show that mechanical tree reduction should be implemented at low to mid phases of tree infilling to best retain shrubs. Perennial herbaceous and cheatgrass cover increased most after fire and where mechanical tree reduction treatments were applied at high pretreatment tree dominance (Roundy et al. 2014a). These increases were associated with increased growth of residual perennial plants (Miller et al. 2014) and weeds on some sites (Roundy et al. 2014a) as a result of a substantially longer period of springtime soil water availability (Young et al. 2013; Roundy et al. 2014b), as well as increased available nitrogen (Young et al. 2014b). Increased resource availability after shredding supports establishment of both perennial grass and cheatgrass seedlings (Young et al. 2013a,b). We also found in the current study that after shredding, increased N availability favors perennial grass and cheatgrass while increased P availability favors native perennial grasses. In the Roundy et al. (2014a) study, tree reduction at high phases of infilling did not result in higher perennial herbaceous cover than occurred at lower phases of infilling. In the current study, higher perennial herbaceous cover than occurred in untreated areas at low tree cover was obtained by shredding and seeding encroached sites or by shredding alone or shredding and seeding tree sites at high pretreatment tree cover (Figure 3). In contrast, Bates et al. (2013) found limited perennial herbaceous recovery and high cheatgrass cover after burning Phase III (high infilling) western juniper woodlands. These results may have been partly due to increased fire severity from cutting some of the trees prior to burning.

Fuel response: No studies were found for comparison that delivered fuel loading estimates across a tree cover gradient and between treatments for the pinyon-juniper forest type. Our observed mean woody debris loading was lower than that reported for pinyon-juniper by Battaglia et al. (2010) for untreated (61% lower), and for masticated (55% lower). Because Battaglia et al. (2010) did not report pretreatment tree cover values, we are unable to account for this compared loading difference. One of the most apparent results of mechanical mastication is the conversion of large standing-tree fuels (100 and 1,000-hr) into primarily 10-hr surface fuels (Knapp et al. 2011). Fire in heavy fuel loads may imperil site stability due to lethal soil heating, and residual plant mortality, especially with masticated fuel depths of ≥7.5 cm (Busse et al. 2005). Conversely, increased soil moisture as a result of mastication could protect soil from excessive heating (Busse et al. 2005). Soil heating is highly dependent upon the masticated fuelbed depth and soil water content at the time of the fire (Busse et al. 2010, 2005). Busse et al. (2010) found that soil moisture has a strong influence on heat transfer, and volumetric soil moisture content ≥20% suppressed lethal soil heating (>60°C) in a variety of soil types below 2.5 cm depth, while drier soils exceeded 60°C at a depth of 10 cm. Our study found that masticated + burned treatments effectively returned woody surface fuels to pretreatment loading conditions. Prescribed burning could be used outside of the growing season in cool-weather, high-soil moisture conditions to safely remove surface fuels, mitigating the potential for lethal soil heating and plant mortality (Bates and Svejcar 2009; Bradley et al. 2006; Harrod et al. 2008). Further, treating at lower tree cover values will mitigate heavy woody fuel loading.

Loss of woody biomass increased much more over time for 1-hour fuels, compared to larger fuel size classes. Although we did not directly measure decomposition rates, there are several possible explanations for this rapid loss in woody loading over time. Our analysis of standing-tree biomass coupled foliar biomass (scales and needles) with the woody 1-hour fuel size class. In dry coniferous systems, foliar litter decomposition rates generally increase in response to cutting trees (Bates et al. 2007). Killing trees increases soil and litter water availability (Bates et al. 2000; Roundy et al. 2014b; Young et al. 2013b) and may also increase decomposition (De Santo et al. 1993). Bates et al. (2007) found that loss of litter biomass was

60% greater where trees were cut compared to untreated areas. They attributed this loss to increased decomposition associated with addition of canopy foliar litter to the fuelbed, decreasing the C:N ratio. Conversely, Gallo et al. (2006) suggests that in arid systems, decomposition is not strongly correlated with the C:N ratio, but is a function of solar radiation and temperature. Austin and Vivianco (2006) found that litter biomass was reduced 40% during an 18 month period in full sun, and attenuation of solar radiation caused a 60% reduction in litter decomposition. Further, several studies have found photodegradation to be very influential in organic matter decomposition in semi-arid and arid environments (Austin and Vivianco 2006; Gallo et al. 2006, 2009; Vanderbilt et al. 2007). Thus, decomposition of the litter portion of 1-hour fuels certainly contributed to the rapid loss in total loading.

Soil response: Coarse woody debris (CWD) influences microorganisms and ecosystem processes by reducing wind and water erosion thereby stabilizing soil surfaces (Laiho and Prescott, 2004); capturing and governing the release of nutrients and water essential for microbial metabolic activity and higher plant establishment (Miller and Seastedt 2009); and by determining the storage and release of carbon (C) as soil organic matter or CO_2 to the atmosphere (Spears and Lajtha 2004). In semi-desert ecosystems, "shrub or tree islands of fertility" in contrast to barren plant interspaces (Charley and West 1975), are chiefly responsible for differences in microbial processes. Soils beneath trees are enriched with C and other essential elements (e.g., N, P and Ca) due to litter and root inputs and root translocation of elements from interspace soils (Schlesinger et al. 1996; Schlesinger and Pilmanis 1998). Continued encroachment and infilling can increase root and soil C and N (Rau et al. 2011b). Microbes in tree island soils are often C-limited due to the low quality of pine needle litter. For example, respiration per unit microbial biomass is lower in soils beneath trees than in adjacent shrub/grassland soils indicating that microbes are less efficient at C mineralization (Liao and Boutton 2008). Besides metabolic activity, microbial N transformations (e.g., N mineralization and nitrification) are higher beneath tree islands than interspace soils (Schade and Hobbie 2005). However, the impact of CWD on microbial activity in semi-deserts is relatively unknown and the addition of CWD has the potential to alter soil respiration dynamics and N transformations (Rigby 2013). In forest and desert systems, CWD increases soil moisture, thereby affecting respiration and decomposition rates (Fierer et al. 2003; Lipson and Schmidt 2004; Crawford et al. 2005; Göttlicher et al. 2006; Monson et al. 2006; Zobitz et al. 2008), net N mineralization (Evans et al. 1998; Perez et al. 2004), and P (Young 2012). Dead root litter has the potential to influence C and N mineralization (Rau et al. 2011b) even though this C source is more diffuse and spread throughout the soil profile compared to concentrated piles of CWD on the soil surface.

We found that the effects of CWD on nutrient availability were regulated by microsite. Specifically, shredded materials depressed N mineralization in soils at the edge of canopy but enhanced N mineralization in interspace soils. These changes in N mineralization and ultimately the availability of inorganic N for plant species were likely caused by differences in litter quality. For example, shredded materials have a high C:N ratio and microorganisms decomposing these more recalcitrant C substrate may retain what N is contained in these C substrates for metabolic processes and growth. Alternatively, in interspace soils, the increase in grass frequency and, thus, higher inputs of root and aboveground litter may have allowed microorganisms to overcome any potential N-limitations and release N as they decomposed these more labile C substrates with lower C:N ratios. Basically, we are suggesting that microorganisms in interspaces did not have to contend with CWD additions and benefited from the increased frequency of grasses. As for P, the addition of CWD generally increased P availability, with the biggest increase occurring in canopy soils that already contained the highest levels of P prior to shredding. Thus, P accumulation was already occurring in soils beneath the canopy as trees actively concentrated P in duff layers and shredding only accentuated this accumulation by either stimulating the decomposition of existing duff or shredded materials. This is consistent with the tree-island of fertility effected noted by others. For example, there was a steep gradient of nutrient dynamics under *Juniperus occidentalis* canopy, with the highest concentrations of P, K, and Ca occurring closest to the bole of the tree and decreasing concentrations as distance from the bole increased Miwa and Reuter (2010).

N mineralization in deeper soils and P availability in surface soils influenced the frequency of native *P. spicata* and exotic cheatgrass. After we accounted for soil variability occurring in microsites and sites, our CNA modeling efforts linked six of ten soil characteristics to shredding and five of these six influenced plant frequency. Native perennial grasses may be stimulated by mechanical tree reduction and the addition of shredded material (Bates et al. 2007; Ross et al. 2012; Bybee 2013; Bybee et al. 2014; Miller et al. 2014, Roundy et al. 2014a). Also, cheatgrass may decrease with the addition of CWD (Wicks 1997; Wolk and Rocca 2009). Our modeling results support the links between perennial responses to shredding but also demonstrate that the susceptibility of sites to invasion is, in part, mediated through soil characteristics. Our current study is more robust and unusual in that it also relates species plant frequency to specific soil characteristics. We found that as N mineralization increased in deeper soils the frequency of *P. spicata* and exotic cheatgrass also increased, suggesting these species benefited from relative deeper reservoirs of N. Thus, if sites contain higher N mineralization rates at rooting depth they are more susceptible to invasion. However, the potential for invasion decreases as time progresses since shredding, demonstrating that if sites aren't invaded in the first couple of years they may be less likely to be invaded in the first decade at least. In contrast, P availability had little impact on plant frequency.

Future work needed

Now that regional short term vegetation responses to tree reduction by fire and mechanical methods have been reported (Bybee et al. 2014; Miller et al. 2014; Roundy et al. 2014a), there are two critical needs relative to vegetation response: 1) determine environmental factors associated with differential responses among sites, and 2) determine long term (10+ years) responses. Although many sites move toward perennial herbaceous dominance after tree reduction, drier and warmer sites may move toward weed dominance (Chambers et al. 2014). One of the most frequent questions land managers have about tree-reduction treatments is whether or not the treatments will result in cheatgrass or other weed dominance on their sites. Additional analysis of our current data, as well as analysis of longer term responses is needed to determine site characteristics associated with weed compared to perennial herbaceous plant dominance. Because shredding pinyon and juniper trees is a relatively new practice, our oldest treatments were implemented only 8 years prior to measurement. Important responses that should be measured in the long term include perennial herbaceous versus annual weed dominance, shrub recover (especially when treatments were implemented at high tree cover), and tree recovery. Trees may recover more quickly after mechanical treatments than after fire.

Resources made available by tree reduction may be used by tree seedlings (Chambers et al. 1999), small trees that were not treated, or trees sprouting from incomplete mechanical treatment. Infilling after chaining steadily reduced growth of understory cover and production starting at 5 to 8 yr after treatment (Tausch and Tueller 1977). Studies of mechanical treatments suggest that western juniper will return to dominance within 50 years (Bates et al. 2005, 2006; O'Connor et al. 2013).

Relative to fuels, future work should monitor surface fuel loss over longer time periods (10+ years). In addition, models need to be developed that accurately model fire behavior for coarse woody surface fuels (Battaglia et al. 2010). Measurements of fuel and vegetation response to spot burning of shredded fuels could also be useful, since managers are currently implementing those kinds of treatments.

Our study suggested some interesting associations of native perennial grasses and annual cheatgrass relative to microbial activity, soil nutrients, tree fertile islands, and CWD from shredding. Longer term effects of these relationships relative to former tree canopy, edge, and interspace microsites may suggest ways to promote perennials and disfavor cheatgrass. Since native perennial grasses are critical to avoiding the crossing of biotic and abiotic thresholds in these systems, a better understanding of these ecological relationships could aid in better management to enhance resilience (Briske et al. 2006).

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Deliverables and status.

Deliverable type	Description	Status
Initial Fact Sheet posted on SageSTEP website	Project introduction, goals, approach	SageSTEP Newsletter Issue 16, Fall 2011
Updates posted on SageSTEP website	Current progress and findings	SageSTEP Newsletter Issue 22, Fall 2013
Final report	Summary report on procedures and findings	June 2014 (because of 1-year no-cost extension)
Presentations	Procedures and results on fuel, vegetation, and soil responses	Young, Kert and Bruce Roundy. 2011. Mechanically shredding Utah juniper. Abstract 230. Society for Range Management 64th annual meeting, February 6-10, 2011, Billings, Montana. Young, Kert and Bruce Roundy. 2011. Mechanically shredding Utah juniper and soil characteristics. 15-19 Oct 2011. Soil, Crop, and Environmental Sciences international annual conference. San Antonio, TX. Young, Kert and Bruce Roundy. 2011. Mechanically shredding Utah juniper and soil characteristics. 14-17 Nov 2011. Interior West Fire Ecology conference. Salt Lake City, UT. Bybee, J., B. Roundy, K. Young, A. Hulet, and D. Roundy. 2012. Mechanical shredding as a fire surrogate in restoring sagebrush grasslands. Fifth International Fire Ecology and Management Congress, 3-7 December 2012, Portland, Oregon. Rigby, D., Z. Aanderud, and B. Roundy. 2012. Investigating the potential for piñon and juniper mastication to influence soil health and invasibility. Society for Range Management 65th Annual Meeting, 8 January-3 February 2012, Spokane, Washington. Roundy, B., K. Young, A. Hulet, R. Miller, R. Tausch, and J. Chambers. 2012. Relating woodland fuel treatments to resilience across an invasion gradient. Utah Section Society for Range Management Meeting. 2 Dec 2012, Utah

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		Bunting. 2013. Utilizing NAIP imagery to estimate tree cover and biomass in pinyon and juniper woodlands. Society for Range Management 66th Annual Meeting, 2-8 February 2013, Oklahoma City, Oklahoma. Roundy, B. A. 2013. Does tree cover matter when we treat pinyon-juniper to control fuels? Factors in cost-effective restoration of sage grouse habitat in northern Nevada. 10 January 2013. Nevada Society for Range Management Winter Meeting, Elko, Nevada. Roundy, B., R. Miller, R. Tausch, J. Chambers, A. Hulet, and K. Young. 2013. Effects of fuel control treatments on vegetation responses across a pinyon-juniper tree invasion gradient. Society for Range Management 66th Annual Meeting, 2-8 February 2013, Oklahoma City, Oklahoma. Roundy, D., A. Hulet, and B. Roundy. 2013. Estimating pinyon and juniper tree cover using NAIP imagery across Utah. Society for Range Management 66th Annual Meeting, 2-8 February 2013, Oklahoma City, Oklahoma. Young, K. and B. Roundy. 2013. Plant establishment and soil microenvironments in Utah juniper masticated woodlands. Society for Range Management 66th Annual Meeting, 2-8 February 2013, Oklahoma City, Oklahoma. Aanderud, Z. T., D. Rigby, and B. A. Roundy. 2014. Tree-islands of fertility and masticated debris decrease metabolic efficiency of soil microorgamisms in cold deserts. Society for Range Management 67th Annual Meeting, 8-13 February 2014, Orlando, Florida. Bybee, J., B. A. Roundy, A. Hulet,K. Young, and D. Roundy. 2014. Understory vegetation response to mechanical mastication of piñon-juniper forests. Society for Range Management 67th Annual Meeting, 8-13 February 2014, Orlando, Florida.

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Theses	Graduate student theses on vegetation, fuel, and soil responses	Bybee, Jordan. 2013. Understory vegetation response to mechanical mastication of piñon and juniper woodlands. MS Thesis, Brigham Young University, Provo, Utah. 28 p. Rigby, Deborah. 2013. Microbial responses to coarse woody debris in *Juniperus* and *Pinus* woodlands. MS Thesis, Brigham Young University, Provo, Utah. 42 p. Shakespear, W. 2014. Fuel response to

		mechanical mastication of pinyon-juniper woodlands in Utah. MS Thesis, Brigham Young University, Provo, Utah. 37 p.
Refereed publications	Scholarly journal manuscript submission At least 4 additional ones in preparation	Young, K. R., B. A. Roundy, and D. Eggett. 2013. Plant establishment in masticated Utah juniper woodlands. Rangeland Ecology and Management 66:597-607. Young, K. R., B. A. Roundy, and D. Eggett. 2013. Tree reduction and debris from mastication of Utah juniper alter the soil climate in sagebrush steppe. Forest Ecology and Management 310:777-785. Bybee, J., B. A. Roundy, K. R. Young, A. Hulet, D. B. Roundy, L. Crook, Z. Aanderud, and D. L. Eggett. 2014. Vegetation response to tree shredding of piñon and juniper woodlands. Submitted to Journal of Environmental Management June 2014. Roundy, B. A., R. F. Miller, R. J. Tausch, K. Young, A. Hulet, B. Rau, B. Jessop, J. C. Chambers, and D. Eggett. 2014. Understory responses of piñon –juniper treatments across tree dominance gradients in the Great Basin. Accepted to Rangeland Ecology and Management Feb. 2014 Young, K. R., B. A. Roundy, S. C. Bunting, and D. L. Eggett. 2014. Utah juniper and two needle piñon reduction alters fuel loads. Accepted pending revision for International Journal of Wildland Fire Young, K. R., B. A. Roundy, and D. Eggett . 2014. Mechanical mastication of Utah juniper encroaching sagebrush steppe increases inorganic soil N. Accepted by Applied and Environmental Soil Science, April 2014.
Final Fact Sheet posted on SageSTEP website and printed	Summary of project findings and management guidelines	SageSTEP Newsletter Issue 22, Fall 2013