

# Fuel Reduction, Seeding, and Vegetation in a Juniper Woodland

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Western juniper has increased in density and distribution in the interior Pacific Northwest since the late 1800s. Management goals for many juniper woodlands are now focused on reducing tree densities and promoting biodiversity, prompting the use of fuel reduction treatments. Fuel reduction often involves mechanical cutting and disturbances such as slash pile burning and skid trail formation. While these activities may reduce tree densities, the extent to which they will restore native biodiversity and community composition, particularly in woodlands invaded by exotic annual grasses, is unclear. We evaluated the effects of juniper cutting in two experiments of disturbance type (slash piles and skid trails) followed by three native seeding treatments (cultivar, locally sourced, and no seed) on vegetation in central Oregon. Prior to cutting, native perennial grass cover and richness were positively associated and exotic grass cover was negatively associated with juniper basal area. After cutting and 2 yr after seeding, species composition was altered for both disturbance types. Some seeded areas had higher total species richness, higher native species richness, higher cover of seeded species, and higher overall cover compared to areas that were not seeded. But seeding effectiveness in mitigating exotic species spread varied based on exotic species functional group, pretreatment propagule pressure, and experiment disturbance type. Neither seed mix lowered exotic grass cover. There was limited evidence that the cultivar mix outperformed the locally sourced native seed mix. In the short term, fuel reduction activities may have facilitated further conversion of this woodland to an exotic grassland, but longer-term evaluation is needed. In juniper woodlands that have been invaded by exotic species, fuel reduction activities may facilitate further invasion, and exotic species control may be needed to limit invasion and promote native vegetation.

**Key Words:** big sagebrush, biotic resistance, exotic invasive plants, juniper encroachment, *Juniperus occidentalis*, thinning

## INTRODUCTION

Western juniper (*Juniperus occidentalis* Hook.) woodlands represent the northwestern extension of the pinyon-juniper woodlands of the Great Basin (Young and Evans 1981). Pinyon and juniper woodlands have expanded their range since the late 1800s at the expense of sagebrush steppe and other Great Basin plant communities. This expansion has been attributed to climatic variability, historically heavy livestock grazing around the turn of the 20th century, fire suppression, and increases in atmospheric CO<sub>2</sub> (Young and Evans 1981; Eddleman 1987; Miller et al. 1987; Miller and Wigand 1994; Knapp and Soulé 1998; Miller and Rose 1999; Miller et al. 2005). As a result of the increased density and distribution of western juniper, significant impacts on a variety of ecosystem functions have been reported, such as soil resources, plant community structure and composition (loss of shrubs, lower herbaceous cover), water and nutrient cycles, wildlife habitat, and biodiversity (Bates et al. 2000; Miller et al. 2000, 2005; Pierson et al. 2007). Control of western juniper has been a major land management focus since the early 1960s.

Fuel reduction treatments that reduce juniper density, such as cutting and burning, have been implemented by land managers to reduce wildfire severity and restore sagebrush steppe and grassland ecosystems. Various treatments to reduce juniper density can increase herbaceous production and cover following treatment, although failures are reported if sites lack adequate herbaceous perennial vegetation to prevent invasive species invasion (Young et al. 1985; Vaitkus and Eddleman 1987; Bates et al. 2000, 2005, 2011; Miller et al. 2005; Bates and Svejcar 2009; Baughman et al. 2010; Condon et al. 2011). Disturbances such as tree cutting, slash pile burning, and skid trail formation alter resource availability by opening forest canopies and reducing above- and belowground competition (Davis et al. 2000). Slash pile burning typically results in complete vegetation mortality, probable reduction of seed bank resources (Korb et al. 2004), and ash bed formation. Skid trail disturbance typically leads to limited mortality of existing vegetation. However, trails expose large areas of bare compacted soil by repeated disturbance as logs are transferred to landings. Examination of vegetation recovery following fuel reduction, slash pile burning, and skid trail formation can assist the development of appropriate management practices.

Restoration of native biodiversity in juniper woodlands after fuel reduction activities often depends on the threat of exotic plant invasion, especially invasive annual grasses (e.g., cheatgrass, *Bromus tectorum* L., medusahead, *Taeniatherum caput-medusae* [L.] Nevski, and North Africa grass, *Ventenata dubia* [Leers] Coss.), particularly at lower-elevation sites that are highly disturbed (Miller et al. 2005; Davies et al. 2011). Exotic invasive species are well adapted to exploit postdisturbance environments frequently characterized by increased resources. Factors that

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contribute to the invasibility of a juniper woodland after fuel reduction include pretreatment understory composition, and abundance and vigor of existing exotic invasive plants (Bates et al. 2000; Chambers et al. 2007). As western juniper begins to dominate a plant community, understory species and soil seed banks typically decline (Bates et al. 2000; Miller et al. 2000), and soil resources become less available (Bates et al. 2002). The opportunity for exotic invasive plants to establish after tree canopy removal increases on sites where the understory shrub and herb layers are low in abundance prior to cutting (Miller et al. 2005). Bates et al. (2000) concluded that a minimum of 2–3 bunchgrasses · m<sup>-2</sup> may be necessary to avoid dominance by annual grasses. This suggests that later seral and more closed canopy stands may be at higher risk for postcutting plant invasion, as understory composition may be depleted.

Seeding is used after disturbances and management activities to increase establishment of perennial plant species and lower the probability of exotic species dominance. Seeding has been successful in increasing native species diversity in cheatgrass-dominated rangelands (Ratzlaff and Anderson 1995; Cox and Anderson 2004). However, only a limited number of species or species mixes, in particular perennial grasses, have proven to effectively displace invasive exotic species (DiTomaso 2000; Davies 2010). A current debate regarding seeding practices is whether locally sourced plant seed should be used or whether “off-the-shelf” cultivars are more appropriate. A cultivar is a distinct, intentionally developed subset of a species that will behave uniformly and predictably when grown in an environment to which it is adapted. Cultivars are often used in large quantities because they are less expensive than locally sourced native seed. Some have also been selected for aggressive traits (vigorous vegetative growth or high fecundity; Gustafson et al. 2004), inferring possible increased competitive advantage. Interest in using locally sourced native seed has increased in response to concerns over both the conservation of local genetic resources and unknown impacts on species and plant community diversity and function as a result of cultivar use (Aubry et al. 2005).

The objective of this study was to examine the effect of fuel reduction and seeding (cultivar, locally sourced) on herbaceous plant communities. We developed two experiments focused on major disturbances associated with fuel reduction: slash pile burning and skid trail formation. We hypothesized that 1) seeded areas would have less exotic species cover and higher native plant cover, 2) the cultivar mix would outperform the locally sourced mix, and 3) areas with lower juniper abundance prior to cutting would have less exotic plant invasion because they would have a more abundant and diverse pretreatment perennial plant understory and lower exotic plant cover and richness and would be disturbed less by cutting.

## METHODS

### Study Site

This study was located just east of the Cascade Mountains in the Blue Mountains ecoregion approximately 21 km southwest of Madras, Oregon, on the Crooked River National Grassland (CRNG; lat 44°31'26.22"N, long 121°20'01.04"E). The 70-ha study area is a big sagebrush woodland (*Juniperus occidentalis*

*Artemisia tridentata*/Thurber's needlegrass (*Achnatherum thurberianum* [Piper] Barkworth) managed by the US Department of Agriculture (USDA) Forest Service, located at a mean elevation of 823 m. Subspecies identification of big sagebrush was not clear based on morphological traits, although habitat and elevation conditions suggest that *wyomingensis* may be the correct subspecies (Richard Halse, Oregon State University, personal communication, October 2013). Nomenclature follows the USDA–Natural Resources Conservation Service (USDA-NRCS 2013). The stage of woodland succession (sensu Miller et al. 2000) at the site was variable, with open to closed canopy patches present at the site. The historical climax plant community for the site is juniper shrubby loam 10–12 PZ, with a vegetative composition of approximately 80% grasses, 5% forbs, and 15% shrubs/trees (USDA-NRCS 2014). The site could be classified as a “wooded shrubland” based on Romme et al. (2009). Other common shrubs included antelope bitterbrush (*Purshia tridentata* [Pursh] DC.) and yellow rabbitbrush (*Chrysothamnus viscidiflorus* [Hook.] Nutt.). Other common grasses included Sandberg bluegrass (*Poa secunda* J. Presl), bluebunch wheatgrass (*Pseudoroegneria spicata* [Pursh] Å. Löve), and Idaho fescue (*Festuca idahoensis* Elmer). Forbs made up a small proportion of the flora. Common exotic species included cheatgrass, medusahead, and dense silkybent (*Apera interrupta* [L.] P. Beauv.). Small quantities of North Africa grass were also present. The study area was grazed by sheep up until the 1980s, but since then no livestock grazing has occurred.

Mean water year annual precipitation (MWYAP; October–September) was 215 mm for a 27-yr period (1985–2012; based on Haystack, Oregon RAWS USA Climate Archive, 18 km southwest of the study area [Western Regional Climate Center 2013]). In 2010, water year annual precipitation (WYAP) was 163% of MWYAP, and in 2011, WYAP was 105% of MWYAP. Study plots were established on the Bakeoven-Agency-Madras soil type (0–3% slopes). These soils were formed on lava plains and hills and are mostly moderately deep (60%) and well drained, with very shallow (30%) inclusions (USDA-NRCS 2002). Average depth to bedrock, consisting of basalt or tuff, for this area was about 25 cm to 35 cm (USDA-NRCS 2002). Texture data of the upper 10 cm was collected in 2011 in the field by the “texture-by-feel” method (Thien 1979). Almost all plots were sandy loams or loamy sands, with inclusions of sandy clay loams and loams.

The fuel reduction activities were designed to reduce the amount of western juniper to pre-European settlement (hereinafter presettlement) levels to restore and/or rehabilitate native shrub and grassland vegetation (USDA Forest Service 2004). Activities consisted of chainsaw cutting, piling and burning, or removing all young or postsettlement western juniper by skid steer. Old-growth juniper trees were not cut and were identified using growth-form characteristics, such as twisted, gnarled trees, and not by size (Miller et al. 2005).

### Experimental Design

Our study included two experiments, each focused on a specific disturbance type including 1) burned slash piles ( $n=20$ ) and 2) skid trails ( $n=15$ ). The slash piles and skid trails were located within the same cut study area. About 15% of the study area

was highly disturbed by slash piles, skid trails, and landings. About 13% of the area was neither thinned nor disturbed. The experimental design for each experiment was completely randomized (plots randomly located). Because juniper canopy cover varied across the study area, we categorized juniper patches into two broad classes using NAIP imagery from 2005 in a GIS: low juniper abundance (canopy cover was approximately 13%) and high juniper abundance (canopy cover approximately 47%).

**Slash Pile Disturbance Experiment.** Slash piles formed the experimental units. Prior to cutting, we established and permanently marked random plot center locations on the ground where 2-m-diameter slash piles would be burned. We randomly selected an equal number of points within each juniper canopy cover class for each treatment in a GIS. Slash piles were at least 1.5 m away from any tree bole and 15 m from old-growth tree boles. Slash piles could not be located in an area with state-listed noxious weeds. Cutting was done in the late summer of 2008 and hand piled in the fall. Piles were ignited by drip torch and burned in December 2009. Seed mixes were randomly assigned to plots: cut+burn+cultivar seeded (cultivar), cut+burn+local source seeded (local), and cut+burn+no seed (no seed). Use of the term “treatment” in this article refers to these specific combinations. The slash pile experiment was a balanced design with 20 replicates (20 uncut, untreated plots as described below and 20 plots each of the cultivar, local, and no seed).

Twenty plots in 16 patches were also established prior to cutting, and these plots serve as no-treat areas (no treat: not cut, seeded, or disturbed). These untreated patches were not established randomly, but plots within patches were. The average size of untreated patches, which were nested within the treated area, was about 36 m in diameter (range 26–46 m). These plots serve as no-treat areas for both experiments.

**Skid Trail Disturbance Experiment.** Skid trail segments were the experimental units. Skid trails were formed by a skid steer in the fall/winter of 2008 during cutting and slash removal and could not be determined a priori; therefore, skid trail plots were randomly selected for seeding in May 2009. Skid trails were plotted in the field using GPS, and maps were created in a GIS. Trails were divided into 20-m-segment lengths, and segments were randomly selected in a GIS; 45 segments were selected in the field if they met plot criteria. Average skid trail width was 3.7 m. Plot criteria included clear evidence of skid steer usage and exposure of bare soil for seeding and being at least 25 m from study, landing, and no-treat boundaries. Plot centers were a minimum of 25 m from each other. Seeding treatments were randomly assigned to plots: cut+skid formation+cultivar seeded (cultivar), cut+skid formation+local source seeded (local), and cut+skid formation+no seed (no seed). Use of the term “treatment” in this article refers to these specific combinations. The skid trail experiment is a robust though unbalanced design with 20 uncut, no-treat plots and 15 replicates each of the cultivar, local, and no-seed treatments.

### Seeding

For both experiments, two different seed mixes were used: 1) cultivar, a mix of available native cultivars, and 2) local, a mix

of the same species but locally sourced. Locally sourced seed was collected from the CRNG at a similar elevation and with similar soil types as our study areas. In coordination with local area managers, the following species were selected for seeding: squirreltail (*Elymus elymoides* [Raf.] Swezey [ELEL]), common yarrow (*Achillea millefolium* L. [ACMI]), and bluebunch wheatgrass (PSSP). Squirreltail is a short-lived early seral native perennial bunchgrass, and common yarrow is a rhizomatous native forb that is known to increase aggressively and rapidly in disturbed areas (Aleksoff 1999). Bluebunch wheatgrass is a large deep-rooted native perennial and is a target species for restoration in the area. Mature large perennial bunchgrasses, such as bluebunch wheatgrass, are important contributors to invasion resistance in these ecosystems (Chambers et al. 2007; Davies 2008; James et al. 2008). The names of the cultivars used were PSSP “Anatone,” ELEL “Toe Jam Creek,” and ACMI “Eagle Mountain.”

All seeded slash pile plots were broadcast seeded by hand onto snow after piles cooled in December 2009. Skid trail plots were also seeded at this time and were hand raked prior to and immediately after seeding. Seeding rates were approximately 2.4 kg PLS · ha<sup>-1</sup> for bluebunch wheatgrass, 1.8 kg PLS · ha<sup>-1</sup> for bottle brush squirreltail, and 0.2 kg PLS · ha<sup>-1</sup> for western yarrow. Seeding rates approximated those recommended by Sheley et al. (2008) for broadcast seeding, except the rate for ACMI was higher because the cultivar seed mix was premixed at the higher rate.

### Sampling

Prior to the onset of all fuel reduction activities, data were collected in 2008 (slash piles only) and again in 2011, the second growing season after seeding and three growing seasons after cutting and skid trail formation. Data were collected using the same methods for both experiments. At each plot center, a 10-m-radius permanent plot was established. Mature trees (> 1.37 m) and all stumps were tallied, and diameter at breast height or two perpendicular stump diameters were recorded. Overstory tree cover was measured using a moosehorn densitometer at the plot center and 5 m and 10 m in each cardinal direction (eight points per plot).

Cover data were collected for all species using a square 1 × 1 m plot frame located at the plot center. Cover plot size was based on the size of the slash piles (2-m diameter) and our desire to use the same sampling procedures for both experiments. Plant canopy cover refers to aerial canopy cover and is the percentage of ground area beneath the aerial canopy of a given species or life form. Cover was visually estimated to the nearest percentage using systematic marks on a plot frame. Half percent designations were used up to 3% cover. Any species less than 1% was recorded as 0.5%. Throughout the project, only one person recorded cover. Cover was also recorded for bare soil, rock (> 2 mm), and litter (all dead plant material, e.g., pine needles, bark, and dead grass).

### Data Analysis

Response variables were the same for both experiments and included vegetation cover (total and by functional group) and richness. Species-specific analyses were limited to the three seeded species of interest (squirreltail, common yarrow, and

bluebunch wheatgrass) and Sandberg bluegrass. Sandberg bluegrass was analyzed separately because it is smaller and its phenological development occurs earlier than other perennial bunchgrasses. Other species were combined into one of six functional groups based on life history and morphology (Table 1). Shrubs had not yet recovered and were not analyzed. Groups used to analyze richness were more broad and included perennial grasses, perennial forbs, annuals, and all exotics.

Response variables for each experiment were analyzed separately as a completely randomized analysis of covariance design using Proc Mixed in SAS 9.2 (SAS Institute Inc. 2008). All effects were fixed. Juniper abundance prior to cutting was used as a covariate (hereinafter referred to as precut juniper basal area) because it can be an important determinant of differences in vegetation and potentially influence posttreatment responses. We chose juniper basal area as a measure of abundance because density does not account for tree size. Because precut data were not collected for the skid trail experiment, precut juniper basal area was estimated using stump diameters.

The full model for both experiments included treatment (no treat, cultivar, local, no seed;  $df=3$ ), the covariate precut juniper basal area (a continuous variable;  $df=1$ ), and the treatment by juniper basal area interaction ( $df=3$ ). The full model error term included 72 degrees of freedom for the slash pile component and 57 degrees of freedom for the skid trail experiment. However, we removed the interaction term if it was insignificant ( $P > 0.10$ ) and then reran the analysis. Likewise, if precut juniper basal area was insignificant ( $P > 0.10$ ), it was removed from analysis, and a completely randomized analysis of variance was run. We discuss statistically significant ( $P < 0.05$ ) and marginally different ( $0.10 > P > 0.05$ ) results and present  $P$  values whenever practical. Differences between all treatments were assessed using Tukey's adjusted values. Many variables were transformed to improve skewed distributions and heteroscedasticity based on an assessment of residuals (logit, log, or, rarely, square root). For presentation of results, least squares means or uncorrected back transformed least squares means associated with transformed data are presented, along with 95% confidence intervals.

We used the Multi-Response Permutation Procedure (MRPP) in PC-ORD 5.10 (McCune and Mefford 2006) to examine whether treatments differed from one another in community composition before and after treatment (Mielke 1991; Biondini et al. 1988; McCune and Grace 2002). For each comparison,

statistical significance was evaluated by a Monte Carlo method using 4 999 randomizations. While the resulting  $P$  value is useful for evaluating statistical significance (compared to chance), a description of the effect size that is independent of sample size is also provided—the chance-corrected within-group agreement ( $A$ ). Values of  $A$  greater or less than 0 ( $A$  does not exceed 1) are indicative of effect size. An  $A$  of 0.30 in community ecology is fairly high, and most values are commonly below 0.10 (McCune and Grace 2002). If MRPP results were significant, indicator species analysis was used to interpret these results by determining which species were most abundant and frequent within each treatment (McCune and Grace 2002). Indicator values ( $IV$ , relative abundance multiplied by relative frequency) range from 0 to 100. For both the MRPP analysis and the indicator species analysis, species with only three occurrences were removed and cover data square root transformed.

## RESULTS

### Slash Piles—Treatment Response

Juniper basal area and density did not differ among treatments prior to cutting ( $P=0.785$  and  $0.920$  respectively). Juniper basal area was reduced from  $7.7 \text{ m}^2 \cdot \text{ha}^{-1}$  in the no-treat plots to  $\sim 1.3 \text{ m}^2 \cdot \text{ha}^{-1}$  after treatment ( $P < 0.0001$ ). Similarly, juniper density was reduced from  $182 \text{ trees} \cdot \text{ha}^{-1}$  to an average of  $7.5 \text{ trees} \cdot \text{ha}^{-1}$  in the treated plots. Bare soil also doubled in all treated plots in 2011 compared to the no-treat cover of 35% ( $P < 0.0001$ ). Vegetation cover and richness for the various species and functional groups did not differ prior to treatments.

Juniper basal area influenced herbaceous cover and richness prior to cutting. We found a positive relationship between juniper basal area and 1) cover of bluebunch wheatgrass ( $P=0.013$ ) and Sandberg bluegrass ( $P=0.076$ ) and 2) richness of perennial grasses ( $P=0.006$ ). The opposite trend was found for exotic grass cover ( $P=0.003$ ), which decreased as juniper basal area increased.

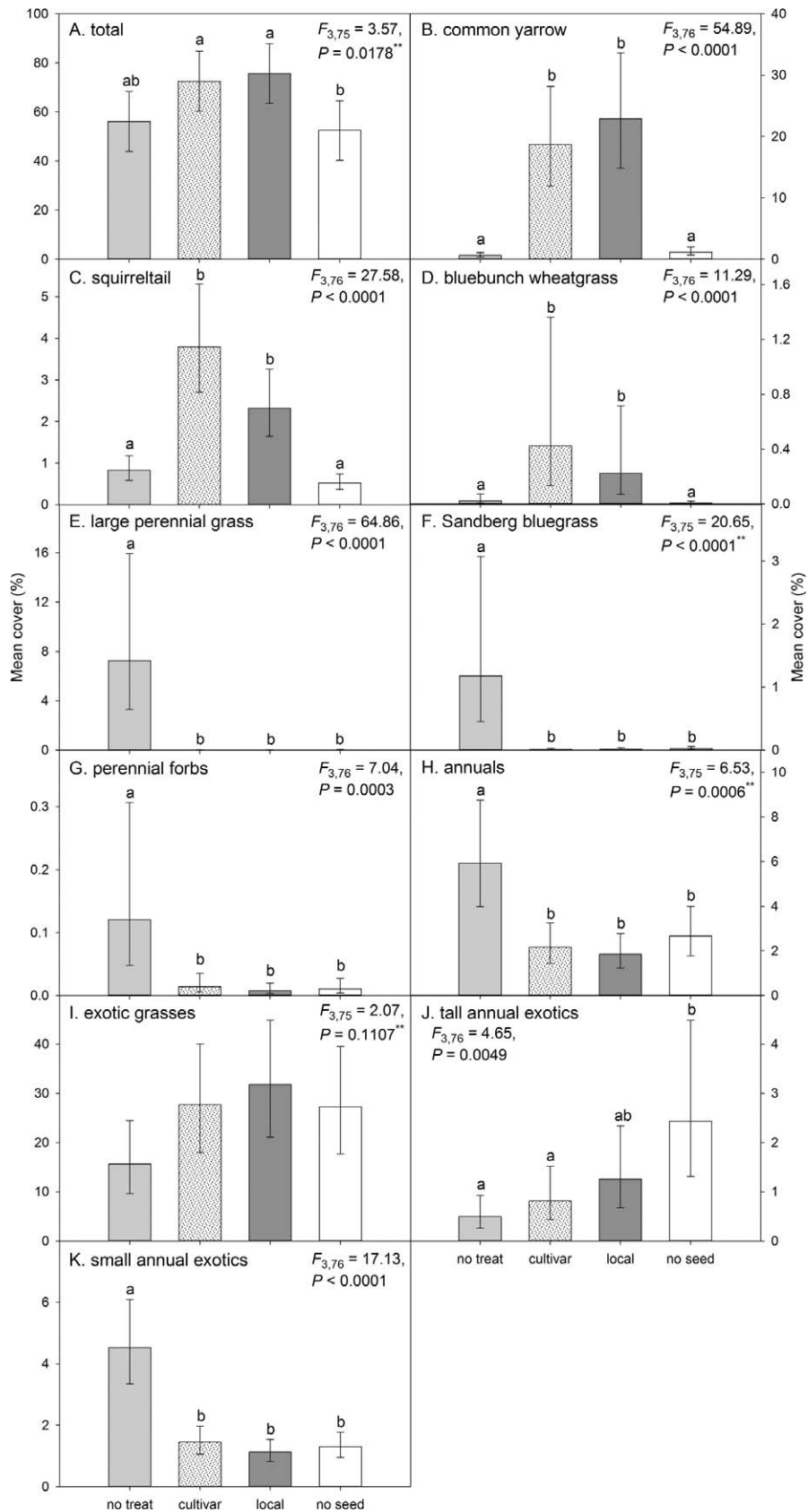
After treatments, total herbaceous cover increased in the no-treat plots from about 31% in 2008 to 56% in 2011. In 2011, total cover varied by treatment (Fig. 1A). Local and cultivar had higher and marginally higher total cover compared to areas that were not seeded ( $P=0.04$  local,  $P=0.10$  cultivar). The cultivar and local treatments did not differ ( $P=0.984$ ). Across all treatments, total cover had a negative relationship with the

**Table 1.** Vascular plant functional groups used in the analysis, dominant species, and origin (native or exotic) of species within each group. Nomenclature follows US Department of Agriculture–Natural Resources Conservation Service (2013).

Functional group	Dominant species	Origin
Large perennial grasses	Thurber's needlegrass, Idaho fescue	Native
Perennial forbs	Low pussytoes ( <i>Antennaria dimorpha</i> [Nutt.] Torr. & A. Gray), velvet lupine ( <i>Lupinus leucophyllus</i> Douglas ex Lindl.)	Native
Annuals <sup>1</sup>	Pacific popcornflower ( <i>Plagiobothrys tenellus</i> [Nutt. ex Hook.]), small fescue ( <i>Vulpia microstachys</i> [Nutt.] Munro)	Native
Exotic grasses <sup>2</sup>	Cheatgrass	Exotic
Tall annual exotics	Tall tumblemustard ( <i>Sisymbrium altissimum</i> L.), littlepod false flax ( <i>Camelina microcarpa</i> Andr. ex DC.)	Exotic
Small annual exotics	Jagged chickweed ( <i>Holosteum umbellatum</i> L.), spring draba ( <i>Draba verna</i> L.)	Exotic

<sup>1</sup>Mostly forbs with the exception of the one grass noted in example.

<sup>2</sup>Dominated by cheatgrass but contains a few perennial species.



**Figure 1.** Least squares (LS) mean or back-transformed LS mean functional group cover (%) and 95% confidence intervals from analysis of variance (ANOVA) or analysis of covariance (ANCOVA) results for 2011, 3 yr after juniper cutting, and 2 yr after slash pile burning and seeding. Native forbs, bluebunch wheatgrass, large perennial grasses, and Sandberg bluegrass were log transformed. All other variables were logit transformed except total cover (untransformed). Different lowercase letters denote statistically significant differences among treatments ( $P < 0.10$ ). An asterisk next to a  $P$  value indicates that precut basal area was a significant covariate in the model.

precut juniper basal area—higher total cover was associated with lower precut juniper basal area ( $P=0.007$ ).

The three seeded species (common yarrow, bottlebrush squirreltail, and bluebunch wheatgrass) established in the seeded plots (Figs. 1B–1D). Cover for all seeded species was significantly higher in local and cultivar plots compared to no-seed and no-treat plots, and there was no significant difference in cover between the cultivar and local seed mixes for each species. Differences among treatments for bluebunch wheatgrass were very small ( $<0.5\%$  cover).

The native plant groups (large perennial grasses, Sandberg bluegrass, perennial forbs, and annuals) showed little to no recovery after treatment (Figs. 1E–1H). The differences detected for the perennial forb group were extremely small ( $\sim 0.1\%$ ) and likely not biologically significant. As detected prior to cutting, higher posttreatment Sandberg bluegrass cover was associated with higher precut juniper basal area, although the posttreatment relationship was stronger statistically ( $P=0.012$ ). Annuals showed the opposite trend; higher native annual cover was associated with lower precut juniper basal area ( $P=0.016$ ; Fig. 1K).

The increase in total herbaceous cover was driven by an increase in exotic grass cover. Exotic grass cover increased throughout the study area in 2011, even in the no-treat plots. Exotic grass cover did not vary based on treatment (Fig. 1I), although no treat did have lower mean exotic grass cover compared to treated plots. The significant negative relationship between exotic cover and juniper basal area noted prior to cutting was still present ( $P=0.0006$ ). For tall annual exotic cover, no-seed plots had higher cover compared to the no-treat plots ( $P=0.003$ ; Fig. 1J) and marginally higher cover compared to cultivar plots ( $P=0.071$ ). For small annual exotics, cover for this group was lower in all treated areas compared to the no treat ( $P<0.0001$  for all comparisons; Fig. 1K).

In 2011, more and new exotic species were also found in the study area, some of which had not been observed prior to treatment. Cheatgrass remained the most abundant exotic species, but dense silkybent and soft brome (*Bromus hordaceus* L.) increased in cover. Species in the tall annual exotic group that were observed only along roadsides prior to cutting invaded many of the slash pile plots. Canada bluegrass (*Poa compressa* L.), present in trace amounts, was recorded only in the local and cultivar seeded plots, suggesting that both seed mixes were contaminated with this exotic species.

Juniper cutting, slash pile burning, and seeding impacted richness across all functional groups (Fig. 2). In general, no-treat plots tended to have the highest species richness, followed by the cultivar and local treatments, with the no-seed treatment having lower richness, although the patterns were not always significant, and we caution that some differences were less than one species for some comparisons. These patterns are related largely to the complete mortality of most species following slash pile burning and the influence of seeding. The small difference in richness for perennial forbs for the seeded treatments is probably simply due to the presence of the seeded species common yarrow; over 90% of seeded plots contained yarrow, while only 40% of the no-seed plots did. Areas with higher precut juniper basal area had higher perennial grass richness across all treatments ( $P=0.007$ ). The opposite relationship was detected for annual richness ( $P=0.037$ ).

MRPP results suggest that treatments influenced species composition for the slash pile experiment. Prior to cutting, there were no differences among treatments related to species composition ( $P=0.171$ ,  $A=0.0044$ ). In 2011, differences in species composition emerged ( $P<0.001$ ). The strongest differences were between the no-treat and cultivar and local plots ( $A=0.19$  for cultivar vs. no treat and  $A=0.21$  for local vs. no treat). No-seed plots vs. the seeded plots also differed, although the effect was smaller ( $A=0.11$  for cultivar vs. no seed and  $A=0.12$  for local vs. no seed). There was a difference in species composition between the no-treat and no-seed plots, but the effect was weak ( $A=0.09$ ). There was no difference in species composition between the local or cultivar ( $P=0.60$ ). No-treat plots were characterized by Thurber's needlegrass ( $IV=78\%$ ), small fescue ( $IV=69\%$ ), and Sandberg bluegrass ( $IV=65\%$ ). Cultivar and local plots were characterized by seeded species such as squirreltail ( $IV=48\%$ ) for cultivar and common yarrow ( $IV=44\%$ ) for local. The strongest indicator for the no-seed plots was cheatgrass, with an  $IV$  of only 26%.

### Skid Trails—Treatment Response

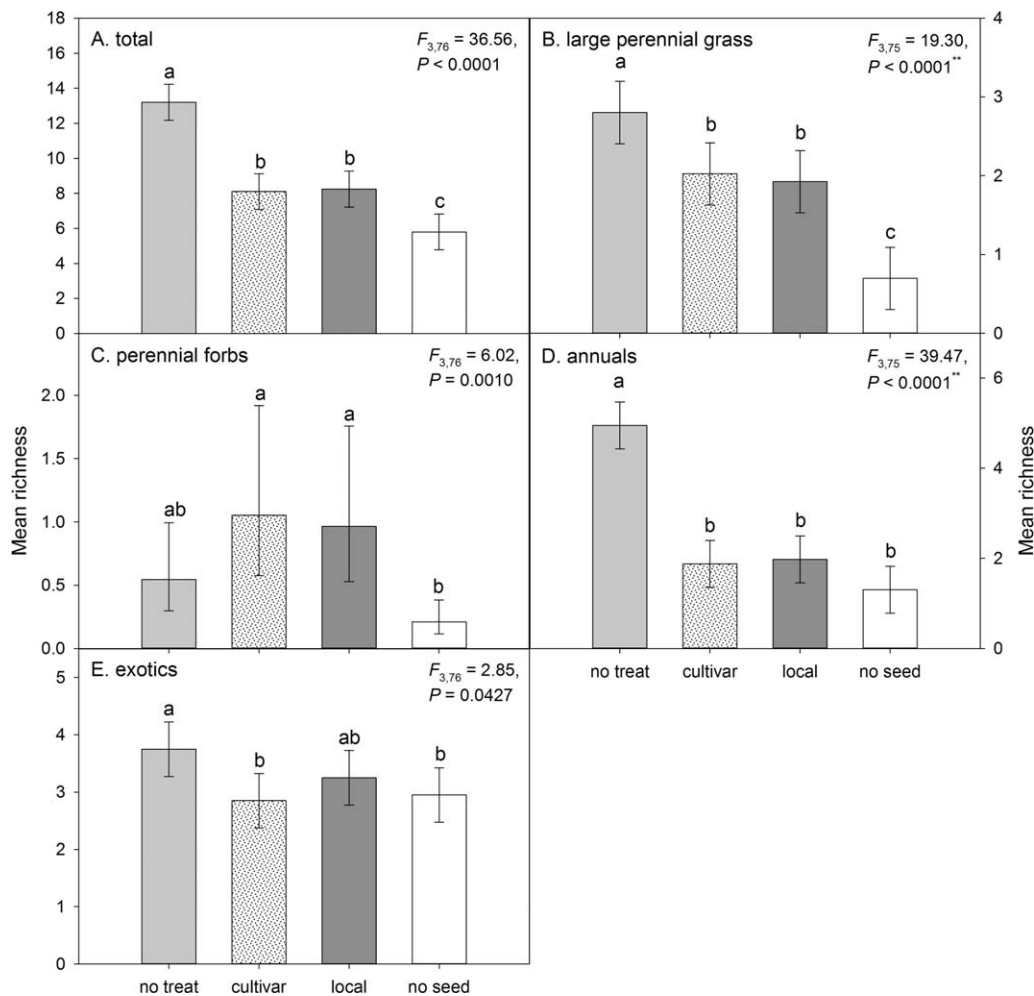
Juniper basal area and density prior to cutting (estimated from stump data) were not different ( $P=0.149$  and  $0.348$ , respectively). Live juniper basal area was 17 times and density over 100 times lower in the treated plots than in the no treat after cutting ( $P<0.0001$  for both variables). After treatment, no-seed plots had a 20% increase in bare soil cover compared to the no treat ( $P=0.022$ ).

In 2011, there was a significant interaction between treatment and precut juniper basal area ( $P=0.029$ ) for total herbaceous cover. Total cover was significantly lower in no-treat plots compared to all other treatments but only in areas with high precut juniper basal area (Fig. 3I). However, in these plots (high precut juniper basal area), there was no difference in cover between the cultivar or local plots and no-seed plots ( $P=1.0$  for both comparisons).

The seeded species did not establish well or have a consistent response in 2011 (Figs. 3A–3C). Only common yarrow showed significantly higher cover values in the cultivar and local seed compared to the no-treat and no-seed plots ( $P<0.005$ ).

Treatment was marginally significant for large perennial grass cover, but the response again depended on precut juniper basal area ( $P=0.065$ ; Fig. 3J). In areas with low precut juniper basal area, no-treat plots had higher large perennial grass cover than the local seed treatment ( $P=0.022$ ). Cover of Sandberg bluegrass was greater in the no-treat compared to the other treatments, although the difference between the no treat and no seed was only marginally significant ( $P=0.06$ ; Fig. 3D). No treatment differences were detected for the perennial forb group. For annuals, the response depended on precut juniper basal area (Fig. 3K). For high precut juniper basal area, local seed had marginally higher annual cover compared to no treat ( $P=0.099$ ).

The cultivar and local treatments did not impact exotic grass cover compared to the no seed (Fig. 3F). However, not seeding skid trails after cutting led to marginally higher exotic grass cover compared to the no treat ( $P=0.083$ ). For the tall annual exotic group, the local seed treatment had marginally higher cover compared to the no treat, where these species were not detected ( $P=0.068$ ; Fig. 3G). Unlike the slash pile plots,



**Figure 2.** Least squares (LS) mean or back-transformed LS mean functional group richness and 95% confidence intervals from analysis of variance (ANOVA) or analysis of covariance (ANCOVA) results for 2011, 3 yr after juniper cutting, and 2 yr after slash pile burning and seeding. Only perennial forbs were log transformed. Different lowercase letters denote statistically significant differences among treatments ( $P < 0.10$ ). An asterisk next to a  $P$  value indicates that precut basal area was a significant covariate in the model.

cultivar and local treatments did not impact tall annual exotic cover. For small annual exotics, all treated plots had significantly lower cover compared to the no treat ( $P < 0.030$  for all comparisons; Fig. 3H).

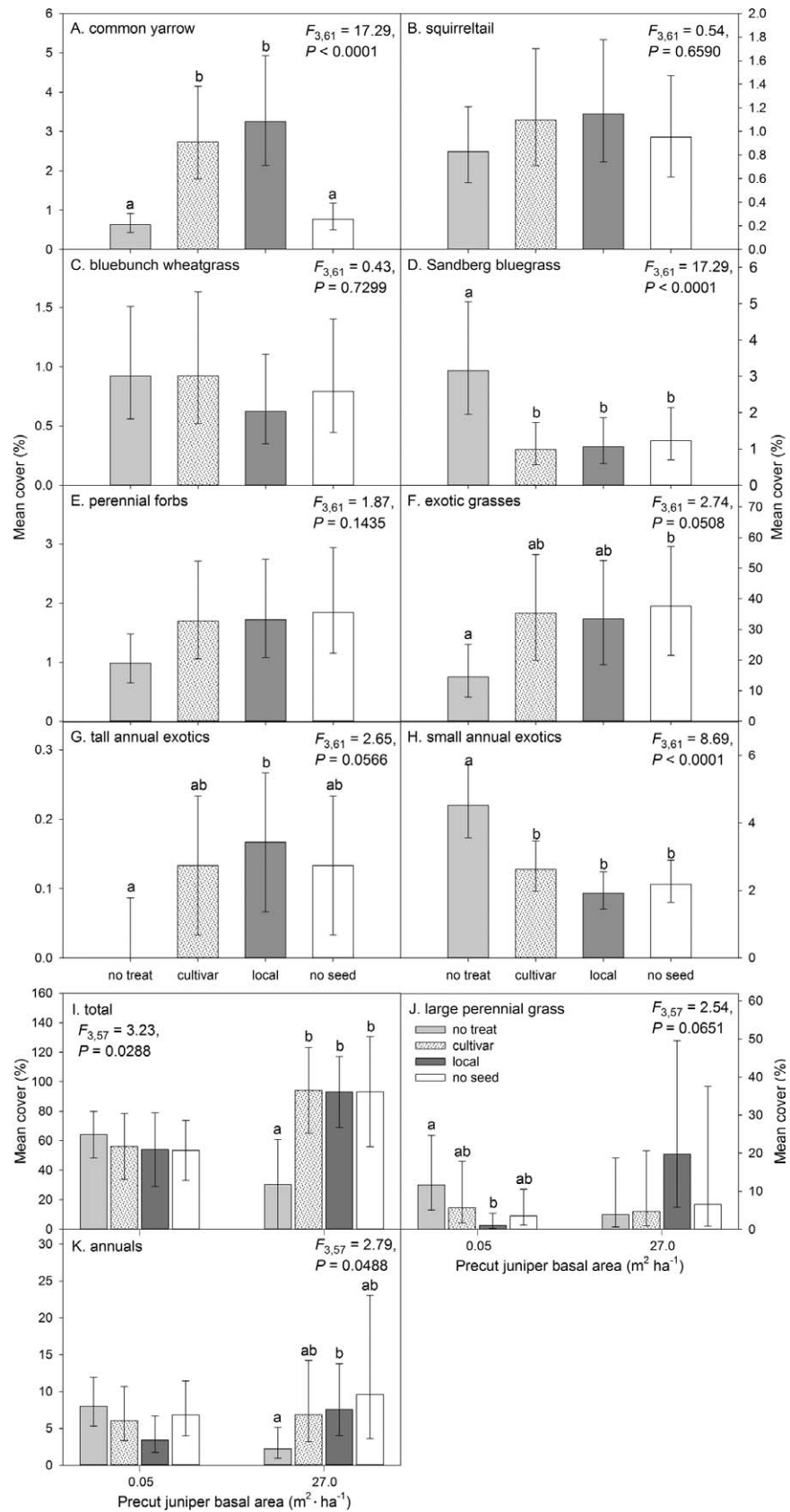
Juniper cutting, skid trail formation, and seeding impacted richness across most functional groups (Fig. 4). Total richness differed across treatments only marginally. The cultivar seed plots had marginally higher total richness compared to the no-treat plots ( $P = 0.060$ ; Fig. 4A). No patterns in relation to treatment were detected for perennial grass richness (Fig. 4B). Perennial forb richness differed by treatment, as the cultivar plots had marginally higher richness compared to the no treat ( $P = 0.04$ ; Fig. 4C). A similar pattern was detected for annual richness ( $P = 0.008$ ; Fig. 4D). Exotic richness varied for the local and cultivar plots, depending on precut juniper basal area (Fig. 4E). The local plots had marginally higher exotic species richness compared to the cultivar plots but only when precut juniper basal area was high ( $P = 0.067$ ).

MRPP results from 2011 suggest that the fuel reduction treatments significantly influenced species composition, but the effect was small ( $A = 0.031$ ,  $P < 0.001$ ). The only differences detected were between the no-treat and the other treatments

( $A = 0.0354$  for no treat vs. cultivar,  $0.043$  for no treat vs. local, and  $0.037$  for no treat vs. no seed). There was no difference in species composition between the cultivar and local plots ( $P = 1.0$ ) and no difference between cultivar and no seed ( $P = 0.300$ ) and local and no seed ( $P = 0.204$ ). Indicator species across all treatments were weak. The no treat was characterized by Sandberg bluegrass ( $IV = 38\%$ ) and the exotic forbs jagged chickweed (*Holosteum umbellatum* L.;  $IV = 36\%$ ) and spring draba (*Draba verna* L.;  $IV = 34\%$ ). The cultivar plots were characterized by the seeded species common yarrow ( $IV = 37\%$ ) and cheatgrass ( $IV = 27\%$ ). The local seed plots had similar results (common yarrow  $IV = 39\%$ , cheatgrass  $IV = 25\%$ ). The no-seed plots were characterized by the exotics cheatgrass ( $IV = 28\%$ ) and jagged chickweed ( $IV = 23\%$ ) and native annuals, such as small tarweed (*Madia exigua* [Sm.] A. Gray;  $IV = 26\%$ ).

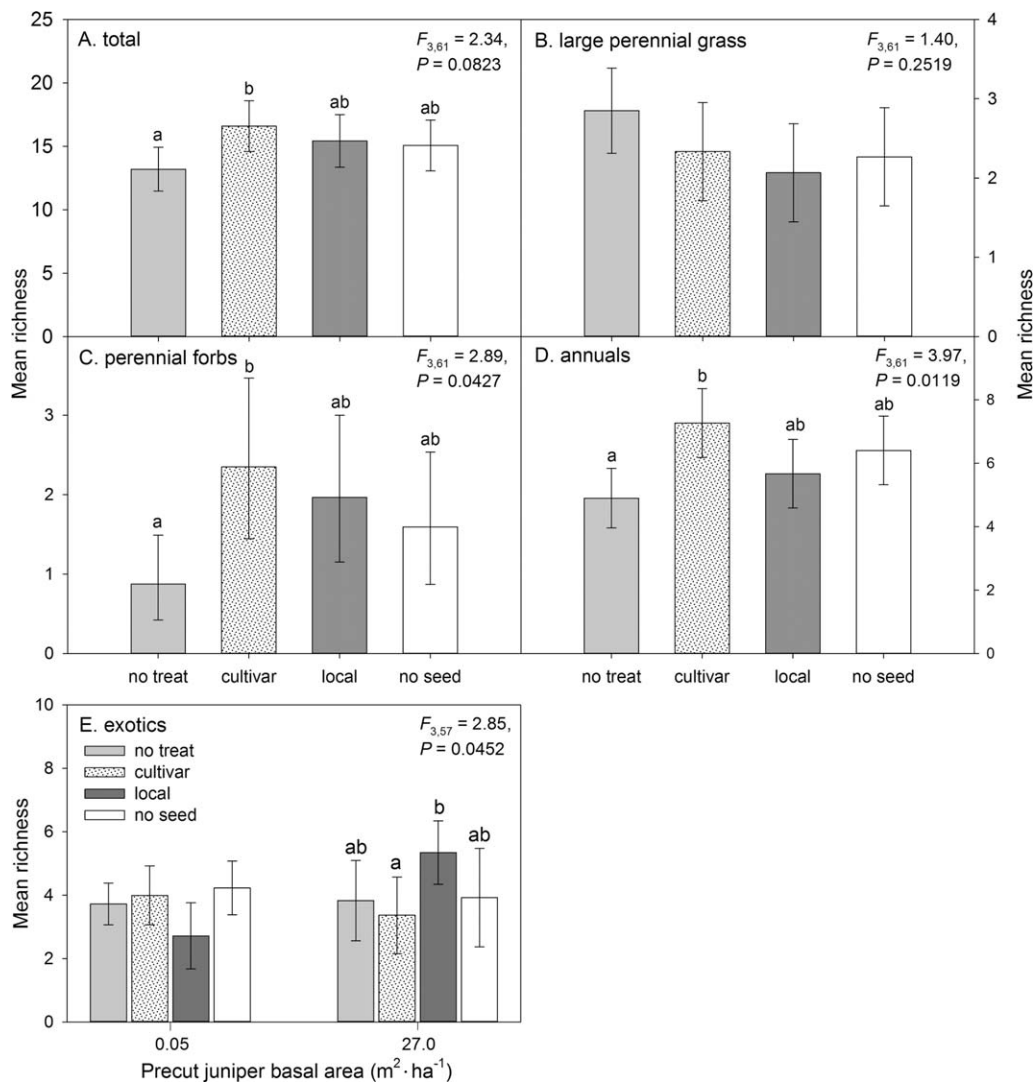
## DISCUSSION

We expected that after fuel reduction treatments, plots seeded with local and cultivar mixes would have lower exotic invasive



**Figure 3.** Least squares (LS) mean or back-transformed LS mean functional group cover (%) and 95% confidence intervals from analysis of variance (ANOVA) or analysis of covariance (ANCOVA) results for 2011, 3 yr after juniper cutting and skid trail formation, and 2 yr after seeding. Most variables were logit transformed, except Sandberg bluegrass was log transformed and total and tall annual exotics were untransformed. Different lowercase letters denote statistically significant differences among treatments ( $P < 0.10$ ). An asterisk next to a  $P$  value indicates that precut basal area was a significant covariate in the model. Some functional group responses were dependent on precut juniper basal area (significant interaction), and results are shown as appropriate.





**Figure 4.** Functional group least squares (LS) mean or back transformed LS mean richness and 95% confidence intervals from analysis of variance (ANOVA) or analysis of covariance (ANCOVA) results for 2011, 3 yr after juniper cutting and skid trail formation, and 2 yr after seeding. Most variables were untransformed, except perennial forbs were square root transformed. Different lowercase letters denote statistically significant differences among treatments ( $P < 0.10$ ). Some functional group responses were dependent on precut juniper basal area (significant interaction), and results are shown as appropriate.

species cover, higher native plant cover, and higher native richness compared to areas that were not seeded. Some seeded treatments did have higher total and native species richness, higher cover of seeded species, and higher overall cover compared to areas that were not seeded. Our seeding strategy appears to have been somewhat successful in establishing native plant cover in all plots but was less successful in skid trail areas. The lower seeding success in skid trails compared to burn piles is probably related to soil compaction, the presence of competitive extant vegetation, and possibly temporal issues associated with seeding. Seeding could not be done until slash piles dried for 1 yr, leaving skid trails exposed to colonization during the first growing season. However, overall seeding success was undoubtedly related to the ideal precipitation conditions in the spring and early summer of 2010 and 2011. It is often acknowledged that seeding success is tightly coupled to favorable climatic conditions (Hardegree et al. 2011). Two years after seeding, common yarrow was the most successfully

established species for both experiments, with values exceeding 60% cover on some slash pile plots.

Seeding effectiveness in mitigating exotic species spread varied based on disturbance type, exotic species functional group, and exotic species cover prior to cutting. The cultivar mix successfully lowered the cover of tall annual exotic species in slash pile areas. This outcome could be related to the success of seeding common yarrow in slash pile areas. Common yarrow is similar in growth form to species found in the tall annual exotic group. Limiting similarity theory predicts that plant communities are able to better resist invasion when resident species are more similar to potential invaders (Emery 2007). Our data also support recent evidence that early successional forbs can be successful as seeded species in preventing exotic invasions (Abella et al. 2012; Herron et al. 2013). Moreover, propagule supply for the tall annual exotic group of species was probably relatively low in the study area prior to cutting. Tall exotic forb species existed only on patches

along the roadway and were not present in the study plots in 2008. DiVittorio et al. (2007) found that native species dominated disturbances when exotic propagule pressure was low but failed to establish when exotic propagule pressure was high.

Exotic grass cover, which was largely composed of the annual grass cheatgrass, increased in 2011 after fuel reduction activities, including in the no treat. In both experiments, neither seed mix effectively lowered exotic grass cover compared to no seeding. Exotic grasses are of great management concern in the area, as these species can reduce native biodiversity and alter ecosystem processes and have been shown to increase regional fire activity across the arid western United States (D'Antonio and Vitousek 1992; Davies et al. 2011; Balch et al. 2013). Short-term invasion (5 yr and less) and spread of annual grasses following juniper control has been reported elsewhere (Young et al. 1985; Haskins and Gehring 2004; Baughman et al. 2010), but longer-term measurements suggest that this increase might be ephemeral for some sites (Vaitkus and Eddleman 1987; Davis and Harper 1990; Eddleman 2002; Bates et al. 2005, 2007; O'Connor et al. 2013). However, longer-term measurements at more sites are needed. For example, Bates et al. (2007) noted that although annual grass cover and density dropped 11 yr after cutting (compared to 1 yr after), differences were still detected. Moreover, the decline in annual grasses was documented only in 2003 and might simply reflect climatic patterns.

Our result that neither seed mix effectively lowered exotic grass cover in the short term is probably due to several factors. First, our study was conducted over two very favorable years for exotic annual grass growth. The newly established seeded species would have benefited from these conditions as well but had limited influence on annual grass growth and spread. Precipitation and its effects on available soil water appear to be the primary control on cheatgrass invasibility when temperature is not a limiting factor in sagebrush ecosystems (Chambers et al. 2007). Years with above-average precipitation provide ideal conditions for cheatgrass establishment and growth (Ganskopp and Bedell 1979). Second, while mature perennial native bunchgrasses can compete effectively with invasive annual species (Chambers et al. 2007; Davies 2008; James et al. 2008), they are slow growing, and our study examined only a short-term response. Posttreatment cover for squirreltail and bluebunch wheatgrass was low in both disturbance types. While common yarrow was somewhat established for both seed mixes across both disturbance types, it did not appear effective at interfering with annual exotic grass establishment and growth.

We expected that the cultivar seed mix would outperform the locally sourced seed. Our data provide some evidence that the cultivar seed mix may have been more effective, but the evidence was not strong. Most tests showed no clear evidence that the cultivar mix outperformed the local mix, and some data indicate that the local seed performed as well as the cultivar mix. While managers may be better off economically to use cultivars, those concerned with preserving local genetic resources may opt for using local native seed sources. Our data suggest no clear advantage from the perspective of weed suppression. We suggest that issues related to exotic species propagule pressure prior to fuel reduction activities, distur-

bance type, and the selection of functional analogs may be more critical than cultivar vs. native seed sources when considering postdisturbance mitigation treatments.

We also expected that areas with lower juniper abundance would have higher native perennial plant cover and lower exotic invasive species cover after treatment when compared to areas with higher juniper abundance. This expectation is based on the assumption that areas with lower juniper abundance started out with a more abundant understory. However, our data do not strongly support this hypothesis. The cover of some native perennial grasses (Sandberg bluegrass and bluebunch wheatgrass) and perennial grass richness were actually associated with higher juniper abundance prior to treatment. The pattern also persisted for Sandberg bluegrass and perennial grass richness after cutting and slash pile burning. Miller et al. (2000) reported that herbaceous cover and species diversity decline with increasing juniper dominance in mountain big sagebrush/Thurber's needlegrass communities, although the negative impact of juniper may not be as pronounced in deeper soils. However, Vaitkus and Eddleman (1991) found that for many native understory species, production was positively associated with proximity to juniper trees, particularly large trees. These authors concluded that the original perennial community dominants appeared to become spatially segregated beneath canopies of large trees, while species such as cheatgrass increased in abundance across the entire site. Exotic annual grasses are generally shade intolerant, and perennial grass species may find that areas under and near juniper canopies provide a competitive refuge. It is also possible that areas with more juniper might simply be more productive, and differences among site conditions may confound implications related to juniper abundance.

Slash pile results also suggested that exotic grass species abundance and therefore propagule pressure were somewhat mitigated by precut juniper abundance. In both 2008 and 2011, exotic grass cover declined as juniper abundance (or precut juniper abundance) increased. It is likely that juniper trees effectively create a sort of abiotic resistance to invasion by fundamentally altering microhabitat conditions (light, litter, and water) and creating a physiological inability for invaders to tolerate the habitat. Therefore, areas with higher juniper abundance might have some ecological resistance to invasion, at least in the short term, even after the canopy is removed. It remains to be seen how long this pattern will persist. Strong abiotic resistance is less likely to be overwhelmed by high propagule pressure and may be more important than biotic resistance in regulating invasions (Levine et al. 2004). Miller et al. (2005) noted that higher overstory tree canopy cover is probably one of the only factors stemming the spread of annual exotic grasses into many western juniper woodlands.

However, we do have limited data from our skid trail plots that suggest that areas with high juniper abundance might have lower overall ecological resistance. Areas with higher precut juniper basal area had higher posttreatment exotic richness but only for the local seed treatment, and the difference was only marginally significant for one comparison. Areas with higher precut juniper basal area also had higher posttreatment total cover compared with no treat (although cover was composed largely of exotic grasses) and higher native annual cover (only for the local seed treatment). As noted earlier, the skid trail

plots were highly disturbed as equipment was repeatedly used in these areas, skid trails were 3.7 m wide, and extensive seed transfer, particularly of annual exotics, was highly likely. Therefore, we suggest that the patterns related to high pre-cut juniper basal area may be related to the fact that these areas were more highly disturbed.

The fuel reduction activities appear to have, at least in the short term, facilitated further conversion of this woodland to an exotic grassland. It remains to be seen how long or if these patterns will persist. Miller et al. (2005) noted that juniper restoration may be particularly problematic in woodlands invaded by annual grasses at elevations less than 1525 m. A longer-term evaluation is needed to fully understand the impacts of the treatments presented in this study. Evaluation of herbaceous recovery and exotic invasive species persistence and the development of management suggestions are hampered by the lack of long-term vegetation measurements after juniper cutting (Bates et al. 2005). Yet we caution that in western juniper woodlands invaded by exotic annual grasses similar to those in this study, fuel reduction activities, even if followed by posttreatment seeding, may not effectively restore native grassland composition in the short term unless significant investments in exotic invasive control are used. We recognize that the extensive resources needed for effective invasive species control are often lacking and that limited options may be available to managers. Managers may opt to limit fuel reduction activities until better solutions related to exotic annual grass control are found. However, it is also possible that not reducing fuels may result in catastrophic wildfire and potential dominance of exotic annual grasses. When sagebrush communities are threatened by annual grass invasion and conifer encroachment, conservation efforts may be most limited by annual grass invasion, and therefore management actions that address the annual grass issue are most relevant (Davies et al. 2011).

## MANAGEMENT IMPLICATIONS

We acknowledge that we report only short-term responses from a small study area. Several studies have shown that short-term results are poor predictors of both success and failure of long-term recovery and seeding outcomes (Bates et al. 2005; Rinella et al. 2012). Miller et al. (2005) suggest that as long perennial bunchgrass densities are adequate, eventual recovery of cut juniper ecosystems is usually predictable. Assessments of these treatments across a broader landscape and in the longer term are needed. Our study was also focused largely on highly disturbed areas (slash piles and skid trails), and we examined only herbaceous species response. In addition, we did not include a comparison of areas that were cut but not disturbed by either skid trails or slash piles. Comparing skid trail and slash pile plots back to our no treat could be misleading if this caveat is not recognized. While we do not have data regarding areas that were only cut and not highly disturbed, we observed that increases in exotic species in the cut area appeared to be comparable to our highly disturbed and no-treat plots.

Therefore, results from our study should be interpreted cautiously but may be applicable to similar western juniper/big sagebrush/Thurber's needlegrass woodlands. In areas similar to

our study with similar fuel reduction treatments and operations, managers may consider that 1) exotic invasive species increase in these ecosystems with disturbances such as juniper cutting in combination with skid trail formation or slash pile burning; 2) extensive exotic species control may be necessary after treatments; 3) native seeding in disturbed areas may help establish native plant species, particularly if done immediately after treatment, which could be important for long-term succession; 4) seeding of faster-growing, more aggressive species may provide better outcomes in terms of exotic species control than using native slow-growing bunchgrasses in the short term; 5) seeding in skid trail areas may require better seed bed preparations than hand raking, such as drilling; 6) locally sourced native seed mixes may be just as effective as cultivars, but pretreatment abundance of exotics, disturbance type, and selection of functional analogs are probably more critical to consider than seed source when deliberating seeding mitigation for exotic species; and 7) short-term responses may not be good predictors of long-term trends, and more time is needed to determine if seeded species are effective at limiting exotic invasion and restoring native plant communities.

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