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Understory Vegetation Response to Mechanical Mastication of Piñon and Juniper Woodlands

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ABSTRACT

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Piñon and juniper encroachment and infilling can alter ecosystem processes and decrease resilience and resistance in sagebrush grasslands. Land managers employ a variety of techniques to eliminate these trees and mitigate their negative effects. Mechanical mastication or shredding is an increasingly popular method of removing these trees in Utah. It is a versatile treatment that can reduce canopy fuels, increase infiltration, and reduced sediment loss.

We compared vegetation cover for annual and perennial vegetation functional groups on shredded and adjacent unshredded areas across a range of sites. Our approach was to categorize sites by ecological site type (encroachment or tree) and subplots by treatment (untreated, shredded, and shredded-seeded) and initial tree cover. Mixed model analysis of covariance and the Tukey-Kramer test were used to determine significant differences among ecological site type and treatment combinations for each 5% increment of untreated or initial tree cover.

Shrub cover was unaffected by treatment and decreased with increasing tree cover. In general, perennial herbaceous understory cover increased after shredding to equal or exceed initial encroachment and infilling levels. This held true for both ecological site types and treatments, even at high pretreatment tree cover percentages. Cheatgrass also increased in cover after tree shredding although this trend was dampened in the seeded treatments indicating some suppression of cheatgrass by seeding. Shredding when there is high cover of perennial herbaceous plants and shrubs or seeding in conjunction with shredding where initial tree cover exceeds 35-40% will help discourage dominance by weeds.

Keywords: mechanical mastication, encroachment, infilling, sagebrush

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Introduction

Piñon (*Pinus spp.)* and juniper (*Juniperus* spp.) woodlands occupy >30 million ha in the western United States (West 1999). These woodlands are generally considered to be either tree climax (tree sites) or expansion woodlands (encroachment sites). Tree climax sites or persistent woodlands (Romme et al. 2009) typically have shallow (<0.5-m deep), rocky soils, trees >150 years old, (NRCS 1997) and support infrequent fire. In contrast, encroachment sites or wooded shrublands (Romme et al. 2009) typically have trees <150 years old and are associated with deeper, less rocky soils (NRCS 1997) where trees have encroached into sagebrush (*Artemisia spp.*) steppe. Tree encroachment and infilling is facilitated by fire suppression, overgrazing, and climate change that favors tree establishment (Miller et al. 2000; Miller and Tausch 2001; Tausch 1999a).

Sagebrush grasslands offer a multitude of ecosystem services that can be negatively impacted by increased tree establishment. These ecosystem services include wildlife habitat, soil stability, forage production, and biodiversity (Chambers et al. 2013b; Bestelmeyer and Briske 2012). Some wildlife species like the sage-grouse (*Centrocercus urophasianus*) wholly rely on sagebrush habitat for reproduction and survival (Connelly et al. 2004). Stressors such as piñon and juniper encroachment or infilling can alter resilience and put sites at risk to cross a biotic or abiotic threshold after which ecosystem services are diminished or lost.

Currently, managers are concerned about the risks associated with increased tree density and cover in some persistent woodlands, as well as that caused by encroachment and infilling in wooded shrublands (Page et al. 2013). As tree density and cover increase over time, canopy fuel loads also increase while cover and production of desirable understory shrubs, grasses and forbs decrease (Archer et al. 2011; Miller and Rose 1999; Young et al. 2013c). Increased fuel loads may lead to high intensity, fast spreading crown fires (Gruell 1999; Tausch 1999 a, b). Reduced understory cover results in an increase in erosion on some sites (Aldrich et al. 2005; Miller et al. 2005; Pierson et al. 2010). When tree encroachment occurs on a large scale there is a loss in landscape heterogeneity, wildlife habitat, and watershed function (Miller et al. 2005). As tree cover increases, loss of perennial herbaceous cover, erosion, and high intensity fires may cause the sagebrush ecosystem to pass biotic or abiotic thresholds into an alternative state of weed dominance and recurrent fire (D'Antonio and Vitousek 1992; Miller and Tausch 2001). The magnitude of change can be so great that ecosystem processes are altered and ecosystem services are diminished to where it is difficult if not impossible to restore them (Bagchi et al. 2013; Chambers et al. 2013b).

In practice, avoiding a threshold is best accomplished by maintaining or creating a resilient ecological state (Bestelmeyer and Briske 2012). As tree infilling proceeds, response to tree reduction treatments becomes a test of both site resilience and resistance. High resilience after tree reduction is indicated by a return to similar shrub and perennial herbaceous cover as was present early on in encroachment and infilling. High resistance would be indicated by lack of a transition to weed dominance after tree reduction.

To mitigate the effects of encroachment and infilling, land managers employ prescribed fire and a variety of mechanical tree reduction techniques such as cutting or mastication (shredding) using a toothed drum (Cline et al. 2010). Shredding is easier to implement than prescribed fire. It can be selectively implemented (e.g., thinning, clear-cutting, or mosaics) almost any time of year when the surface soil is dry. Debris from shredding increases infiltration and reduces sediment production on some microsites (Cline et al. 2010). Unlike cutting, shredding converts large canopy fuels to smaller 1 and 10-hour fuels, which can greatly reduce fire spread and facilitate containment (Young et al. 2013c).

To help managers decide at which stage of infilling to apply treatments wooded shrublands have been categorized into phases (Johnson and Miller 2006). Shrubs and herbaceous perennials dominate Phase I communities with minimal tree cover. In Phase II, trees and shrubs/perennial herbaceous plants are co-dominant. At Phase III trees are dominant and often form near-closed canopy stands with sparse perennial herbaceous and shrub cover. Effects of infilling, such as decreased shrub and perennial herbaceous cover, generally become evident at Phase II (Roundy et al. 2013a). Piñon and juniper trees have dense lateral root systems and well-developed tap roots that allow them to reduce the availability of soil water and nutrients for understory shrubs, forbs, and grasses (Krimer et al. 1996; Leffler and Ryel 2012; Rau et al. 2011; Roundy et al. 2013b; Ryel et al. 2010; Young et al. 2013a). Lack of perennial understory cover at higher encroachment phases leads managers to seed some sites to avoid dominance by invasive weeds.

The wide-spread use of shredding across Utah prompts questions relative to vegetation response: 1) How do responses vary for tree and encroachment ecological sites? 2) What is the effect of initial tree cover (degree of infilling) on response? 3) How does seeding of shredded treatments affect response? Ours is a retrospective study to determine the effects of ecological site type (ES), initial tree cover (TC), and treatment (TRT- untreated, shredded-not seeded, shredded-seeded) on vegetation response.

Methods

Our approach was to compare vegetation variables on shredded and adjacent unshredded areas across a range of sites. We used pretreatment National Agriculture Imagery Program

(NAIP) imagery to select sample subplots to compare vegetation across similar low to high ranges of tree cover for both untreated and treated areas.

Study sites

Study sites are 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 Forest Service (USFS) (Figure 1). Our study sites encompass a range of time since treatment (1-8 years) and two ecological site types, as determined by NRCS (1997) criteria (encroachment or tree sites), and range in untreated and pretreatment tree cover from 2 to 90% (Table 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). 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 areas on the same ecological sites on each of 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. 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. 2013) on the potential subplots. We randomly selected 3 subplots each on untreated and treated areas 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 6 (1 tree cover category x 2 treatments-untreated and shredded x 3 subplots each= 6) to 18 (3 tree cover categories x 2 treatments x 3 subplots each= 18). The only exception to this sampling scheme was for three sites originally treated and measured in a previous study known as SageSTEP (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.

Measurements

Vegetation measurements on each 30 by 33-m subplot were made according to the protocol of McIver et al. (2010) and Miller et al. (2013). We used the line-point intercept method to measure cover by species on 5 30-m transects on each subplot. Transects were located at the 2, 7, 15, 23, 28-m marks. We dropped a pin flag every 0.5-m starting at the 0-m mark (60 points/transect x 5 transects/subplot= 300 points/subplot). At each point, we recorded canopy and/or foliar hits on vegetation as well as the ground surface code. 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 functional groups. Ground surface codes included soil, rock, biotic crust, bedrock, moss, duff, and embedded litter. To calculate cover for a subplot we summed all of the hits on a particular species or functional group and divided by 300.

We also counted density of sagebrush seedlings and juveniles (< 5 cm height) in 0.25-m^2 quadrats placed every odd numbered m on the 7, 15, 23-m transects (15 quadrats/transect x 3 transects/subplot = 45 quadrats/subplot).

Analysis

We used mixed model analysis of covariance (Proc Glimmix, SAS v9.3, SAS Institute, Inc., Cary, NC) to compare responses of functional groups. These cover groups included total shrubs, sagebrush, tall, short, and total perennial grass (tall, short, and rhizomatous grasses), cheatgrass, perennial forbs, sage grouse food (Connelly et al. 2004; Nelle et al. 2000; Pyle and Crawford 1996; Rhodes et al. 2010), and annual forb, total perennial herbaceous (total perennial grass plus perennial forb), biotic crust and bare ground. We initially wanted to determine effect of years-since-treatment on responses. In a preliminary analysis, years since treatment was tested as a covariate with ecological site type (ES- encroached or tree) and treatment (TRT- Untreated, shredded-not seeded, shredded-seeded) included as fixed factors. Site within ES and TRT within site were considered random. However, we dropped years-since-treatment from the model for a number of reasons. First, it was not possible to select many similar sites with different years since treatment. This confounds site and site potential with years-since-treatment in the analysis. Second, the years-since-treatment covariate was only significant ($P < 0.05$) for one response variable. While the years-since-treatment covariate interaction with treatment was significant for two response variables, graphing of data indicated that these responses were more associated with site differences than consistent years-since-treatment patterns. Subsequent analysis included only ES and TRT as fixed factors. Untreated and pretreatment tree cover (TC), estimated from NAIP imagery was analyzed as a covariate (Roundy et al. 2013a). Site was considered random and subplots (557 total across 44 sites) were treated as subsamples. 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. 2013a). We adjusted for false positives from multiple comparisons by using a $P < 0.01$. Vegetation response cover data were normalized by the arcsin squareroot transformation, but tree cover covariate data were not transformed. Observation of residual plots indicated that assumptions were met for analysis of covariance.

Results

Functional group response varied in significance in the mixed model analysis according to ecological site type, treatment, and untreated or pretreatment tree cover as the covariate (Table 2).

How did functional group cover vary for tree and encroachment ecological sites?

Ecological site type was significant ($P< 0.05$) for three of the twelve functional groups, sagebrush, short, and tall perennial grass cover (Table 2). Sagebrush cover was slightly higher on encroachment than tree sites across all treatments (0.3%, Figure 2). Encroachment sites had 5% more Sandberg bluegrass (*Poa secunda* J. Presl) and 4.5% more total perennial grass cover than tree sites across all treatments and tree cover ranges (Figure 3).

What is the effect of initial tree cover (degree of infilling) on functional group cover?

Total shrub and sagebrush cover significantly decreased with increasing tree cover across all treatments, with shrub cover approaching zero between 60-80% TC (Figure 2). 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 tree cover reached 80% on tree sites and only until it reached 70% on encroachment sites. Perennial and sage grouse forb cover were low for all treatments <6.2 and <8%) these variables were not significantly ($P>0.05$) related to tree cover (Figure 2). There was a significant decrease in short grass cover with increasing tree cover across both ecological site types and treatments (Table 2, Figure 3). For tall and total perennial grass cover, TC was not significant ($P>0.05$), but the interaction of ES and TC was (Figure 3). Tree sites had higher tall and total perennial grass cover at high TC than encroachment sites on shredded plots (Figure 3). Bare ground significantly decreased ($P<0.001$) with increasing tree cover (Table 2, Figure 4).

While TC was not significant (P>0.05) for total perennial herbaceous cover, the interaction of TC and ES was (Table 2, Figure 4). For treated plots, encroachment sites had higher total perennial herbaceous cover at low TC but tree sites had higher total perennial herbaceous cover at high TC. Biotic crust and annual exotic forb cover were limited and not significantly (P>0.05) associated with TC (Table 2, Figure 4).

How does seeding of shredded treatments affect response?

Total shrub or sagebrush cover was not affected by shred or shred and seed treatments (Table 2, Figure 2). Shredding alone did not significantly (P>0.05) 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 tree cover (Figure 2). Neither shredding or shredding and seeding increased sage grouse forb cover on encroachment sites (Figure 2). Treatments did not significantly affect short grass cover (Table 2, Figure 3). Shredding with and without seeding increased tall grass cover on both encroachment and tree sites (Figure 3). However, the ES by TRT interaction was marginally significant (P<0.0731) for tall grass and significant (P<0.0298) for total perennial grass cover (Table 2). Also, the TRT by TC interaction was significant (P<0.0004) for these responses (Table 2). On encroachment sites, shredding increased tall and total perennial grass cover across the range of tree cover (Figure 3). While shredding increased tall and total perennial grass cover most at high tree cover, perennial grass cover still decreased with increasing tree cover. In contrast, on tree sites, shredding had little effect at low initial tree cover, but increased tall and total perennial grass cover with increasing pretreatment tree cover (Figure 3, Table 3). Shredding and seeding showed a similar pattern by increasing tall and perennial grass cover as initial tree cover increased for both encroachment and tree site types. However, shredding and seeding increased tall and total perennial grass cover most on tree sites and seeding after shredding was most effective at high initial tree cover (Figure 3, Table 3).

The TRT by TC interaction was significant for cheatgrass cover (Table 2, Figure 3). Shredding increased cheatgrass cover with increasing initial tree cover for both ecological site types (Figure 3). Shredding without seeding significantly ($P < 0.05$) increased cheatgrass cover compared to no treatment at 30-90% initial tree cover while seeding after shredding increased cheatgrass cover at 40-75% initial tree cover (Figure 3, Table 3). Cheatgrass cover varied widely across the study sites (Figure 5). A few sites had > 18% cheatgrass cover (6 of 44 sites for untreated plots; 9 of 44 sites for shredded or shredded-seeded plots). Plotting site by treatment means indicated that sites with > 35-40% perennial herbaceous cover had < 10% cheatgrass cover (Figure 5). Nevertheless, there were some subplots with both high perennial herbaceous and cheatgrass cover.

Shredding decreased bare ground from 10-70% initial tree cover, while seeding after shredding decreased bare ground compared to no treatment from 15-90% initial tree cover (Table 3, Figure 4). Shredding increased total perennial herbaceous cover up to 12% on encroachment and 23% on tree sites (Figure 4). Untreated tree sites had less total perennial herbaceous cover than encroached sites at low tree cover, but treated tree sites still had a very positive response to shredding, increasing at 0-90% initial tree cover (Table 3). Seeding after shredding produced a positive response from 35-90% initial tree cover with a maximum increase of 24% on encroachment sites, and from 15-90% initial tree cover and a maximum of 31% on tree sites (Figure 4, Table 3). Biotic crust cover was limited on our study sites and did not differ with treatment (Figure 4). Shredding increased annual forb cover at mid to higher tree cover even with seeding (Figure 4, Table 3). Annual forb cover was limited (<10%) and especially low on tree sites.

Sagebrush seedlings

We observed sagebrush seedlings on 61% of the 44 study sites. For these sites, the number of seedlings m^2 was 0.7 ±0.15 (n=14) on untreated plots, 5.1 ±1.8 (n=16) on shredded not- seeded plots, and 6.8 ±4.3 (n=9) on shredded-seeded plots.

Discussion

Sagebrush is less responsive than perennial herbs after tree infilling and subsequent tree reduction. Although shredding did not reduce total shrub or sagebrush cover, cover of the shrub component has not significantly increased across our sites with tree shredding (Fig. 2). We did observe greater shrub twig growth on treated compared to untreated areas on some of our sites. Since shrub cover is slow to respond to tree reduction, and shrub cover decreases with increasing tree cover, maintenance of higher shrub cover dictates reducing trees at low to mid phases of encroachment and infilling. As shrub cover decreases with increasing TC, biodiversity and quality of wildlife habitat are compromised (Huber et al. 1999; Miller et al. 2005). A potential tradeoff of treating at a low TC to maintain shrub cover is that there are fewer trees so fewer soil water resources are made available by tree removal (Roundy et al. 2013b), resulting in only a slight increase in desirable understory cover. Conversely, treating at low tree cover and where perennial herbaceous cover is high provides fewer resources to cheatgrass, and may increase resistance to weed dominance (Chambers et al. 2013b; Davis et al. 2000) (Figure 5). With lack of fire, trees have been encroaching and infilling for more than 150 years resulting in significant extent of land in the mid to high TC range (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 (Bates et al. 2005; Ziegenhagen 2004). Sagebrush seedlings established on a 61% of our 44 sites, indicating recovery potential where there is a seed source.

Perennial forbs were a limited component of the understory, though seeding did increase their cover on both ecological sites and across all tree cover. Cover of sage-grouse forbs ranged from 0-8% on the untreated, 0-13% on treated non-seeded, and 0-27% on the seeded subplots 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 about effects of tree shredding on sage-grouse forbs.

In general, our study sites exhibited high perennial herbaceous recovery after tree shredding. For example, 50% of perennial herbaceous cover was lost at tree cover >45% on encroachment sites and >55% on tree sites. Tall grass cover, especially, and perennial herbaceous cover generally, increased even at high initial TC after tree shredding to similar or greater cover than that on untreated areas at low TC (Figures 3 and 4). This greater cover than at low tree cover on untreated plots may have been associated with greater resource availability to perennial herbs. Where shredding reduces trees at high TC, shrubs are lacking so recovering perennial grasses may have more resources available to them than at low tree cover where shrub cover is much higher. Shredding of Phase III wooded shrublands benefits grass seedling establishment by increasing the nitrogen supply rate and increasing the time of available water in spring (Young et al. 2013a, b; in review). Increased perennial herbaceous cover after tree shredding is associated with increased time of available water (Roundy et al. 2013b) and accelerated growth of residual species (Tausch and Tueller 1977). Miller et al. (2013) considered that perennial grass recovery of prescribed fire and tree cutting was mainly associated with already-established residual grass plants rather than seedlings. It is possible that increasing perennial understory growth and cover could help prevent invasion of cheatgrass by increasing resilience (Roundy et al. 2013a) through increased competitive advantage (Chambers et al. 2007). Nevertheless, residual species may not recover fast enough to use all of the soil water made available by reducing trees at high initial TC, potentially leaving open resources for cheatgrass (Roundy et al. 2013b).

Shredding increased cheatgrass cover with increasing initial tree cover for both ecological site types. This can be explained by an increase in the amount of resources made available by removing the trees. The more trees present on a site the more resources that are made available once the trees are reduced (Roundy et al. 2013b). When there is a surge of unused resources a plant community becomes more susceptible to invasion (Davis et al. 2000). Thus any increase in the availability of resources, whether by increased precipitation during a wet year, decreased water use from tree reduction, or both can increase the susceptibility of a community to invasion (Davis et al. 2000). Although cheatgrass was highly variable across our sites in relation to perennial herbaceous cover, cheatgrass cover was limited to < 10% when perennial herbaceous cover was > 35-40% on a site (Figure 5).

In addition to resource availability there are ecophysiological constraints on cheatgrass invasion and dominance. At high elevations, growth and reproduction of cheatgrass is limited by cool temperatures (Chambers et al. 2007). Spatial and temporal variations in soil water limit establishment at low elevations (Chambers et al. 2007). Hence, Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young) communities that fall in between these elevations may be less resistant to cheatgrass invasion than high elevation mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle) and mountain shrub communities (Wisdom and Chambers 2009). However, at our sites, cheatgrass was often present in mountain big sagebrush communities (only 2 of 11 mountain big sagebrush communities did not have cheatgrass). Chambers et al. (2013a) found that on Wyoming and piñon-juniper sites environmental condition was a greater factor than the subspecies of sagebrush in defining whether or not a site was resistant to cheatgrass. Our sites did follow the pattern of less cheatgrass on our high elevation sites (5 of the 9 sites with no cheatgrass on untreated plots occurred at elevations above 2000-m).

The presence of cheatgrass negatively impacts resilience, complicates restoration, alters fire return intervals, and ultimately creates threshold conditions that are often irreversible without intensive management actions (Bagchi et al. 2013). Only 9 of our untreated and 4 of our treated 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). On the other hand, only 6 untreated and 9 treated sites had > 18% cheatgrass cover, suggesting that some sites are much more susceptible to cheatgrass dominance than others (Figure 5). It is important to note that 5 of the 6 sites that had high cheatgrass cover on untreated plots were also sites that had high cheatgrass cover on treated plots.

Bare ground decreased in both shredded and shred and seeded 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 increases there was a corresponding increase in shredded material. This debris may be especially important in reducing potential erosion at mid-high TC, while understory cover reestablishes (Cline et al. 2010), thereby, preventing crossing an abiotic threshold on highly erodible sites. However, the shredded material creates a warmer and wetter soil environment in the spring that is favorable to both cheatgrass and bluebunch wheatgrass

(*Pseudoroegneria spicata* (Pursh) Á. Löve) seedling establishment (Chambers et al. 2007; Roundy et al. 2013b; Young et al. 2013a).

Shredding produces somewhat similar results to cut and drop, prescribed fire, and chaining for most functional groups over time. Like shredding, cutting did not affect shrub or sagebrush cover (Miller et al. 2013). Shrubs are slow to respond to chaining, taking 5 or more years to recover (Tausch and Tueller 1977). Prescribed fire exhibited a significant decrease in shrub cover post treatment and did not experience substantial recovery after 3 years (Miller et al. 2013). Over the short term (up to 3 years after treatment) shredding and cutting trees increased desirable perennial plant cover of Phase II and Phase III wooded shrublands on 4 sites in Utah (Roundy et al. 2013a). Chaining similarly increased perennial herbaceous understory cover 2-4 years post treatment (Tausch and Tueller 1977). After declining the first year post fire, perennial herbaceous cover increased to above the untreated control after 3 years (Miller et al. 2013). However, cutting and shredding also increased cover of invasive cheatgrass on some sites, especially when treatments were implemented at Phase II and Phase III encroachment. Prescribed fire also increased cheatgrass cover above that of the mechanical treatments (Miller et al. 2013).

Conclusion

Management that retains high density and cover of perennial plants best resists cheatgrass dominance by reducing available soil water in the resource growth pool that cheatgrass depends on for growth and seed production (Roundy et al. 2013b). A majority of our sites (35 of 44) had cheatgrass in the untreated areas underscoring the importance of timing shredding to maintain residual perennial cover or seeding to avoid cheatgrass dominance. The understory component is especially important when tree cover is high and there is a lack of shrubs as well as limited perennial herbaceous cover. At this point the best option to restoring a functional plant community and resist cheatgrass invasion is to seed.

Intense disturbances, either natural or human-caused, can alter the ecological processes of an ecosystem and decrease both resistance and resilience. Ecosystems with low resistance and resilience are in jeopardy of crossing thresholds into alterative states such as cheatgrass dominance and associated high-frequency fire (Wisdom and Chambers 2009). We consider that the best management to avoid this degradation pathway on both encroachment and infilling tree sites is to either reduce trees at low TC or to seed if the TC exceeds 35-40%. Appropriate management of sagebrush grasslands and piñon-juniper woodlands can help to maintain or improve resistance and resilience of the ecosystem.

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APPENDIX

Table 1. Characteristics of 44 sites across Utah. ES-Ecological Site: E=Encroachment, T=Tree. TRT=Treatment: NS=Shredded not seeded, S=Shredded-seeded. TC=Tree Cover. PP=Physiographic province: GB=Great Basin, CP=Colorado Plateau

Site Name	ES	TRT	TC (%)	PP	Vegetation Type	Site Name	ES	TRT	TC (%)	PP	Vegetation Type
Anderson Mountain	E	S	6-67	GB	Mountain Big Sage/ Squirreltail	Natural Bridges PJ	T	NS	20-40	CP	Mormon Tea-Wyoming Big Sage/ Indian Ricegrass
Augusi	T	NS	65-78	CP	Utah Service Berry-Mountain Mahogany/ Bluebunch Wheatgrass-Squirreltail	Natural Bridges SB	T	NS	16-41	CP	Wyoming Big Sage-Mormon Tea-Narrowleaf Yucca
Black Dragon	E	S	3-25	CP	Utah Service Berry-Mountain Big Sage/ Western Wheatgrass-Crested Wheatgrass	North Grouse Creek	T	NS	32-49	GB	Wyoming Big Sage/Bluebunch Wheatgrass-Sandberg Bluegrass
Blue Valley	E	S	2-63	GB	Mountain Big Sage/Sandberg Bluegrass-Muttongrass	Onaqui	E	NS	3-32	GB	Wyoming Big Sage/Bluebunch Wheatgrass
Bowery ARTR	T	S	43-63	CP	Mountian Big Sage/ Squirreltail	Ray Mesa II East	E	NS	2-90	CP	Little Sage-Utah Service Berry/ Sandberg Bluegrass
Bowery Springs	T	S	58-63	CP	Mountain Big Sage/ Squirreltail	Sand Hollow	E	S	34-62	GB	Mountain Big Sage/Squirreltail-Sandberg Bluegrass
Chokecherry	E	NS	4-43	GB	Wyoming Big Sage/Bluebunch Wheatgrass-Sandberg Bluegrass	Scipio	E	NS	6-46	GB	Wyoming Big Sage/Bluebunch Wheatgrass
Columbia	T	S	48-69	CP	Singleleaf Ash/Indian Ricegrass	Sharpes Valley	E	NS	11-69	GB	Wyoming Big Sage-Antelope Bitterbrush/ Bluebunch Wheatgrass- Sandberg Bluegrass
Cook Canyon	T	S	21-58	GB	Wyoming Big Sage-Antelope Bitterbrush/ Bluebunch Wheatgrass-Sandberg Bluegrass	South Beaver	E	S	5-52	GB	Mountain Big Sage-Antelope Bitterbrush/ Indian Ricegrass-Crested Wheatgrass
Eight Mile Bench	E	S	50-59	GB	Utah Service Berry-Sonoran Scrub Oak/ Squirreltail	South Creek	T	S	20-68	CP	Mountain Big Sage-Antelope Bitterbrush/ Blue Grama-Squirreltail
Goslin	T	S	7-30	CP	Mountain Big Sage/Bluebunch Wheatgrass-Muttongrass	South Hills	E	S	7-18	GB	Mountain Big Sage/Indian Ricegrass-Sandberg Bluegrass
Government Creek	E	S	21-39	GB	Wyoming Big Sage/Bluebunch Wheatgrass-Sandberg Bluegrass	Steinaker	T	NS	8-48	CP	Wyoming Big Sage/ Bluebunch Wheatgrass-Sandberg Bluegrass
Grantsville	E	S	2-35	GB	Wyoming Big Sage/Bluebunch Wheatgrass-Sandberg Bluegrass	Terra East	E	NS	40-56	GB	Stansbury Cliffrose/Sandberg Bluegrass
Greenville	E	NS	2-56	GB	Wyoming Big Sage/Needle and Thread-Bluebunch Wheatgrass	Unit B	E	S	54-75	GB	Mountain Big Sage/Squirreltail-Sandberg Bluegrass
Hiawatha	T	S	5-34	CP	Wyoming Big Sage/Blue Grama-Squirreltail-Indain Ricegrass	West Carbon	T	S	38-51	CP	Black Sage/Crested Wheatgrass
Highway 56 ARNO	E	S	4-31	GB	Black Sage/Needle and Thread-James' Galleta	West Oak Brush	E	NS	4-33	GB	Wyoming Big Sage/ Bluebunch Wheatgrass-Sandberg Bluegrass-Crested Wheatgrass
Highway 56 ARTR	E	S	6-14	GB	Wyoming Big Sage/James' Galleta-Needle and Thread	West Onaqui ARNO	E	NS	35-51	GB	Black Sage/Bluebunch Wheatgrass-Sandberg Bluegrass
Hyatt Springs	E	NS	15-22	GB	Wyoming Big Sage-Utah Service Berry/ Squirreltail	West Onaqui ARTR	E	NS	2-57	GB	Wyoming Big Sage/ Bluebunch Wheatgrass-Sandberg Bluegrass
Indian Springs	T	S	16-84	CP	Utah Service Berry-Gambel Oak-Mountain Mahogany/Kentucky Bluegrass	West Onaqui High	E	NS	30-42	GB	Stansbury Cliffrose-Antelope Bitterbrush/ Bluebunch Wheatgrass-Sandberg Bluegrass
James Ranch	E	NS	3-25	GB	Wyoming Big Sage/Bluebunch Wheatgrass-Sandberg Bluegrass	West Onaqui Mid	E	NS	18-40	GB	Stansbury Cliffrose-Antelope Bitterbrush/ Bluebunch Wheatgrass-Sandberg Bluegrass
Keg Springs	T	NS	7-40	GB	Black Sage/Bluebunch Wheatgrass	Winter Springs Lower	E	NS	4-45	GB	Wyoming Big Sage/ Bluebunch Wheatgrass-Sandberg Bluegrass
Muddy Creek	E	S	26-61	CP	Antelope Bitterbrush-Mormon Tea/Squirreltail	Winter Springs Upper	E	NS	49-65	GB	Wyoming Big Sage-Black Sage/Bluebunch Wheatgrass-Sandberg Bluegrass

Table 2. F-significance for mixed model analysis of ecological site type (encroached, tree), treatment (untreated, shredded-not seeded, shredded-seeded) in relation to pretreatment tree cover as a covariate for different cover response variables across 44 piñon and juniper sites in Utah.

	Total shrub	Sagebrush	Perennial forb	Sage grouse forb	Short grass	Tall grass	Total perennial grass	Cheatgrass	Bare ground	Total perennial herbaceous	Biotic crusts	Annual forbs
Ecological site (ES)	0.290	**0.033**	0.900	0.819	**0.034**	0.204	**0.028**	0.111	0.065	0.061	0.347	0.076
Treatment (TRT)	0.766	0.093	**0.045**	0.334	0.646	0.432	0.828	0.579	**0.021**	0.714	0.449	0.382
ES*TRT	0.521	0.562	0.596	0.399	0.694	0.073	**0.032**	0.338	0.474	**0.036**	0.523	**0.010**
Tree cover (TC)	**<0.0001**	**<0.0001**	0.784	0.086	**0.026**	0.891	0.662	0.370	**<0.0001**	0.485	0.300	0.186
TC*ES	**0.007**	**0.000**	0.107	0.285	0.601	**0.022**	**0.030**	0.726	0.880	**0.038**	0.146	0.770
TC*TRT	0.689	0.922	0.184	0.634	0.856	**<0.0001**	**<0.0001**	**0.000**	**0.023**	**<0.0001**	0.506	**0.006**
TC*ES*TRT	0.365	0.580	0.769	**0.043**	0.936	0.059	0.389	0.279	0.780	0.354	0.136	**0.003**

Table 3. Pretreatment tree cover ranges over which cover responses of shredded-not seeded (NS) or shredded-seeded (S) treatments compared to untreated (UT) either were similar or differed from adjacent untreated plots for encroached and tree ecological site types in Utah.

		Tree cover range (%)	
Variable	Response	Encroached	Tree
Total shrub	NS=UT	0-90	0-90
	S=UT	0-90	0-90
Sagebrush	NS=UT	0-90	0-90
	S=UT	0-90	0-90
Perennial forb	NS=UT	0-90	0-90
	S=UT	0-90	0-90
Sage grouse forb	NS=UT	0-90	0-90
	S=UT	0-90	0-90
Short grass	NS=UT	0-90	0-90
	S=UT	0-90	0-90
Tall grass	NS>UT	15-90	30-90
	S>UT	30-90	20-90
Perennial grass	NS>UT	25-90	NS=UT
	S>UT	45-90	40-90
Cheatgrass	NS>UT	35-90	NS=UT
	S>UT	25-90	S=UT
Bare ground	NS<UT	10-70	10-70
	S<UT	15-90	15-90
Perennial herbaceous	NS>UT	15-90	0-90
	S>UT	35-90	15-90
Biotic crusts	NS=UT	0-90	0-90
	S=UT	0-90	0-90
Annual forbs	NS>UT	50-90	NS=UT
	S>UT	65-90	S=UT

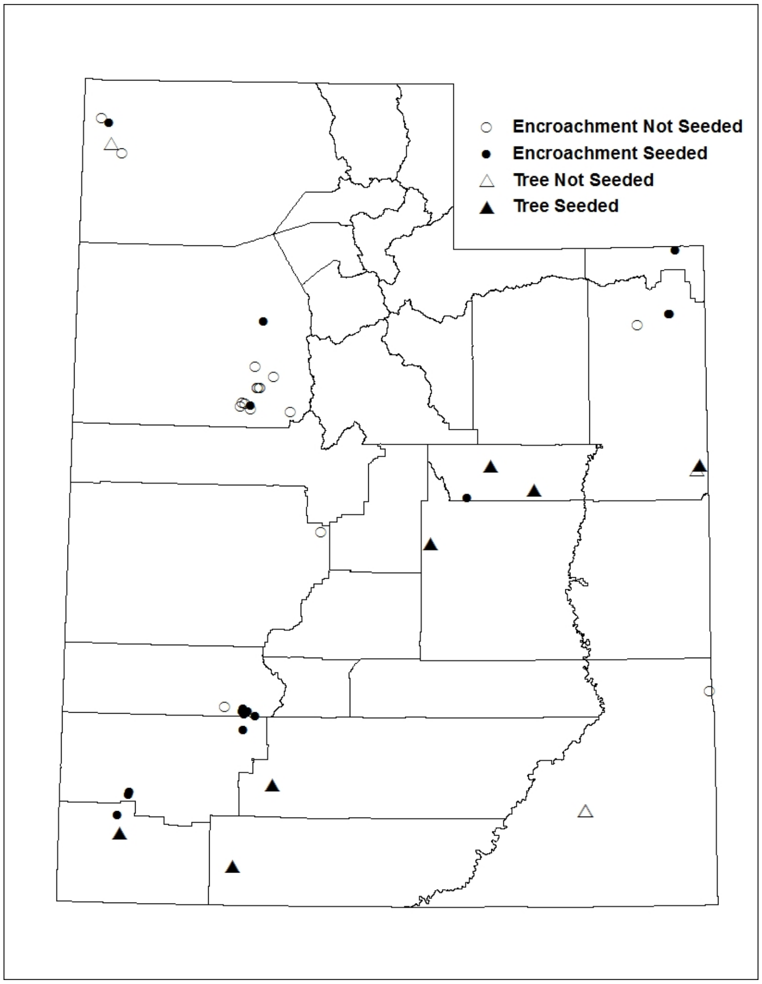

Figure 1. Location of 44 research sites by ecological site and treatment.

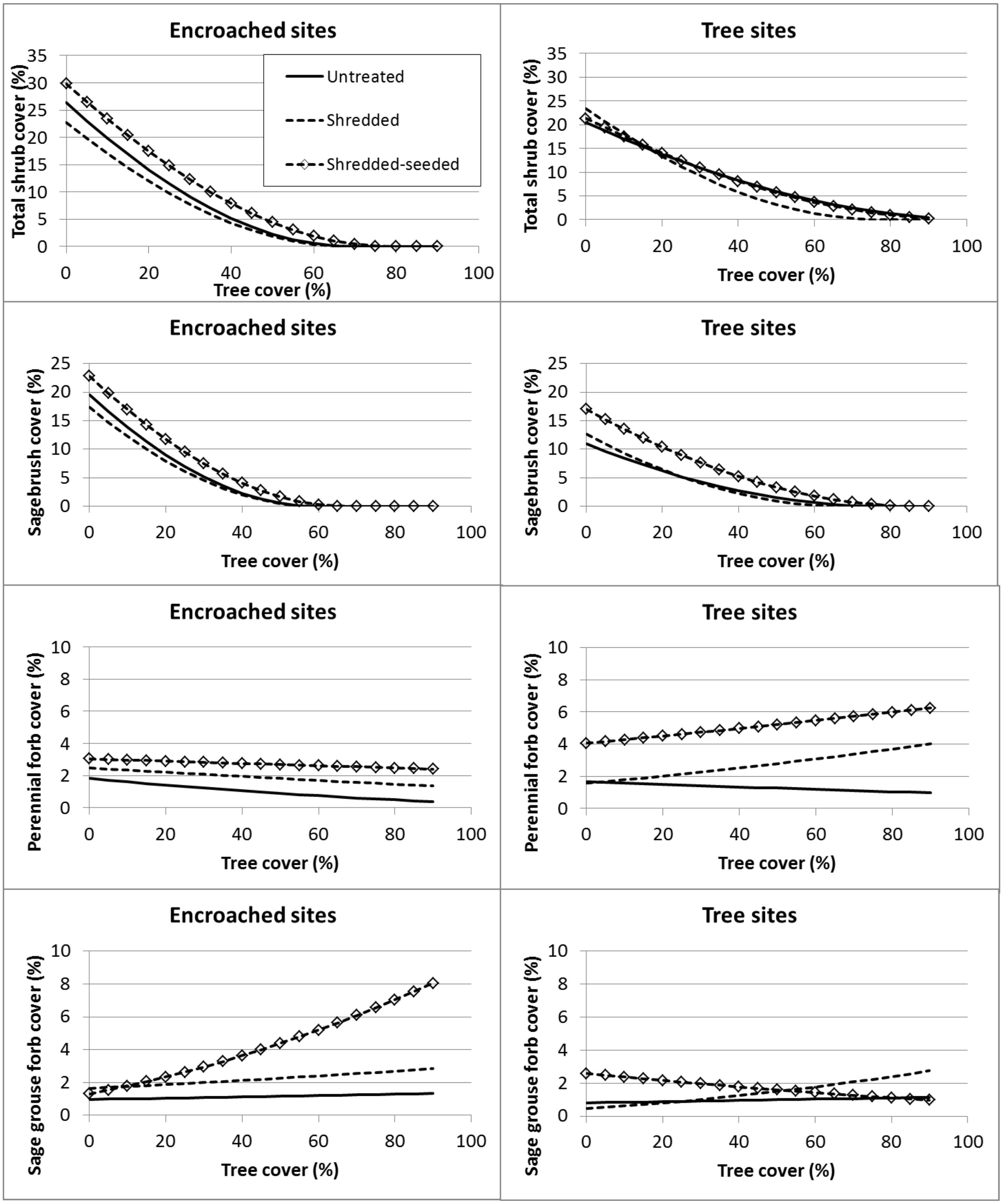

Figure 2. Cover of shrubs and forbs for piñon and juniper sites encroached into shrublands (left) and for tree sites (right) in relation to pretreatment tree cover on untreated, tree-shredded, and tree-shredded-seeded treatments in Utah. See table 2 for significant differences.

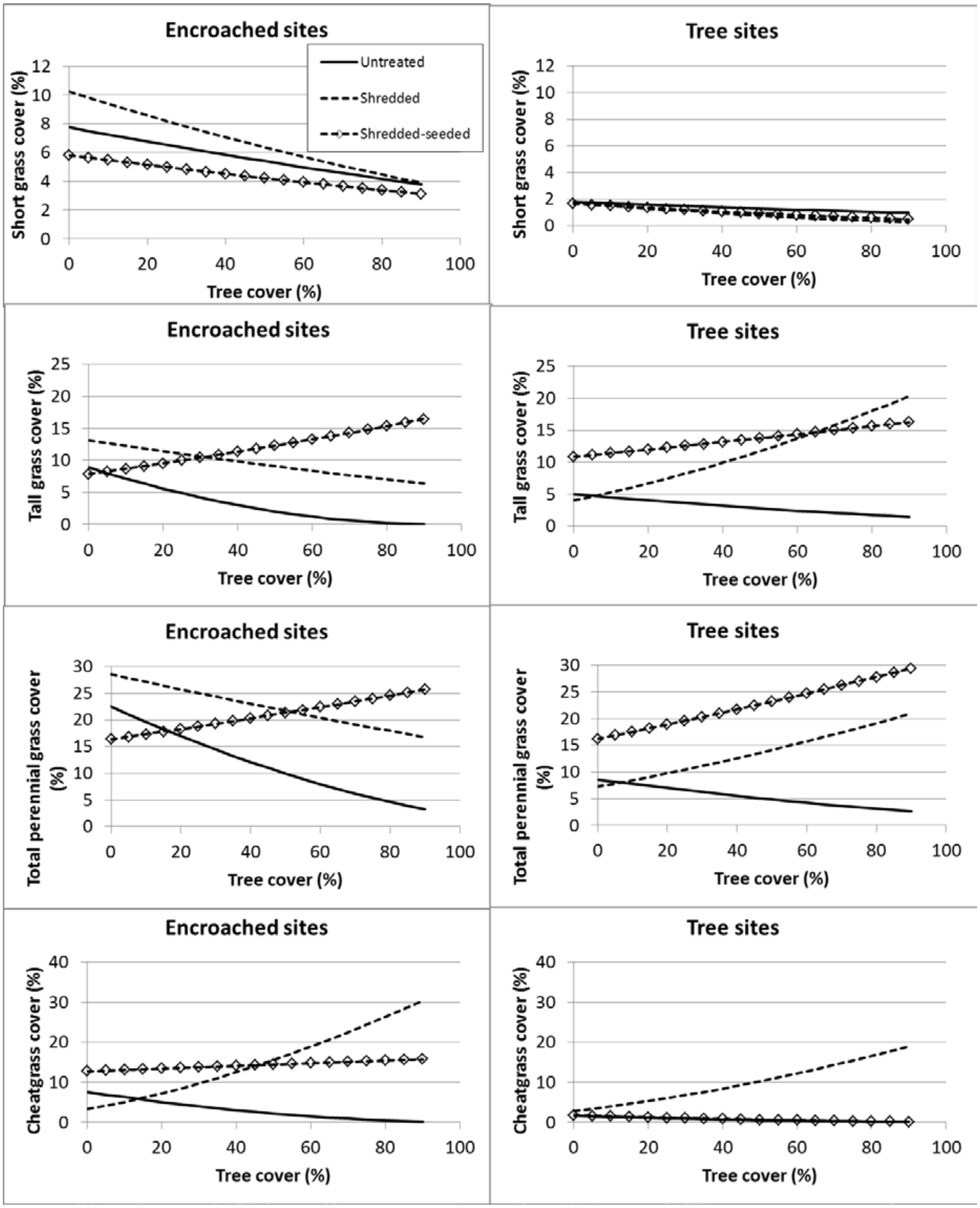

Figure 3. Cover of grasses for piñon and juniper sites encroached into shrublands (left) and for tree sites (right) in relation to pretreatment tree cover on untreated, tree-shredded, and tree-shredded-seeded treatments in Utah. See table 2 for significant differences.

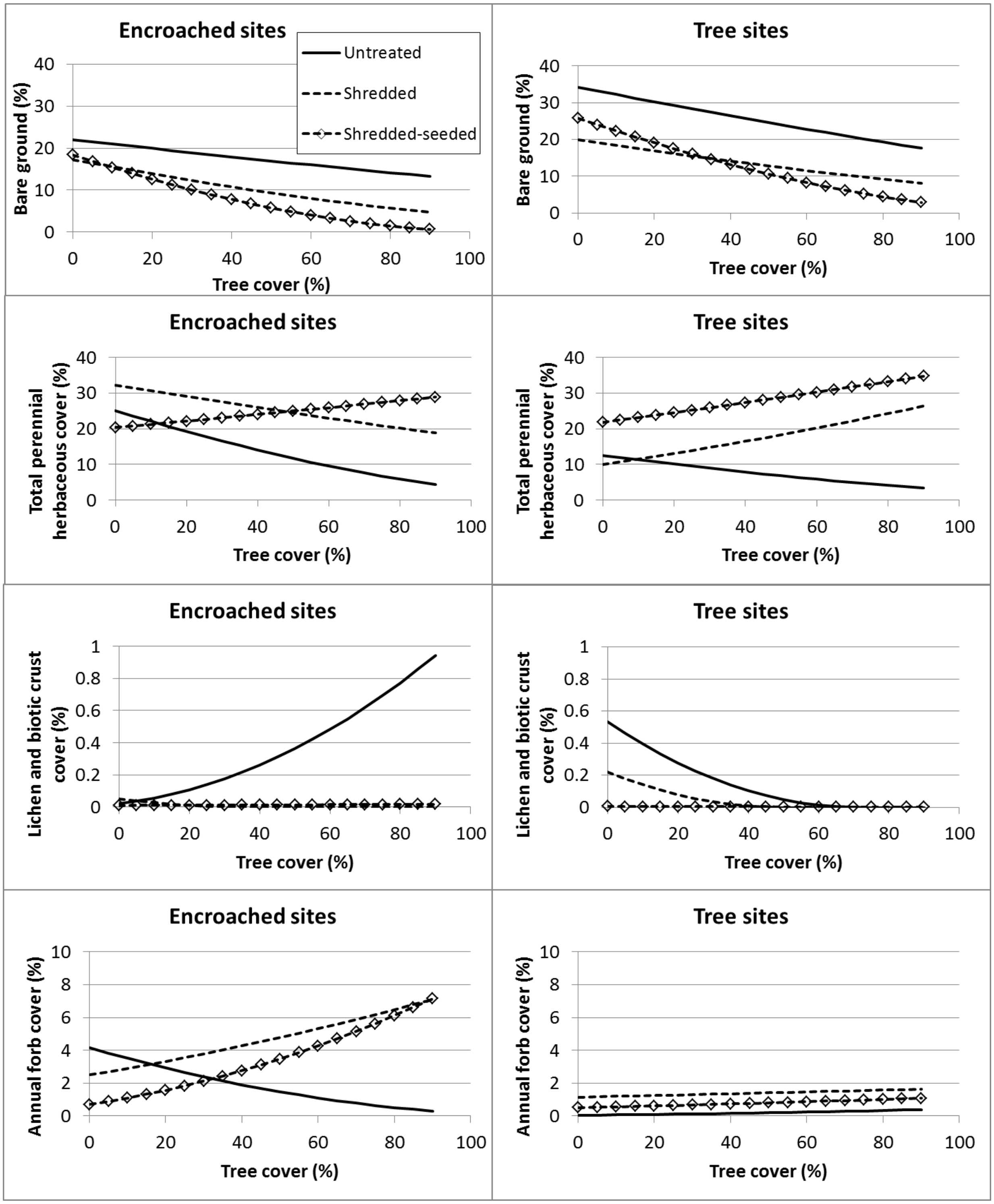

Figure 4. Cover for piñon and juniper sites encroached into shrublands (left) and for tree sites (right) in relation to pretreatment tree cover on untreated, tree-shredded, and tree-shredded-seeded treatments in Utah. See table 2 for significant differences.

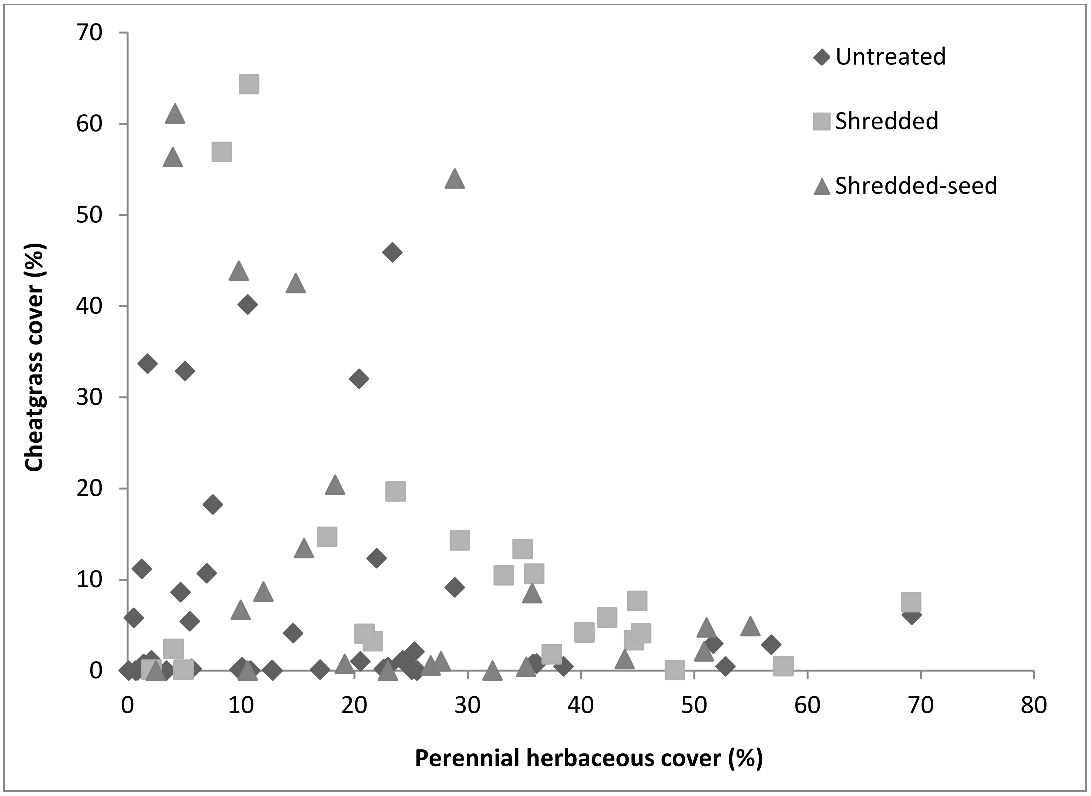

Figure 5. Mean cheatgrass cover in relation to total perennial herbaceous cover by site on untreated, tree-shredded, and tree-shredded-seeded treatments in Utah.