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Vegetation Response to Piñon and Juniper Tree Shredding☆

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ABSTRACT

Piñon (*Pinus* spp.) and juniper (*Juniperus* spp.) expansion and infilling in sagebrush (*Artemisia* L.) steppe communities can lead to high-severity fire and annual weed dominance. To determine vegetation response to fuel reduction by tree mastication (shredding) or seeding and then shredding, we measured cover for shrub and herbaceous functional groups on shredded and adjacent untreated areas on 44 sites in Utah. We used mixed model analysis of covariance to determine significant differences among ecological site type (expansion and tree climax) and treatments across a range of pretreatment tree cover as the covariate. Although expansion and tree climax sites differed in cover values for some functional groups, decreasing understory cover with increasing tree cover and increased understory cover with tree reduction was similar for both ecological site types. Shrub cover decreased by 50% when tree cover exceeded 20%. Shredding trees at ≤ 20% cover maintained a mixed shrub (18.6% cover)—perennial herbaceous (17.6% cover) community. Perennial herbaceous cover decreased by 50% when tree cover exceeded 40% but exceeded untreated cover by 11% (20.1% cover) when trees were shredded at 15—90% tree cover. Cheatgrass (*Bromus tectorum* L.) cover also increased after tree shredding or seeding and then shredding but was much less dominant (<10% cover) where perennial herbaceous cover exceeded 42%. Sites with high cheatgrass cover on untreated plots had high cheatgrass cover on shredded and seeded-shredded plots. Seeding and then shredding decreased cheatgrass cover compared with shredding alone when implemented at tree cover ≥ 50%. Vegetation responses to shredding on expansion sites were generally similar to those for tree cutting treatments in the SageSTEP study. Shredding or seeding and then shredding should facilitate wildfire suppression, increase resistance to weed dominance, and lead toward greater resilience to disturbance by increasing perennial herbaceous cover.

Introduction

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). Piñon (*Pinus* spp.) and juniper (*Juniperus* spp.) expansion and subsequent infilling in sagebrush communities can increase the risk of fast-spreading crown fires (Gruell, 1999; Miller et al., 2014; Young et al., 2015). High-severity fire may cause the system to pass a biotic threshold into an alternate stable state of weed dominance and recurrent fire (D'Antonio and Vitousek, 1992; Miller and Tausch, 2001). The resulting highly flammable, annual weed-dominated community makes system restoration difficult (Bagchi et al., 2013; Chambers et al., 2014). 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 sagebrush-obligate wildlife species like sage-grouse *(Centrocercus urophasianus)* (Connelly et al., 2004) and pygmy rabbit *(Brachylagus idahoensis)* (Wilson et al., 2011), as well as for vegetation management that increases carbon sequestration by avoiding weed dominance (Prater et al., 2006; Rau et al., 2011, 2012).

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Managers seek to use fuel treatments to increase the ability of the ecosystem 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 nutrients made available by tree reduction increase growth and cover of both desirable vegetation and invasive species (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 expansion 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).

Although prescribed fire best controls the amount of fuels (Young et al., 2015), mechanical tree reduction by cutting or shredding (i.e., mastication) offers a number of operational and ecological advantages. Mechanical tree control is much easier to implement than prescribed fire and can be selectively applied (e.g., thinning, clear-cutting, mosaics) almost any time of year when the soil surface is dry. Shredding trees with a large, toothed drum (Cline et al., 2010) converts canopy fuels to small 1- and 10-hour surface fuels. By moving fuels from the canopy to ground level, flame lengths are reduced, which better facilitates wildfire containment (Young et al., 2015). Debris from shredding also increases infiltration and reduces sediment production on some microsites (Cline et al., 2010). Effects of mastication have been compared with prescribed fire in chaparral (Potts and Stephens, 2009; Potts et al., 2010) and with fire and cutting in ponderosa pine (*Pinus ponderosa* P. & C. Lawson) forests (Kane et al., 2010) but have received little attention in piñon and juniper woodlands, although managers have been using it on hundreds of hectares in Utah since about 2003.

Because understory cover decreases with increasing tree cover (Miller and Rose, 1999; Archer et al., 2011; Archer and Predick, 2014; Roundy et al., 2014a), system resilience should be reinforced by reducing trees while there is still sufficient desirable understory cover to dominate and suppress weeds after tree reduction. Effects of increasing tree cover, such as decreased shrub and perennial herbaceous cover, generally become evident at Phase II infilling, when trees and perennial understory plants have similar relative cover (Roundy et al., 2014a). Piñon and juniper trees have dense lateral root systems and well-developed tap roots, which reduce availability of soil water and nutrients for understory shrubs, forbs, and grasses (Krimer et al., 1996; Ryel et al., 2010; Rau et al., 2011; Leffler and Ryel, 2012; Young et al., 2013a; Roundy et al., 2014b).

Lack of perennial understory cover at advanced phases of tree expansion leads managers to seed some sites to minimize erosion and dominance by invasive weeds. Mixtures of native and introduced grasses, forbs, and shrubs are usually aerially sown before tree shredding or other mechanical tree reduction methods. Shredding after seeding is considered to help bury seeds and improve establishment, similar to chaining after aerial seeding burned piñon-juniper sites (Ott et al., 2003). Ross et al. (2012) reported increases of > 20%, 15%, and 5% cover for perennial grasses, perennial forbs, and annual grasses after aerial seeding and mastication of upland piñon-juniper sites on Shay Mesa in southeastern Utah. Shredding of dense piñon or juniper trees in Utah increased spring soil degree days, available water, and seedling growth of hand-sown perennial and annual grasses (Young et al., 2013a, b). In those studies, tree reduction had a much greater effect on soil degree days (probably due to removal of tree shading) and soil water availability (probably due to decreased transpiration) than did increased woody debris. Vegetation response to aerial seeding and shredding trees across a range of sites has not been reported.

Besides choosing a method of fuel reduction, managers must also decide which sites to treat. Piñon and juniper trees occupy tree climax sites or sites originally occupied by sagebrush communities (expansion sites). 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; contain trees > 150 years old; and support infrequent fire. In contrast, expansion 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. Infilling (increase in tree density) and increases in tree cover are occurring on both kinds of sites (Romme et al., 2009). Generally, tree reduction treatments are directed at expansion sites but they are also implemented to increase understory cover on tree climax sites, especially in the Colorado Plateau (Ross et al., 2012). However, there have been no reports of effects of tree reduction on expansion compared with tree climax sites.

Piñon and juniper expansion into sagebrush steppe occurs across a broad geographical area in the western United States, and therefore so do tree reduction treatments. These sites vary in elevation, temperature, and precipitation, as well as in soil depth and texture, which relate to tree cover and understory composition and cover. A major objective of the Sagebrush Steppe Treatment Evaluation Project (SageSTEP) research project was to determine patterns of regional response to tree control treatments without seeding (McIver and Brunson, 2014). Our current study was to determine vegetation responses to increasing tree cover and shredding across a broad range of sites in Utah and how our patterns of response compare to regional responses reported by SageSTEP (Chambers et al., 2014; Miller et al., 2014; Roundy et al., 2014a). In these previous studies, mechanical treatments mainly included cut and drop, except for four Utah sites, which also included a tree shredding treatment (Roundy et al., 2014a). Cut-and-drop treatments produced similar vegetation responses as tree shredding on the Utah sites (Roundy et al., 2014a). In short-term results from the SageSTEP study (3 years post-treatment), mechanical treatments increased perennial herbaceous cover and maintained shrub cover better than prescribed fire. Mechanical tree reduction even increased perennial herbaceous cover at high initial tree cover but also increased cheatgrass cover on warmer sites (Chambers et al., 2014; Roundy et al., 2014a).

To better guide implementation of tree reduction treatments aimed at enhancing system resilience across a wide range of sites, we asked three questions about vegetation response to tree shredding across a tree expansion-infilling gradient in Utah: 1) how do expansion sites respond compared with tree climax sites, 2) what is the effect of seeding, and 3) how do patterns of response to shredding across Utah compare with responses to cutting across the Great Basin found in the SageSTEP study?

Methods

Our study included data collected from the SageSTEP woodland experiment (Roundy et al., 2014a) in 2009 and 2010, as well as data collected from a retrospective study in 2011 and 2012. For both the SageSTEP and retrospective studies, the experimental design was a randomized block with site considered as a block. Data from the SageSTEP experiment came from treatments implemented on nine sites of tree expansion into sagebrush steppe vegetation. Each site had treatment plots, 8–20 ha in size, including untreated control, prescribed fire, and cut-and-drop that encompassed a range of pretreatment tree cover (Roundy et al., 2014a). At three Utah sites, a shredded-tree treatment plot was also added. Data from the SageSTEP study that were used in the current study included vegetation response for untreated control and cut treatment plots, 3 years after treatment. For the cut treatment, all trees > 2 m in height were cut and debris left in place either in 2006 or 2007, depending on the site (Miller et al., 2014; Roundy et al., 2014a). We also remeasured and used data of vegetation response on the untreated control and shredded plots for the three SageSTEP Utah sites at 5 (Scipio and Greenville) and 6 (Onaqui) years post treatment (2012).

In addition, we conducted a retrospective study (Utah shred study) comparing vegetation cover for untreated areas with nearby shredded or seeded-shredded areas for an additional 41 expansion and tree climax sites in Utah, United States. On both the SageSTEP Utah sites that included a shred treatment and on the retrospective study sites, trees > 2 m in height were shredded or masticated using a rotating, toothed drum or Fecon Bullhog attachment (Fecon, Inc. Lebanon, Ohio) as described by Cline et al. (2010). For the retrospective study, we used pretreatment National Agriculture Imagery Program (NAIP) data to select sample subplots to compare vegetation across similar low to high ranges of untreated tree cover for subplots on untreated areas and pretreatment tree cover for subplots on nearby shredded or seeded-shredded areas on the same ecological site type. On a few sites where trees were either thinned or some mature trees left untreated, measurements of the shredding treatment were made on subplots > 20 m from existing trees.

Study Sites

Study sites for the SageSTEP experiment included four sites of western juniper (*Juniperus occidentalis* Hook) expansion in California and Oregon (Blue Mountain, Bridge Creek, Devine Ridge, and Walker Butte), two sites of singleleaf piñon (*Pinus monophyla* Torr. & Frém.) and Utah juniper (*Juniperus osteosperma* [Torr.] Little) expansion in central Nevada (Marking Corral and Seven Mile), and three sites of Utah juniper—Colorado piñon (*Pinus edulis* Engelm.) expansion in Utah (Onaqui, Scipio, and Greenville). Details of these sites and maps of their locations are in McIver et al. (2010), McIver and Brunson (2014), and Miller et al. (2014).

Study sites for the retrospective shred study were located within the state of Utah in the Great Basin and Colorado Plateau physiographic provinces on lands managed by either the Bureau of Land Management (BLM) or US Department of Agriculture Forest Service (Fig. 1). The retrospective Utah shred study and three Utah SageSTEP sites with a tree-shredded treatment encompassed a range of time since treatment (1–8 years, mean = 3.9 years) and two ecological site types as determined by NRCS (1997) criteria (expansion—“Range” or tree climax—“Forest” sites). Hereafter, we refer to the sites as either expansion or tree climax sites. Global Positioning System coordinates for each retrospective study site were used to determine soil and ecological site type using the Web Soil Survey (http://websoilsurvey.nrcs.usda.gov). For a few sites where the ecological site type had not been determined by a soil survey, we categorized the site as an expansion site if soil depth was > 50 cm (NRCS, 1997). Across all sites, tree cover ranged from 2% to 90% on untreated areas and pretreatment tree cover ranged from 3% to 79% on treated areas (Table S1, available online at http://dx.doi.org/10.1016/j.rama.2016.01.007).

For the retrospective Utah shred study and the three SageSTEP sites with a shredding treatment, the majority of sites in the Great Basin were expansion sites (26 of 29), while the majority of sites in the Colorado

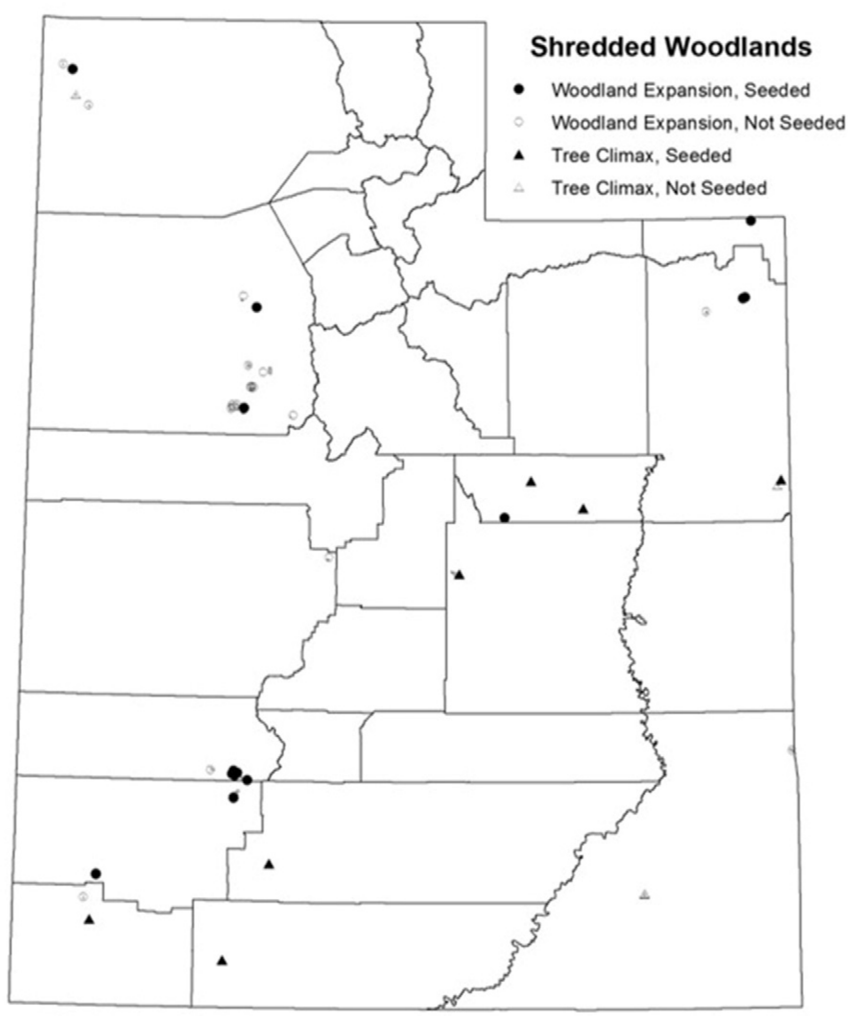

Figure 1. Map of 44 sites in Utah, United States, by ecological site and treatment.

Plateau were tree climax sites (11 of 15). Each site had an untreated control area and either a shredded only or a seeded-shredded treatment. Seed was aerial broadcast before treatment according to specifications of the individual agency and primarily included perennial grasses, shrubs, and forbs. Before data collection, we made field visits and checked soil surveys to select untreated and treated sample areas with the same ecological sites on each of 44 study sites. Within untreated or treated areas at each study site, we randomly selected 12 to 15 potential subplots (0.1-ha) for sampling that represented a range of untreated or an apparent range of pretreatment tree cover. 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 tree cover (Hulet et al., 2014; Roundy, 2015) on the potential subplots. We randomly selected three subplots each on untreated and treated plots from the potential subplot population for each of three tree cover categories: low (<15%), intermediate (15−45%), and high (>45%). Not all study sites had all tree cover categories, so the number of subplots ranged from a minimum of six (1 tree cover category × 2 treatments—untreated and shredded × 3 subplots each = 6) to 18 (3 tree cover categories × 2 treatments × 3 subplots each = 18). The only exception to this sampling scheme was for the three SageSTEP sites in Utah that included a tree-shredding treatment. On those sites, 15 subplots were measured on untreated and shredded plots across a range of pretreatment tree cover. Because sites were either shredded and left unseeded or seeded and then shredded, these two treatments occurred on different sites.

Measurements

Vegetation measurements on each 0.1-ha subplot were made according to the protocol of McIver et al. (2010) and Miller et al. (2014). We used the line-point intercept method to measure cover by species on five 30-m transects on each subplot. We dropped a pin flag every 0.5-m (60 points × 5 transects = 300 points in each subplot). At each point, we recorded canopy and/or foliar hits on vegetation, as well as the ground surface category. Canopy cover was recorded for trees and shrubs where the point fell within the live canopy perimeter. Foliar cover was recorded when the point came in contact with one or more species. Ground surface categories included soil, rock, lichen or biotic crust, bedrock, moss, duff, and embedded litter. To calculate cover for a subplot, we summed all of the hits on a particular species and divided by 300. We counted density of herbaceous species and sagebrush seedlings and juveniles (<5 cm height) in fifteen 0.25-m^2 quadrats placed on three 30-m transects for a total of 45 quadrats in each subplot.

For the SageSTEP study sites, live tree cover of all trees > 2 m tall was measured before treatment within each subplot by measuring the longest (D1) and perpendicular (D2) crown diameters. Crown area (A) was calculated for each tree according to Tausch and Tueller (1990) by:

$$A = \pi(D1 \cdot D2)/4 \quad (1)$$

Tree canopy cover was estimated for each subplot by summing the crown area for each tree in the subplot and dividing by the subplot area. For the retrospective study sites, tree cover was estimated before treatment on both untreated control and treatment areas using National Agricultural Imagery Program (NAIP) imagery. First, live tree cover on untreated areas at each study site was measured on the ground and calculated using the crown diameter method described earlier. Crown diameter tree cover was highly correlated with tree cover estimates derived from object-based image analysis of NAIP imagery ($r = 0.93$ for ENVI Feature Extraction, Roundy, 2015). Therefore, we estimated tree cover in the retrospective study on untreated subplots and on treated subplots before treatment using NAIP imagery and ENVI Feature Extraction. Hereafter we refer to the tree cover variable simply as tree cover, noting that this refers to untreated tree cover and pretreatment tree cover estimated on both the SageSTEP and retrospective studies before any tree-reduction treatments.

Data Analysis

We used mixed model analysis of covariance (Littell et al., 2006; Proc Glimmix, SAS v9.3, SAS Institute, Inc., Cary, NC) to compare responses of functional groups between untreated control and tree-shredded areas. These cover groups included total shrubs; sagebrush; tall, short, and total perennial grass (tall, short, and rhizomatous grasses); cheatgrass (*Bromus tectorum* L. was considered separately due to concern for its dominance); perennial forbs; sage-grouse forbs (Connelly et al., 2004; Nelle et al., 2000; Pyle and Crawford, 1996; Rhodes et al., 2010); and annual forbs, total perennial herbaceous (total perennial grass plus perennial forbs), biotic crust (including lichen), and bare ground. Sandberg bluegrass (*Poa secunda* J. Presl) was considered the only short grass, while all other bunchgrasses were treated as tall grasses in the analysis. A list of species encountered is in Table S2 (available on line at http://dx.doi.org/10.1016/j.rama.2016.01.007). Our analysis included ecological site type and treatment as fixed factors. Tree cover was analyzed as a covariate (Roundy et al., 2014a). When interactions of ecological site type or treatment with the tree cover covariate were not significant ($P <$ 0.05), these terms were removed from the model (Littell et al., 2006). Site was considered as a random factor, and subplots (524 total across 44 sites) were nested as subsamples within treatment and ecological site type. The Tukey-Kramer test was used to determine significant differences among ecological site type and treatment combinations for each 5% increment of tree cover for covariate model estimates of cover for functional cover groups (Littell et al., 2006; Roundy et al., 2014a) using a $P \leq 0.05$. Vegetation 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.

To compare tree-cutting and shredding treatment responses, we compared functional group cover from the SageSTEP study with that from the retrospective study. We conducted analysis of covariance using study (SageSTEP study or retrospective Utah shred study), treatment (untreated and cut-drop for SageSTEP, untreated and shred for Utah shred study) as fixed factors and tree cover as the covariate. Individual study sites were considered random. The data from SageSTEP were from nine tree expansion sites, 3 years after treatment. The data from the Utah shred study was for 16 tree expansion sites that were shredded but not seeded. Average time since treatment for this population was 2.5 years. Significance of study effects or interactions with treatment or covariate effects were used to determine if understory responses to tree cutting and mastication are generally similar.

We used quantile regression to relate cheatgrass cover across sites and treatments to perennial herbaceous cover. This technique permits quantification of the effect of an independent variable on the upper boundary of response of a dependent variable and is therefore useful in determining effects over the range of response where one variable may be most limiting to another (Cade and Noon, 2003).

Results

Vegetation Response to Tree Cover, Ecological Site Type, and Tree Shredding

Functional groups varied in their response to ecological site type, treatment, and tree cover (Table 1). Both total shrub and sagebrush cover decreased with increasing tree cover on untreated and tree-shredded areas and did not differ between the two. Total shrub and sagebrush cover had a significant interaction between ecological site type and tree cover (see Table 1) because they were slightly greater on tree climax than expansion sites at ≥ 60% tree cover (Table 2, Fig. 2). However, the higher shrub cover at high tree cover on tree climax sites was influenced by high shrub cover at the Indian Springs site, which had the highest elevation and precipitation of all tree climax sites. Sagebrush cover was decreased by 50% of maximum when tree cover was > 15% on expansion sites and > 20% on tree climax sites. Measurable shrub and sagebrush cover persisted until tree cover reached 60% on expansion sites and 75% on tree climax sites. We observed sagebrush seedlings on 61% of the 44 study sites. For these sites, the number of seedlings was significantly greater ($P < 0.05$) on shredded (0.43 m^{-2}) compared with untreated plots (0.073 m^{-2}).

Annual forb cover was < 4% at minimum tree cover and decreased to < 1% at maximum tree cover on untreated expansion sites. After shredding, annual forb cover increased at ≥ 45% tree cover on expansion sites to a maximum of 7% (see Table 2). In contrast, annual forb cover was < 2% on tree climax sites for any treatment or tree cover. Sage-grouse forb cover was limited (<2.8% maximum) and similar on untreated and shredded plots (see Table 2). Perennial forb cover was low and not significantly affected by tree cover or by shredding only (see Tables 1 and 2).

Shortgrass cover (Sandberg bluegrass) varied by ecological site type and tree cover, but not by treatment (see Table 1). Sandberg bluegrass cover averaged 5% higher on expansion than tree climax sites at < 50% tree cover (see Fig. 2). Sandberg bluegrass density was 8.3 plants m^{-2} on expansion sites and 3.5 plants m^{-2} on tree climax sites ($P = 0.053$).

Perennial herbaceous cover was not quite as sensitive to initial increases in tree cover as was shrub cover (see Figs. 2–4). Tall grass, perennial grass, and perennial herbaceous cover were reduced by 50% of maximum on untreated plots at tree covers of 30%, 45%, and 50% on expansion sites, and 45%, 55%, and 60% on tree climax sites, respectively. Tall grass and total perennial grass cover had significant two-way interactions of ecological site type with tree cover and treatment with tree cover while total perennial herbaceous cover had a significant interaction only between treatment and tree cover (see Table 1). On expansion sites, tall and total perennial grass cover was higher on shredded than untreated areas but decreased as tree cover increased (see Fig. 4, Table 2). In contrast for tree climax sites that were shredded, tall and total perennial grass cover increased slightly with increasing tree cover (see Fig. 4). Shredding more than doubled tall grass cover from 10% to 22% on expansion sites at ≥ 15% tree cover and more than tripled it from 3.8% to 13% on tree climax sites at ≥ 40% cover (see Fig. 4, Table 3). The two-way interactions of ecological site type and tree cover and treatment and tree cover were significant ($P = 0.02$) for tall grass density. On expansion sites, shredding increased tall grass density by 4.6 plants m^{-2} at 15−50% tree cover.

There was a significant interaction between treatment and tree cover ($P < 0.0001$) for cheatgrass cover (see Table 1, Fig. 3). Shredding increased cheatgrass cover compared with untreated control areas at ≥ 25% tree cover. Cheatgrass cover was < 5% on untreated areas but increased with increasing tree cover to a maximum of 26.5% on shredded only areas (see Fig. 3). Expansion sites had higher cheatgrass density (44.9 plants m^{-2}) than tree climax sites (13.1 plants m^{-2}; $P = 0.057$).

There was a significant interaction ($P = 0.0059$) between treatment and tree cover for bare ground cover, which was significantly lower ($P < 0.05$, 9.5% less) on shredded than untreated areas for all tree cover values (see Table 2, Fig. 2). Bare ground cover was 6% higher ($P = 0.0197$) on tree climax than expansion sites averaged across the range of tree cover and across all treatments. Lichen plus biotic crust soil cover was low (<0.6%) and did not vary significantly with any factors (see Table 1).

Vegetation Response to Seeding

Seeding-shredding cover and density estimates were generally higher than those for shredding alone, although the responses to these two treatments were usually statistically similar (see Table 2). Sagebrush cover was higher (7.3%, $P < 0.05$) on the shredded-seeded than untreated (4.7%) and shredded (4.1%) treatments. On the sites that had sagebrush seedlings (61%), seedling density was 0.65 m^{-2} and significantly greater ($P < 0.05$) than on the untreated plots but statistically similar to shredded, not seeded, plots. Annual forb cover was greater on seeded-shredded plots than untreated plots at a higher tree cover (≥60%) than shredded-only plots (≥45%, see Table 3). Sage-grouse forb cover was increased from 1.2% on untreated plots to 5.5% on seeded-shredded plots at ≥ 35% tree cover. Perennial forb cover was significantly higher ($P < 0.05$, 3.9%) on seeded-shredded than untreated (1.3%) and shredded (2.2%) areas.

Tall grass, perennial grass, and perennial herbaceous cover did not vary significantly between shredded and seeded-shredded plots (see Table 3), but seeding-shredding generally increased cover more compared with untreated plots than did shredding alone, especially on tree climax sites (see Figs. 3 and 4). As with perennial grass cover,

Table 1
Mixed model analysis of covariance results for cover (%) of various species and functional groups.[1] When the covariate (tree cover) or main effect by covariate interactions were not significant ($P > 0.05$), they were omitted from the model (Littell et al. 2006). Short grass was only one species, Sandberg bluegrass

		Total shrub			Sagebrush		
Effect	NDF	DDF	F	P	DDF	F	P
Ecological site (ES)	1	77.08	2.15	0.1469	95.75	5.99	**0.0162**
Treatment (TRT)	2	47.05	1.6	0.2134	45.93	6.4	**0.0035**
ES*TRT	2	47.05	0.78	0.4635	45.93	0.13	0.8793
Tree cover (TC)	1	499.8	318.63	**<0.0001**	501.5	345.78	**<0.0001**
TC*ES	1	499.8	15.08	**0.0001**	501.5	19.16	**<0.0001**
		Annual forb			Sage-grouse forb		
Effect	NDF	DDF	F	P	DDF	F	P
Ecological site (ES)	1	77.45	3.23	0.0764	105.6	0.05	0.8192
Treatment (TRT)	2	115.7	0.97	0.3818	127	1.11	0.3339
ES*TRT	2	115.7	4.84	**0.0096**	127	0.93	0.3991
Tree cover (TC)	1	504	1.76	0.1856	481.2	2.97	0.0857
TC*TRT	2	340.8	5.13	**0.0064**			
TC*ES*TRT	2	408.48	4.01	**0.0078**	451.9	2.05	0.0703
		Perennial forb			Short grass		
Effect	NDF	DDF	F	P	DDF	P	F
Ecological site (ES)	1	44.62	1.11	0.2977	41.68	4.58	**0.0382**
Treatment (TRT)	2	44.66	21.57	**<0.0001**	39.41	0.74	0.4833
ES*TRT	2	44.66	1.87	0.1663	39.41	0.63	0.5354
Tree cover (TC)	1			470.1	10.18	**0.0015**	
		Tall grass			Total perennial grass		
Effect	NDF	DDF	F	P	DDF	F	P
Ecological site (ES)	1	84.58	1.08	0.3020	63.8	4.49	**0.038**
Treatment (TRT)	2	123.3	1.01	0.3672	102.7	0.55	0.581
ES*TRT	2	52.39	0.31	0.7369	43.44	3.53	**0.038**
Tree cover (TC)	1	513.6	0.69	0.4063	502.8	0.84	0.3593
TC*ES	1	505.3	4.00	**0.0460**	494.9	4.04	**0.0449**
TC*TRT	2	387.6	16.2	**<0.0001**	388.7	15.12	**<0.0001**
		Total perennial herbaceous			Cheatgrass		
Effect	NDF	DDF	F	P	DDF	F	P
Ecological site (ES)	1	42.67	1.05	0.3117	43.6	4.47	**0.0403**
Treatment (TRT)	2	99.93	0.79	0.4581	103.6	0.75	0.4769
ES*TRT	2	42.54	3.19	0.0510	43.9	4.36	**0.0187**
Tree cover (TC)	1	496.6	5.51	**0.0193**	504.7	1.26	0.2613
TC*TRT	2	388.8	15.67	**<0.0001**	398.6	15.4	**<0.0001**
		Biotic crust			Bare ground		
Effect	NDF	DDF	F	P	DDF	F	P
Ecological site (ES)	1	42.32	0.01	0.9101	46.43	5.84	**0.0197**
Treatment (TRT)	2	46.93	1.45	0.2439	121	4.00	**0.0199**
ES*TRT	2	46.93	0.08	0.9195	50.65	0.66	0.519
Tree cover (TC)	1				509.3	43.83	**<0.0001**
TC*TRT	2				366.7	5.21	**0.0059**

[1] NDF indicates numerator degrees of freedom; DDF, denominator degrees of freedom calculated according to Kenward and Roger (1997).

Table 2
Pretreatment tree cover ranges over which understory cover responses to treatments were similar (=) or one treatment was greater than another ($P < 0.05$).[1] Numbers in parentheses indicate mean response for the treatments being compared across the listed range of pretreatment tree cover.

	Across ecological sites			By ecological site		
		Tree cover			Tree cover (%) (Response cover %)	
	Response (cover %)	(%)		Response	Expansion	Tree climax
Total shrub	T > E (1.4 > 0.2%)	≥60	Annual forbs	S > UT	≥45 (8.5 > 0.9%)	
				SS > UT	≥60 (5.7 > 0.7%)	
Sagebrush	T > E (0.2 > 0%)	≥60		S = SS	0-90 (4.6, 3.4%)	0-90 (1.4, 0.8%)
Perennial herbaceous	S (20.1%) > UT (9%)	≥15		S = UT		0-90 (1.4, 0.2%)
	SS (27%) > UT (9%)	≥15		SS = UT		0-90 (0.8, 0.2%)
	S (20.7%) = SS (26.2%)	0-90				
			Sage-grouse forbs	S = UT	0-90 (2.2, 1.1%)	0-90 (1.5, 1.1%)
Perennial forb	UT (1.3%) = S (2.2%)	0-90		SS > UT	≥35 (5.5 > 1.2%)	
	SS (3.9) > UT (1.3%)	0-90		S = SS	0-90 (2.3, 4.3%)	0-90 (2.4, 5.5%)
	SS (3.9%) > S (2.2%)	0-90		SS = UT		0-90 (1.7, 1%)
			Tall grass	S > UT	≥15 (9.7 > 2.3%)	≥40 (10.5 > 2%)
Cheatgrass	S (16%) > UT (1.1%)	≥25		SS > UT	≥25 (11.6 > 1.8%)	≥20 (15.5 > 2.6%)
	SS (5.2%) > UT (0.8%)	≥35		S = SS	0-90 (10, 11.1%)	0-90 (9.7, 14.1%)
	S (19.7%) > SS (5.1%)	≥50				
			Perennial grass	S > UT	≥20 (22 > 9.4%)	≥65 (13.6 > 3%)
Bare ground	UT (21.4%) > S (11.9%)	0-90		SS > UT	≥30 (21 > 8.8%)	≥15 (24.5 > 4.7%)
	UT (20.8%) > SS (8.9%)	≥10		S = SS	0-90 (23, 20.3%)	0-90 (12.3, 23%)
	S (11.9%) = SS (10.2%)	0-90				

[1] E indicates tree expansion sites; S, trees shredded; SS, seeded and then trees shredded; T, tree climax sites; UT, untreated.

total perennial herbaceous cover decreased with increasing tree cover on shredded areas but increased with increasing tree cover on seeded-shredded areas (see Figs. 3 and 4). On expansion sites, shredding increased tall and total perennial grass cover compared with untreated areas at lower tree cover than seeding-shredding, while the reverse occurred on tree climax sites (see Table 3). Compared with untreated plots, seeding-shredding increased tall grass cover by 7.6% at ≥ 25% tree cover on expansion sites and by 13% at ≥ 20% tree cover on tree sites.

Increases in tall grasses after seeding and shredding were also reflected in density data. On expansion sites, shredding increased tall grass density by 4.6 plants m^{-2} at 15—50% tree cover, while seeding-shredding increased tall grass density by 6.1 plants m^{-2} at 25—90% tree cover. On tree climax sites tall grass density was not statistically increased by shredding alone, but was increased by seeding and shredding by 8.6 plants m^{-2} at 30—90% tree cover.

Cheatgrass cover varied widely across the study sites (Fig. 5). Fifteen sites had > 18% cheatgrass cover (6 of 44 sites for untreated areas; 9 of 44 sites for shredded or shredded-seeded areas). For many sites, and especially untreated plots, cheatgrass cover was low and there was little relationship between it and perennial herbaceous cover (see Fig. 5). Quantile regression analysis indicated a significant negative slope ($P = 0.0116$) for the 80th percentile of cheatgrass cover for all untreated and treated plots and perennial herbaceous cover (see Fig. 5). The quantile regression equation estimated cheatgrass cover to be < 10% when perennial herbaceous cover exceeded 42%. Nevertheless, there were some plots with both high perennial herbaceous and cheatgrass cover (see Fig. 5).

Cheatgrass cover was increased compared with untreated areas at ≥ 50% tree cover for seeding and shredding. Cheatgrass cover averaged ≤ 5.5% on seeded-shredded areas, across the range of tree cover compared with 13.8% on shredded only areas (see Fig. 3). The three-way interaction of ecological site type, treatment, and tree cover was significant for cheatgrass density ($P < 0.02$). On expansion sites, seeded-shredded areas had higher cheatgrass density than untreated areas at 20—90% tree cover, while there were no differences in cheatgrass density among treatments on tree climax sites.

On seeded-shredded areas bare ground cover was significantly ($P < 0.05$) lower by an average of 11.9% than on untreated areas at 10—90% tree cover (see Fig. 3). Bare ground cover was similar on shredded and seeded-shredded areas throughout the range of tree cover.

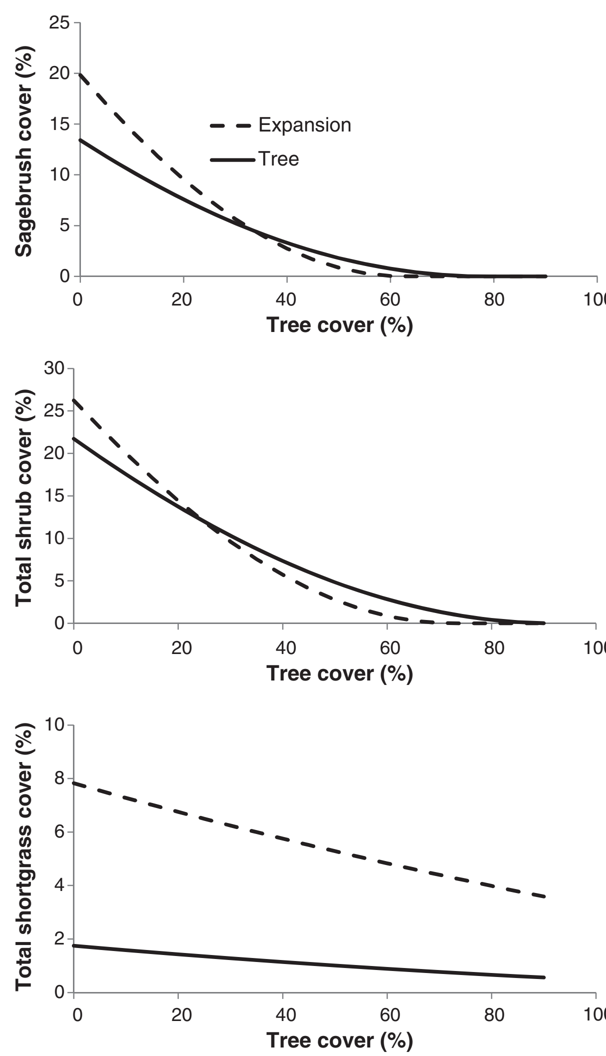

Figure 2. Vegetation cover in relation to tree cover for untreated and tree-shredded areas on piñon-juniper expansion and tree climax sites. The short grass was one species, Sandberg bluegrass. See Table 2 for significant differences.

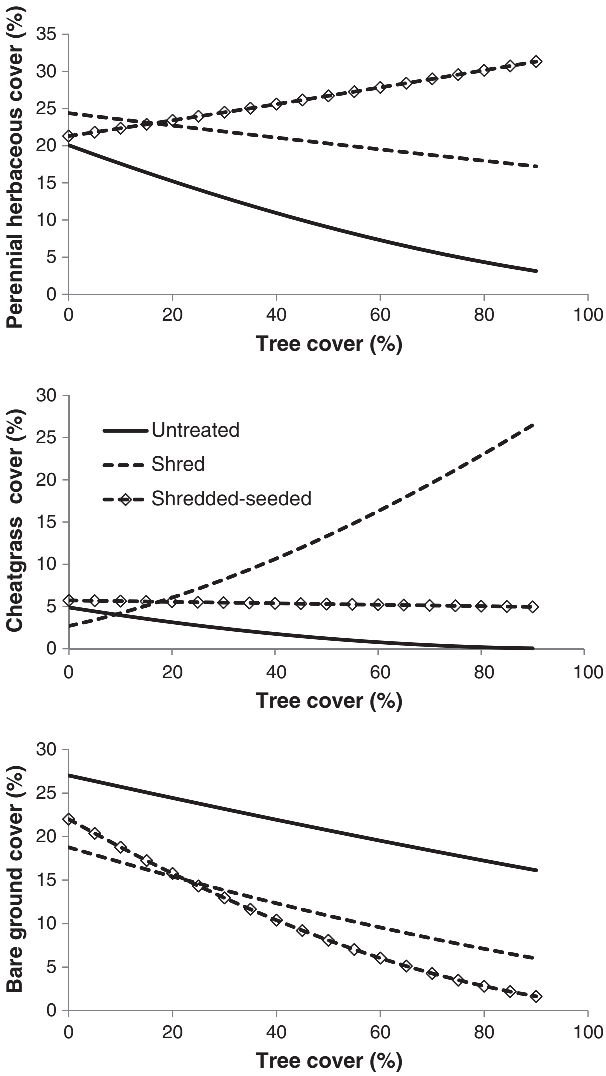

Figure 3. Total perennial herbaceous, cheatgrass, and bare ground cover for untreated, shredded, and seeded-shredded areas in relation to tree cover. See Table 2 for significant differences.

Vegetation Response to Cutting Across the Great Basin Compared with Shredding in Utah

Tree cover before tree reduction treatments for the nine SageSTEP sites across the Great Basin averaged 23.1 ± 0.92% with a maximum cover of 72.2%, while that for the 16 retrospective study Utah expansion sites that were shredded but not seeded averaged 25.7 ± 1.2% with a maximum of 90%. Mechanical treatments did not significantly ($P <$ 0.05) change shrub cover in either study. However, the interaction of study with tree cover was significant ($P = 0.002$) for shrub cover. This was because maximum shrub cover was higher at low tree cover on the Utah shred study expansion sites than the SageSTEP sites (16.2% compared with 22% estimated at 0% tree cover). Shrub cover was decreased to 50% of maximum at tree cover of > 20% for both studies.

For tall grass, total perennial grass, and perennial herbaceous cover, treatment, tree cover, and the interaction of treatment and tree cover were significant ($P < 0.05$), but the effect of study or interactions with study were not. The absence of a significant effect for the study factor indicates a similar response to cutting and shredding. Across both studies, tall grass cover was decreased to 50% of maximum when tree cover exceeded 35% but was significantly ($P < 0.05$) increased by an average of 6.6% after tree cutting or shredding across the range of tree cover. For both studies, total perennial herbaceous cover was decreased to 50% of maximum cover at > 40% tree cover, while mechanical tree reduction increased ($P < 0.05$) cover an average of 10.6% on subplots with 10—70% tree cover. Short grass (Sandberg bluegrass) cover was not increased ($P \geq 0.05$) by tree reduction using either cutting or shredding. Shortgrass cover was an average of 9.4% higher ($P =$ 0.0376) on Utah expansion than SageSTEP sites, across the range of tree cover. On untreated SageSTEP sites, shortgrass cover decreased to 50% of maximum cover at > 45% tree cover. On untreated Utah expansion sites, shortgrass cover was still 10.8% or 73% of maximum cover at 70% tree cover.

The three-way interaction of study, treatment, and tree cover was significant for both cheatgrass cover ($P = 0.0208$) and perennial forb ($P = 0.0232$) cover. Cheatgrass cover on both studies was low for untreated areas (<2.1%) but increased with mechanical treatment. Cheatgrass cover was significantly ($P < 0.05$) increased by tree cutting on the SageSTEP study at tree cover ≥ 25%, while shredding increased ($P < 0.05$) cheatgrass cover on the Utah shred study at ≥ 20% tree cover. The increase was much greater on Utah shred study expansion (average cheatgrass cover = 11%, maximum = 18%) than SageSTEP sites (average = 4.2%, maximum = 6.3%). Perennial forb cover was limited for both studies but higher ($P = 0.0651$, 5.3% maximum) for the SageSTEP than Utah expansion sites (3% maximum). Mechanical tree reduction significantly ($P < 0.05$) increased perennial forb cover a small amount (3%) for the SageSTEP sites at 30—70% tree cover but had no significant ($P \geq 0.05$) effect in the Utah expansion sites.

Warmer soil temperature regimes are generally associated with greater cheatgrass adaptation, while greater tall grass cover decreases cheatgrass cover (Chambers et al., 2014). Soil temperature regimes were predominately mesic for the Utah shred study (32 mesic, nine frigid, three unknown), while five of the SageSTEP sites were mesic and four were frigid (see Table 3). Sites with > 14% tall grass cover after treatment generally had < 10% cheatgrass cover (see Table 3). An exception was the Augusi Utah shred study site, which had exceptionally high cheatgrass cover post treatment, even with high tall grass cover.

Discussion

Because managers apply treatments across a wide variety of sites, they are interested in knowing robust patterns of response, as well as which kinds of sites are most likely to respond negatively. For regional studies across numerous sites, generalized responses as reported in this Utah shred study and the SageSTEP study will be a function of site environmental conditions, pretreatment tree and other vegetation cover ranges, and time since treatment.

Shrub Response

As shrub cover decreases with increasing tree cover, biodiversity and quality of wildlife habitat are compromised (Huber et al., 1999; Miller et al., 2005). With lack of fire, trees have been expanding and infilling for more than 150 years, resulting in substantial areas with mid to high tree cover (Miller et al., 2005). These lands have already lost much of the shrub component, which may be slow to recover after treatment or wildfire, due to lack of proximity of native seed sources or difficulty in consistently establishing sagebrush in range seeding (Ziegenhagen, 2004; Bates et al., 2007). The relative decrease in shrub cover with increasing tree cover was similar for both the SageSTEP and Utah shred study expansion sites; even though shrub cover was lower for the SageSTEP than Utah sites. Previous studies reported by Miller et al. (2005) indicate that sagebrush or total shrub cover is generally decreased to 25% of maximum when tree cover reaches 50% of maximum potential. These estimates are generally consistent with those for total shrub cover for the SageSTEP and Utah expansion sites where shrub cover was reduced to 25% of maximum shrub cover when tree cover exceeded 35%. For tree climax sites, which had lower maximum shrub cover than the expansion sites, shrub cover was reduced to 25%

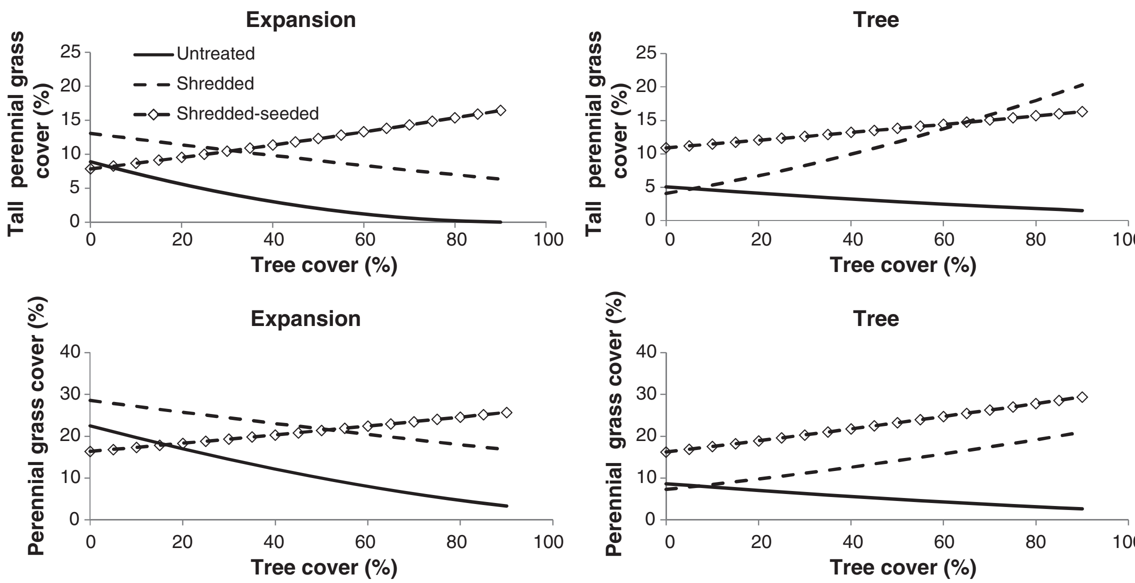

Figure 4. Tall and total perennial grass cover on untreated, shredded, and seeded-shredded areas on piñon and juniper expansion and tree climax sites in relation to tree cover. See Table 2 for significant differences.

of maximum when tree cover exceeded 50%. Chambers et al. (2014) noted an increase of < 5% in shrub cover 4 years after mechanical tree reduction on the SageSTEP expansion sites. In our retrospective Utah shred study, we did not detect an increase in shrub cover when expansion or tree climax sites were measured an average of 4 years after shredding. Shrub response to fire and mechanical fuel reduction treatments is associated with seed dispersal, as well as resprouting and seedbank responses (Potts et al., 2010). Sagebrush seedlings were observed on mechanically treated plots in both the SageSTEP study (Miller et al., 2014) and our Utah shred study, indicating potential for sagebrush recovery. However, for best maintenance of shrubs on most sites, we recommend implementation of mechanical tree reduction before tree cover exceeds 20%. We have consistently found that shrub cover decreases by 50% when tree cover exceeds 20%, and shrub cover increases after mechanical tree reduction can be slow.

Perennial and Annual Herbaceous Response

Cover of perennial tall grasses, especially, and perennial herbaceous plants, generally, is important for resisting cheatgrass dominance and interspace erosion (Pierson et al., 2010, 2013; Williams et al., 2013; Chambers et al., 2014). Perennial herbaceous cover decreased by 50% at higher tree cover on SageSTEP and Utah shred study expansion sites (>40%) and tree climax sites (>55%) than did shrub cover (>20% and 50%). These numbers indicate that perennial herbaceous cover generally appears to be less sensitive than shrub cover to increasing tree cover. Therefore, support of a more resilient (similar to pre-tree expansion), mixed shrub-perennial herbaceous community requires reducing trees at early phases of expansion to maintain shrubs in the community, while treatment at higher tree cover will favor a herbaceous-dominated community.

Perennial herbaceous cover increased similarly with cut-and-drop treatments in the SageSTEP study (Miller et al., 2014; Roundy et al., 2014a) and with tree shredding treatments in the Utah shred study. Perennial herbaceous cover after mechanical tree reduction is associated with increased time of available water (Roundy et al., 2014b) and accelerated growth of residual species (Tausch and Tueller, 1977). Increases in soil water and nitrogen have been found after tree cutting (Bates et al., 2000, 2002). In consequence, cutting or shredding trees at mid to upper tree cover ranges generally resulted in a perennial herbaceous-dominated community for the SageSTEP and Utah expansion sites. Shredding trees at upper tree cover ranges also generally resulted in a perennial herbaceous-dominated community on Utah tree climax sites. One reason for this perennial herbaceous dominance may be that when trees are removed at higher tree cover, fewer shrubs are left to use resources once used by trees, which then become available for increased growth of herbaceous plants (Roundy et al., 2014b). Miller

Table 3
Characteristics of sites receiving mechanical tree reduction treatments.

Study population	Treatment and (years since treated)	Cheatgrass cover (%)	Soil temperature Mesic/frigid sites	Tall grass cover (%)		Cheatgrass/tall grass cover	
				Untreated	Treated	Untreated	Treated
SageSTEP	Cut and drop (3)	<10	4/4	11.8 ± 2.82	19.2 ± 3.7	0.1 ± 0.06	0.1 ± 0.03
		>10	1/0	7.9	12.6	0.3	2.1
Utah expansion	Shredded (2.9)	<10	10/1	7.6 ± 2.25	14.3 ± 3.28	0.2 ± 0.07	0.7 ± 0.25
		>10	3/1	6.5 ± 2.77	12.1 ± 4.33	5.2 ± 4.11	5 ± 2.86
	Seeded-shredded (4.3)	<10	3/0	4.8 ± 2.39	14.1 ± 3.16	1.3 ± 0.94	0.3 ± 0.17
		>10	7/1	5.2 ± 1.86	8.6 ± 1.65	10 ± 4.02	6.4 ± 2.41
Utah tree climax	Shredded (4.5)	<10	2/2	5.2 ± 4.54	9.9 ± 4.55	0 ± 0.02	0.2 ± 0.14
		>10	2/0	6.3 ± 0.15	15.3 ± 3.64	3 ± 1.52	32.4 ± 15.97
	Seeded-shredded (4.8)	<10	4/4	5.3 ± 2.58	16.8 ± 4.3	0.6 ± 0.54	0.1 ± 0.02
		>10	1/0	3.4	10.7	5.3	0.5
Total		<10	23/11	7.4 ± 1.25	15.4 ± 1.71	0.4 ± 0.18	0.3 ± 0.09
		>10	14/2	5.7 ± 1.16	10.8 ± 1.63	6.9 ± 2.29	5.1 ± 1.44

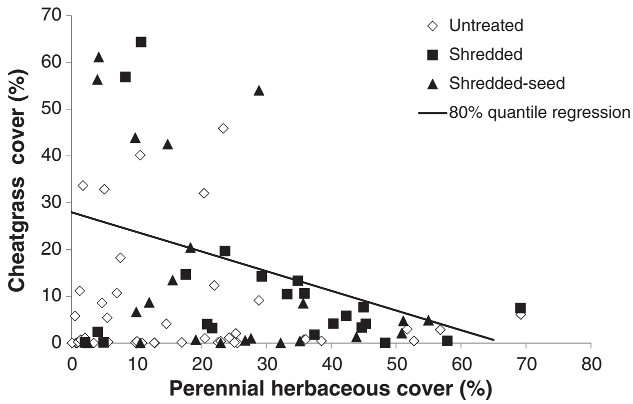

Figure 5. Mean cheatgrass cover in relation to total perennial herbaceous cover by site on untreated, shredded, and seeded-shredded treatments in Utah. Quantile regression line for the 80th percentile was significant at $P = 0.0012$ (intercept) and $P = 0.0134$ (slope).

et al. (2014) considered that increased tall perennial grass cover after tree reduction in the SageSTEP study was associated with increased growth of established plants because density had not increased. In contrast, on Utah shred study expansion sites, tall grass density increased on shredded compared with untreated plots.

Herbaceous vegetation response to shredding in our study was generally similar to that reported for cutting, chaining, and prescribed fire by 3 years after treatment as reported in other studies (Tausch and Tueller, 1977; Bates et al., 2000). In the SageSTEP study, perennial herbaceous cover initially decreased after prescribed fire, then increased to exceed that of untreated plots after 3 years (Miller et al., 2014). Prescribed fire also increased cheatgrass cover but to a greater magnitude than that of the mechanical treatments (Miller et al., 2014). In contrast, mastication in California chaparral increased non-native grass abundance much more than prescribed fire (Potts and Stephens, 2009). Longer-term response (13 years after treatment) to cutting of western juniper was dominated by perennial grasses in interspaces and, eventually, perennial grass replacement of annual grass dominance on former tree mounds (Bates et al., 2007).

Cheatgrass is most likely to dominate where soil temperatures support its growth and perennial bunchgrasses are limited (Chambers et al., 2014; Roundy et al., 2014a). Increasing perennial herbaceous understory growth and cover should help prevent dominance of cheatgrass by increasing resistance (Roundy et al., 2014a) through increased competitive advantage (Chambers et al., 2007). For our study, > 42% cover of perennial grasses and forbs limited cheatgrass cover to < 10% (see Fig. 5). Perennial grasses draw soil water and nitrogen from the same soil depth or resource growth pool as cheatgrass (Ryel et al., 2010; Leffler and Ryel, 2012). Therefore, perennial grasses may limit establishment, growth, and seed production of cheatgrass (Chambers et al., 2007). When managers note an absence of bunchgrass and presence or dominance of cheatgrass on sites being considered for tree reduction, they generally seed perennial species in conjunction with tree control. Our results that show decreased cheatgrass cover after seeding and shredding support this approach.

In the Utah shred study, seeding before tree shredding increased perennial herbaceous cover and decreased cheatgrass cover, especially when sites with higher tree cover were treated (see Figs. 3 and 4). Although seeding and shredding did not significantly ($P < 0.05$) increase perennial herbaceous cover more than shredding alone, it did decrease cheatgrass cover more than shredding alone at tree cover ≥ 55% (see Fig. 3). On tree climax sites, seeding and shredding were especially effective at increasing perennial grass cover compared with untreated areas (see Table 3, Fig. 4). Even though Utah expansion sites had higher maximum cheatgrass cover than tree climax sites on both untreated and seeded-shredded plots, the pattern of depressed cheatgrass cover from seeding was similar for both expansion and tree climax sites (see Fig. 3). Thick debris from mastication can suppress plant establishment (Kane et al., 2010; Young et al., 2013a). However, the masticated debris in piñon and juniper stands is often concentrated on tree litter mounds where juniper litter may limit seed-soil contact or seedling access to light even without masticated debris (Young et al., 2013a, b). Masticated debris from juniper shredding may also inhibit seedling emergence in interspaces where juniper litter is absent, but because it lengthens the time of soil water availability, it increases growth and biomass of emergent seedlings (Young et al., 2013a, b).

One of the main questions posed by managers is which kinds of sites are most likely to result in cheatgrass dominance after fuel treatments. In both the SageSTEP and Utah shred studies, a few sites had much more pretreatment and post-treatment cheatgrass cover than most of the other sites (Roundy et al., 2014a). In the SageSTEP study, Chambers et al. (2014) found that mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* [Rydb.] Beetle) sites encroached by piñon-juniper with frigid to cool-frigid soil temperature regimes had greater resistance to cheatgrass than warmer and drier Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle and Young) sites that were not tree-encroached. Cheatgrass was present in 9 of 11 mountain big sagebrush communities in the Utah shred study, but cover was limited. Five of the nine sites in the Utah shred study with minimal cheatgrass on untreated areas occurred at elevations above 2000 m. Greater growth of perennial competitors, as well as less adaptation of cheatgrass to cool temperatures, make higher-elevation sites more resistant to cheatgrass invasion and dominance (Chambers et al., 2014). Shredding trees may increase soil degree days in late spring by removing live tree shade (Young et al., 2013a). Therefore, shredding could potentially encourage cheatgrass establishment on some cooler, higher-elevation sites.

If cheatgrass cover is high before treatment, it will usually be high after treatment, unless perennial herbaceous cover is increased enough to depress it. In general, sites with a mesic soil temperature regime are considered more susceptible to cheatgrass than those with a frigid soil temperature regime (Chambers et al., 2014). Most of our sites in the Utah shred study were classified as having a mesic soil temperature regime, but many of those mesic sites were not dominated by cheatgrass before or after treatment (see Table 3). A general pattern of association that emerged from grouping study sites and treatments is that even sites with a mesic soil temperature regime had < 10% cheatgrass cover when tall grass cover was > 14% (see Table 3). Similarly, sites and treatments generally had < 10% cheatgrass cover when perennial herbaceous cover exceeded 42% (see Fig. 5). For the Utah shred study, untreated areas on only nine sites and treated areas on only four sites did not have cheatgrass present to some degree. This underscores the rapid spread of this non-native grass through human disturbances and other abiotic and biotic factors (Wisdom and Chambers, 2009). The presence of cheatgrass negatively impacts resilience, complicates restoration, alters fire return intervals, and ultimately creates biotic threshold conditions that are often irreversible without intensive management actions (Bagchi et al., 2013). Additional analyses are needed to better relate cheatgrass and perennial herbaceous cover to climatic and soil conditions on both the SageSTEP and Utah shred study sites.

Specific annual and perennial forbs are considered important for sage-grouse, and these preferred forbs were increased by prescribed fire and mechanical tree reduction in the SageSTEP study (Miller et al., 2014). Annual and perennial forbs were a limited component of the understory on Utah shred sites, although seeding did increase their cover by 2.4% when estimated across both ecological site types and tree cover ranges. Across all sites, sage-grouse forb cover ranged from 0% to 8% on the untreated, 0% to 13% on shredded, and 0% to 27% on the shredded and seeded areas with an average of 1.8%, 2.9%, and 4.2%, respectively. Due to the relatively low cover that occurred on most sites, we are unable to make strong inferences, although there are indications that seeding and shredding could increase sage-grouse forb cover on some sites.

Bare ground cover decreased in both shredded and seeded-shredded treatments. This decline can be attributed to an increase in plant cover (15—30% more than untreated) and the addition of shredded debris. As tree cover increased, there was a corresponding increase in shredded material after treatment. This debris may be especially important in reducing erosion at mid to high tree cover while understory cover reestablishes (Cline et al., 2010), thereby preventing the crossing of an abiotic threshold on highly erodible sites (Pierson et al., 2010, 2013; Williams et al., 2013). Miller et al. (2014) reported for the SageSTEP study that prescribed fire increased bare ground until the third year after treatment while mechanical (cutting) tree reduction decreased bare ground the second year after treatment. By the third year after treatment, there was no difference in bare ground among untreated, fire, and cut treatments.

Management Implications

We found that the best management to maintain both shrub and perennial herbaceous cover is to reduce trees at lower tree cover (<20%). Based on the current study and previous studies, which compared responses to prescribed fire and mechanical tree reduction (Chambers et al., 2014; Miller et al., 2014; Roundy et al., 2014a), we suggest that mechanical tree shredding may support resilience better than prescribed fire. When implemented before shrub cover is lost, it results in shrub and perennial herbaceous cover that is similar to communities before or at early phases of tree expansion. Maintaining shrub cover while increasing perennial herbaceous cover provides an array of ecological services but should also result in a trajectory toward at least as high resilience as the pre-encroached plant community. Shredding trees at higher tree cover (>40%) when shrub cover has decreased to 25% of maximum tends to promote a perennial herbaceous-dominated plant community or one dominated by cheatgrass on some sites. We found consistent tall grass and perennial herbaceous cover increases associated with cutting trees in the region-wide SageSTEP study and shredding trees in Utah on expansion or tree climax sites. This robust result suggests that managers can generally expect increased perennial herbaceous cover after mechanical tree reduction on a wide range of sites. However, on warmer sites that lack tall grass cover and are at risk for cheatgrass dominance, seeding before shredding, especially at high tree cover, may help restrict cheatgrass and reestablish perennial herbaceous cover. Although higher elevation and cooler sites tend to be most resistant to cheatgrass dominance, additional analysis may help determine other site-related factors that affect resistance. Shredding of trees is not generally recommended on old-growth tree climax sites, but shredding could be used on some of these sites to enhance resource values where infilling has resulted in loss of ecological function. Because tree shredding places canopy fuels on the ground, it does not prevent subsequent fire but allows easier containment (Young et al., 2015). One risk of tree shredding is that subsequent wildfire may burn hot and longer near the surface fuel bed and damage growing points of both woody and perennial herbaceous plants (Roundy et al., 2014a). Cool-season prescribed fire to target shred mounds but avoid shrubs and perennial herb patches is a potential solution (Bates and Svejcar, 2009; O'Connor et al., 2013). Mechanical tree reduction may require follow-up treatments to reduce trees that were not originally treated, sprouted after treatment, or established from seeds (Roundy et al., 2014a; Bristow et al., 2014).

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.rama.2016.01.007.

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