

Article

Long-Term Seeding Outcomes in Slash Piles and Skid Trails after Conifer Removal

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Abstract: Conifer removal in interior woodland ecosystems of the western US is a common management treatment used to decrease fire hazard and shift woodlands to more historical states. Woody material is frequently removed by skidding material off site and via slash pile burning. Assessing the long-term outcomes of seeding treatments after such ground disturbing activities is critical for informing future management and treatment strategies. Using two designed experiments from a central Oregon juniper woodland, we resampled slash piles and skid trails 8 years after seeding. Our objectives were to assess the long-term vegetation response to conifer removal, ground disturbance, and seeding source (cultivar and local) in slash piles and skid trails. We found that seeded species persisted in the long term, but abundance patterns depended on the species, seed source, and the type of disturbance. In general, there were more robust patterns of persistence after pile burning compared to skid trails. Seeding also suppressed exotic grass cover in the long term, particularly for the local seed source. However, the invasion levels we report are still problematic and may have impacts on biodiversity, forage and fire behavior. Our short-term results were not predictive of longer-term outcomes, but short- and long-term patterns were somewhat predictable based on species life history traits and ecological succession. The use of a mix of species with different life history traits may contribute to seeding success in terms of exotic grass suppression. Lastly, our results suggest that locally adapted seed sources may perform as well or better compared to cultivars. However, more aggressive weed treatments before and after conifer removal activities and wider seeding application may be needed to effectively treat exotic grass populations.

Keywords: exotic annual grass; sagebrush; functional groups; juniper removal; seeding; restoration

1. Introduction

Conifer encroachment and densification are global phenomena that have been documented in numerous ecosystems, including dry conifer forests and woodlands, oak woodlands, mountain meadows, shrublands and grasslands [1–5]. In the interior west of the US, conifers such as piñon (*Pinus* spp.) and juniper (*Juniperus* spp.) have expanded from two- to 10-fold throughout the historical range of shrublands largely dominated by sagebrush (*Artemisia* spp.) [6–8]. Western juniper (*Juniperus occidentalis* Hook.) expansion has been well documented both in terms of density and range, and has largely been attributed to a number of interacting factors including maturity and infilling of stands established in climatically favorable conditions, increased atmospheric CO₂, fire suppression and livestock grazing [6,9–13]. Afforestation after harvest may also be a contributing mechanism to recent conifer expansion [14]. Conifer encroachment and densification in the western US have resulted in: (1) the listing of over 350 sagebrush dependent plants and animals as species of concern [15,16];

(2) increasing woody fuels and crown fire potential [17]; and (3) the alteration of ecosystem functioning through changes in hydrological processes [18].

Owing to the potential for negative effects associated with conifer encroachment, land managers have sought to remove or reduce trees such as western juniper through mechanical cutting and burning in an effort to halt or reverse ecosystem type conversion and decrease woody fuels. Density management of piñon and juniper woodlands is widespread across the western US and has been ongoing for well over half a century [19]. Today, marketable woody material is commonly removed using ground based skidding operations which create a network of trails that have reduced or removed surface vegetation and increased soil compaction [20]. Nonmarketable material may also be removed on site by broadcast or pile burning. Pile burning initially removes all aboveground vegetation, and can result in soil blackening or reddening, increases in soil nutrient availability and hydrophobicity [21–27]. Aside from impacts on extant vegetation and soils, ground based skidding operations and slash pile burning can also facilitate exotic species invasion [27–30].

Seeding has been used extensively after conifer removal activities with an aim to facilitate cover of native or desired species and broader plant community recovery, and decrease the probability of establishment of exotic invasive species through competition [27,31–33]. However, both the species selected for seeding and potentially the origin of seeds can influence restoration outcomes. Perennial grasses are one of the only functional groups to effectively displace invasive exotic species such as annual grasses [34–36], although perennial grasses may take several years to establish and grow. Early successional forbs can establish quickly and also be successful as seeded species in preventing exotic invasions [37,38]. Additionally, while locally derived seed stocks are recommended for restoration to increase the likelihood that plants are adapted to the site [39], due to their limited availability and high cost [40], land managers typically use seeds from off-the-shelf cultivated plant varieties or cultivars. Cultivars are used extensively in a variety of restoration projects in North America, and are typically developed to have quick establishment, vigorous vegetative growth, and high seed production, which may confer a competitive advantage over local genotypes [41].

In this study, we examine the long-term effects of locally sourced and cultivar seeding treatments in slash piles and skid trails after the mechanical removal of western juniper in central Oregon in an effort to facilitate the cover of native species, suppress exotic weeds, and enhance native plant diversity post-treatment. Short-term results were previously reported by Kerns and Day [30]; however, assessing the long-term success of seeding treatments is critical for informing future management and treatment strategies, particularly in areas invaded by exotic invasive species [42]. Our specific research questions are as follows: (1) Do seeded species persist, suppress exotic invasion, and facilitate broader community recovery? (2) Were the short-term patterns presented in our earlier work [30] predictive of longer term outcomes? (3) Do locally sourced and cultivar seed sources persist and perform similarly? We also discuss our results in terms of the different nature of the two ground disturbance activities (slash piles, skid trails).

2. Materials and Methods

This study was originally initiated in conjunction with managers as part of a United States Department of Agriculture (USDA) Forest Service juniper removal project: The Crooked River National Grassland (CRNG) Westside Wildland Urban Interface Fuel Reduction Project. Management activities were designed to reduce the amount of juniper to pre-European settlement levels and restore native shrub and grassland vegetation [43].

2.1. Study Area and Seeding Treatments

The study was conducted within a 720-acre allotment of the CRNG site located along the convergence of the Deschutes, Crooked and Metolius Rivers in central Oregon, southwest of Madras, OR, USA at the northwestern edge of the Great Basin (Figure 1). The area is a Wyoming big sagebrush (*Artemisia tridentata* ssp. *Wyomingensis* Beetle & Young) western juniper Thurber's needlegrass

(*Achnatherum thurberianum* (Piper) Barkworth) woodland located at a mean elevation of 823 m. Parts of the site included abandoned homesteads and the area was grazed by sheep up until the 1980s, but since then no livestock grazing has occurred. Prior to treatment, the project area had infestations of some of the most invasive annual grasses in western North America: medusahead (*Taeniatherum caput-medusae* (L.) Nevski), cheatgrass (*Bromus tectorum* L.) and North Africa grass (*Ventenata dubia* (Leers) Coss.).

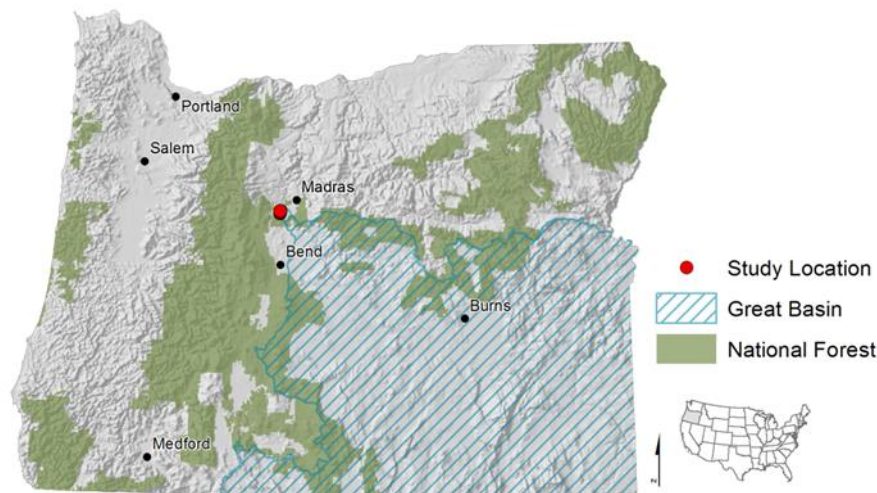


Figure 1. Location of the study area in central Oregon relative to national forestlands and the northwestern edge of the Great Basin as mapped by the Great Basin Restoration Initiative [41].

Mechanical tree removal was initiated in 2008 and most (80%) post-European settlement juniper was cut by chainsaw and removed by skid steer or pile burned (Table 1). No old-growth juniper was cut. Areas of retained juniper were used as a control or no cut treatment. Selection of these areas also focused on protecting wildlife habitat and old-growth retention and was done in collaboration with stakeholders; thus, controls are not completely random. Since juniper cover varied across the study area in a patchy manner, we categorized areas in a Geographical Information System (GIS) [44] (ArcMap Version 9.2) into low (~13%) and high (~47%) cover using NAIP imagery from 2005 prior to establishing plots. Cover was calculated using Feature Analyst and Image Sampler. Pretreatment canopy cover area measured using a moosehorn densitometer 10 m from the plot center in all four cardinal directions was highly variable, averaging 18% (range 0–74%, SE = 19%) across the study area and following juniper removal in 2011 averaged 2% (range 0–49%, SE = 7%).

Table 1. Timeline of activities for this study. Data reported in this paper were collected in 2017, eight growing seasons after seeding in the winter of 2009.

Activity	Date
Control units established	April 2008
Slash-pile plot locations established	April 2008
Slash-pile pretreatment data collected	Spring/Summer 2008
Juniper thinned	Spring–Fall 2008
Slash piled by fire crew	Fall 2008
Trees removed from site	Fall 2008–Summer 2009
Skid trail plots established (no pretreat data)	May 2009
Piles burned	December 2009
Piles seeded	December 2009
Skid trails raked and seeded	December 2009
Data collected (published by Kerns and Day, 2014)	Summer 2011
Data collected	Summer 2017

Two seed mixes were tested in ground disturbed areas created by slash pile burning and skid trail formation, a locally derived seed source and a cultivar source. The same species were used in each mix: bottlebrush squirreltail (*Elymus elymoides* (Raf.) Swezey or ELEM), bluebunch wheatgrass (*Psuedoroegneria spicata* or PSSP), and western yarrow (*Achillea millefolium* L. or ACMI). Squirreltail is a short-lived early seral native perennial bunchgrass, and common yarrow is a rhizomatous native forb known to spread rapidly in disturbed areas [45]. Bluebunch wheatgrass is a large deep-rooted native perennial and is a target species for restoration in the area. The local mix used seeds collected by managers within the CRNG at elevations and soil types that were similar to the study area. The cultivar mix was created using PSSP “Anatone”, ELEM “Toe Jam Creek” and ACMI “Eagle Mountain.” Seeding rates approximated those recommended by Sheley et al. [46] as described by Kerns and Day [30].

Seeding treatments were randomly assigned to slash piles and skid trail plots as described below. Twenty plots were established prior to juniper removal that served as control plots or no treatment (no trt) where juniper was not removed, and no seeding was done. In summary, there were three seeding treatments for each ground disturbance type in addition to the no trt: no seed, local seed mix (local), and cultivar seed mix (cultivar).

Slash pile plots: We established and permanently marked random plot center locations in 2008 on the ground where 2-m-diameter slash piles would be burned after cutting (Figure 2). We randomly selected an equal number of points within each juniper canopy cover class (low, high) for each treatment in a GIS [44] (ArcMap Version 9.2). Slash was hand-piled into small piles and overwintered before burning in December of 2009. Broadcast seeding onto snow for both slash piles and skid trails (see below) was completed in winter 2009. The slash pile experiment was a balanced design, with 20 replicates (20 no trt and 20 plots each of no seed, cultivar and local).



Figure 2. Slash piles after burning (left) and after seeding and several years of recovery (right). The white flowered plant on the right is common yarrow (ACMI), one of three seeded species, and the reddish grass is cheatgrass, the most common exotic annual grass in the study area. A one-meter sampling quadrat is shown on the right.

Skid trail plots: Skid trails were formed during the fall and winter of 2008 by skid steer (Figure 3). Skid trail plots were randomly selected and established for seeding in the spring of 2009. Trails were divided into 20-m segment lengths and each segment was randomly selected in GIS. The placement of plots within the skid trails was constrained to areas with clear evidence of skid trail usage, exposure of bare soil for seeding and >25 m from another plot. There were 15 replicates for each seeding treatment (no seed, cultivar and local). The experiment is robust but unbalanced (15 plots each no seed, cultivar, local and 20 no trt) due to limited area available to meet all plot selection criteria as noted above.

2.2. Sampling

Cover data were collected in summer 2017 by species using a square 1-m plot frame located at the plot center. Plant cover 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 present in less than 1% of the area was recorded as 0.5%. Cover was also recorded for bare soil, rock (>2 mm), and litter (all dead plant material, e.g., pine needles, bark, and dead grass). Mature trees (>1.37 m tall) and all stumps were tallied, and diameter at breast height or two perpendicular stump diameters were recorded.



Figure 3. Skid trails shortly after juniper removal (left) and after several years of recovery (right). Due to the contract time period, some skid trails were formed at least a year before they were seeded.

2.3. Data Analysis

The cover of each seeded species in 2017 was analyzed separately as a response variable (PSSP, ELEL, and ACMI). The cover of other species was combined into similar functional groups based on prior work [30] (Table 2). Functional groups were based on life history, morphology, and origin. Sandberg bluegrass (*Poa secunda* J. Presl) was analyzed separately to align with other regional studies, and because of its unique early phenological development, small stature and rooting depth [47]. Functional groups used to analyze richness were broader, and included perennial bunchgrasses, perennial forbs, native annual forbs, shrubs and all exotic species.

Table 2. Understory plant functional groups used as response variables, the short code used in graphs, and dominant species for each group from undisturbed controls in 2017. Origin is native unless noted in the group name. Seeded species were analyzed separately (*Elymus elymoides* (ELEL), ACMI, *Pseudoroegneria spicata* (PSSP)). The nomenclature follows Meyers et al. [48] for grasses, and Hitchcock and Cronquist [49] for other species.

Functional Group	Code	Dominant Species
Total plant cover	Total	Sandberg bluegrass (<i>Poa secunda</i>), Thurber's needlegrass (<i>Achnatherum thurberianum</i>), Idaho fescue (<i>Festuca idahoensis</i> Elmer)
Perennial large bunchgrasses	PBG	No seeded species, Thurber's needlegrass, Idaho fescue
Shallow, early season grass	POSE	Sandberg bluegrass
Perennial Forbs	NPF	Low pussytoes (<i>Antennaria dimorpha</i> (Nutt.) Torr. & A. Gray), velvet lupine (<i>Lupinus leucophyllus</i> Douglas ex Lindl.)
Native annuals	NAF	Pacific popcornflower (<i>Plagiobothrys tenellus</i> (Nutt. ex Hook.), small fescue (<i>Vulpia microstachys</i> (Nutt.) Munro)
Exotic grasses	EG	Cheatgrass (<i>Bromus tectorum</i>)
Tall annual exotic forbs	TAEF	Tall tumblemustard (<i>Sisymbrium altissimum</i> L.), littlepod false flax (<i>Camelina microcarpa</i> Andr. ex DC.)
Small annual exotic forbs	SAEF	Jagged chickweed (<i>Holosteum umbellatum</i> L.), spring draba (<i>Draba verna</i> L.)
Shrubs	SHRUB	Antelope bitterbrush (<i>Purshia tridentata</i> (Pursh) DC), Wyoming sagebrush (<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i>), green rabbit brush (<i>Chrysothamnus viscidiflorus</i> (Hook.) Nutt.)

Response variables were analyzed based on a randomized design using linear models in R [50] using the `lm` function [51] and `emmeans` package [52] with treatment (no trt, local, cultivar, no seed) and pre-cut juniper basal area (PJBA) and their interaction as fixed effects. We included PJBA as a covariate in the model as we suspected that the variability in pre-cut juniper tree abundance could influence the long-term vegetation response owing to variation in pretreatment understory species composition and the degree of disturbance due to cutting. Although plots were stratified by pre-treatment juniper cover (high, low), actual juniper basal area from 2008 was used as a covariate with an unequal slopes model. Assumptions of normality and constant variance were checked graphically via residual plots, and transformations were unnecessary. Statistical significance is discussed where p values are less than 0.10, although we avoid strict dichotomous distinctions and investigate post-hoc comparisons regardless of global ANOVA p -values as a priori treatment contrasts are of interest. No adjustments were made for multiple comparison tests.

3. Results

3.1. Slash Piles

We found no evidence that juniper removal followed by slash pile burning and seeding influenced total plant cover in the long term, but strong evidence that seeding had a long-term impact on seeded species cover (PSSP, ELEM and ACMI) (Figure 4). For PSSP, cover was about 3–7% higher in plots seeded with both mixes as compared to the no trt and no seed plots. Cover of ELEM was about 3.5% higher for the local seed plots, 2.5% higher compared to seeding with cultivars. Common yarrow (ACMI) cover was also higher in the local seed plots as compared to the no trt and no seed plots, but the difference was less than 1%. In contrast to patterns for seeded species cover, most native perennial functional groups still had much lower cover in treated plots (cultivar, local, no seed), including native perennial forbs (NPF), all perennial bunchgrasses (PBG), and POSE. Conversely, there was little evidence that pile burning and seeding affected short-lived annual forbs (NAF) or surprisingly, SHRUB cover. We note that cover values for the NPF and SHRUB groups are very small.

In contrast to most native perennial plant groups, exotic grass (EG) cover increased in response to juniper removal and slash pile burning, with an 16% increase on average in areas that were not seeded (Figure 4). However, both seed mixes lowered EG cover compared to no seeding (ca. 10% cover vs. 20%), but EG cover remained the lowest where there was no disturbance (no trt). Tall annual exotic forb (TAEF) cover was only observed in trace amounts across the treatments, and SAEF cover was also relatively low.

In 2017, few plant cover groups showed evidence of a relationship to PJBA. PSSP cover was positively associated with high PJBA ($p = 0.016$, interaction with treatment $p = 0.11$), and EG retained a negative relationship with PJBA ($p = 0.08$, interaction with treatment $p = 0.73$). However, we found little evidence of an interaction between PJBA and any treatment.

There was some evidence that seeding increased total richness, and PBG richness increased by about one to two species (Figure 5) in the long term. A small increase in exotic richness was found for the local seed mix, although it was less than one species. The richness of PBG was positively associated with PJBA ($p = 0.007$), although there was little evidence of a treatment interaction ($p = 0.67$).

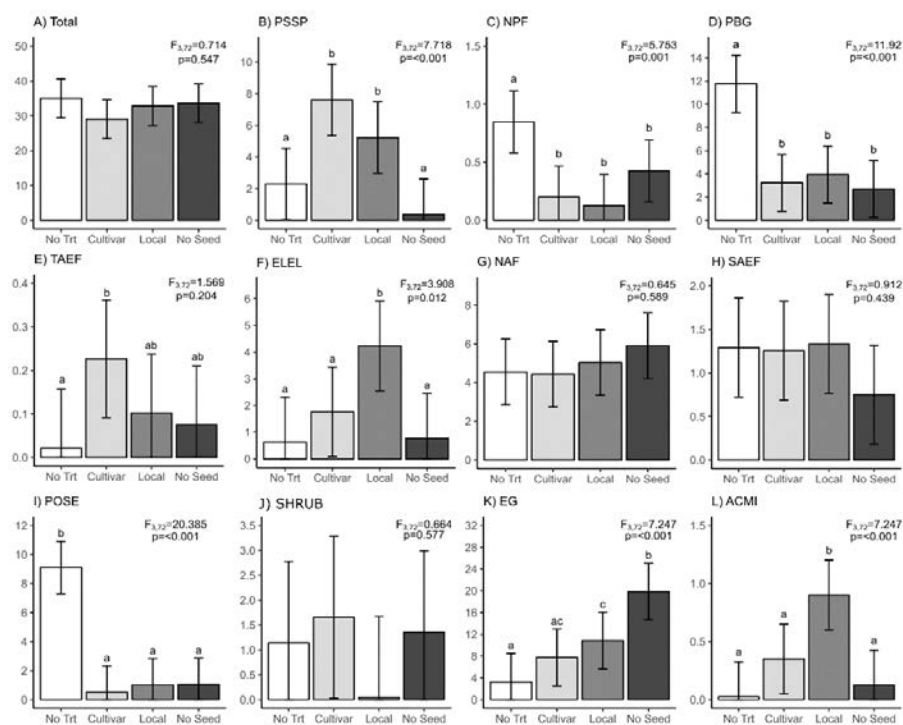


Figure 4. Slash pile functional group mean cover (%) by seeding treatment in 2017: least square means, 95% confidence intervals and ANOVA p -values. Lowercase letters denote significance among treatments (post-hoc contrasts, $p < 0.10$). (A) Total, Total plant cover; (B) PSSP, Psudoroegneria spicata ; (C) NPF, Perennial Forbs; (D) PBG, Perennial large bunchgrasses; (E) TAEF, Tall annual exotic forbs; (F) ELEL, Elymus elymoides ; (G) NAF, Native annuals; (H) SAEF, Small annual exotic forbs; (I) POSE, Shallow, early season grass; (J) SHRUB, Shrubs; (K) EG, Exotic grasses; (L) ACMI, Achillea millefolium.

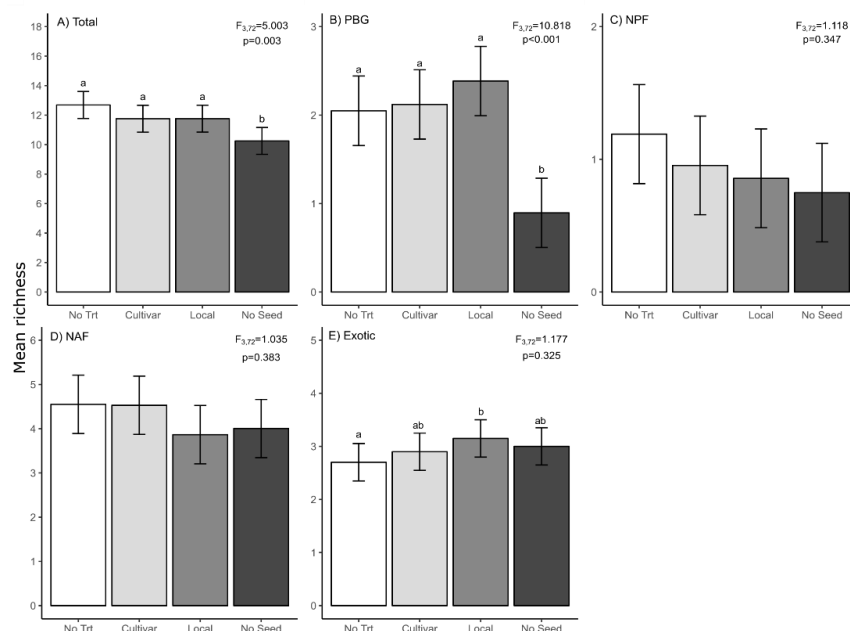


Figure 5. Slash pile functional group mean richness by seeding treatment in 2017: least square means, 95% confidence intervals and ANOVA p -values. Lowercase letters denote significance among treatments (post-hoc contrasts, $p < 0.10$). All perennial bunchgrasses (PBG), native perennial forbs (NPF), native annual forbs (NAF), all exotics (Exotic). (A) Total, Total plant cover; (B) PBG, Perennial large bunchgrasses; (C) NPF, Perennial Forbs; (D) NAF, Native annuals; (E) Exotic, all exotics.

3.2. Skid Trails

We found no evidence that skid trail disturbance and seeding influenced total plant cover in the long term (Figure 6), although there was strong evidence that squirreltail persisted in the long term. Squirreltail cover was 2.5–3.5% higher in the locally seeded plots compared to no seed and no trt (Figure 6). However, PSSP seeding on skid trails did not increase cover in the long term. The response of ACMI depended on PJBA, and in high PJBA areas, the local seed mix had about 3% higher cover compared to the cultivar (Figure 6). The cover of PBG was about 7% lower in plots that were seeded with either mix compared to the no seed treatment. Native perennial forbs (NPF) and SHRUBS showed little difference in cover compared to areas that were not disturbed. However, POSE cover remained 6–7% lower owing to juniper removal and skid trail disturbance (Figure 6). The opposite pattern was found for native annual forbs (NAF), as cover was 4.5–6% higher in the treated plots (cultivar, local and no seed).

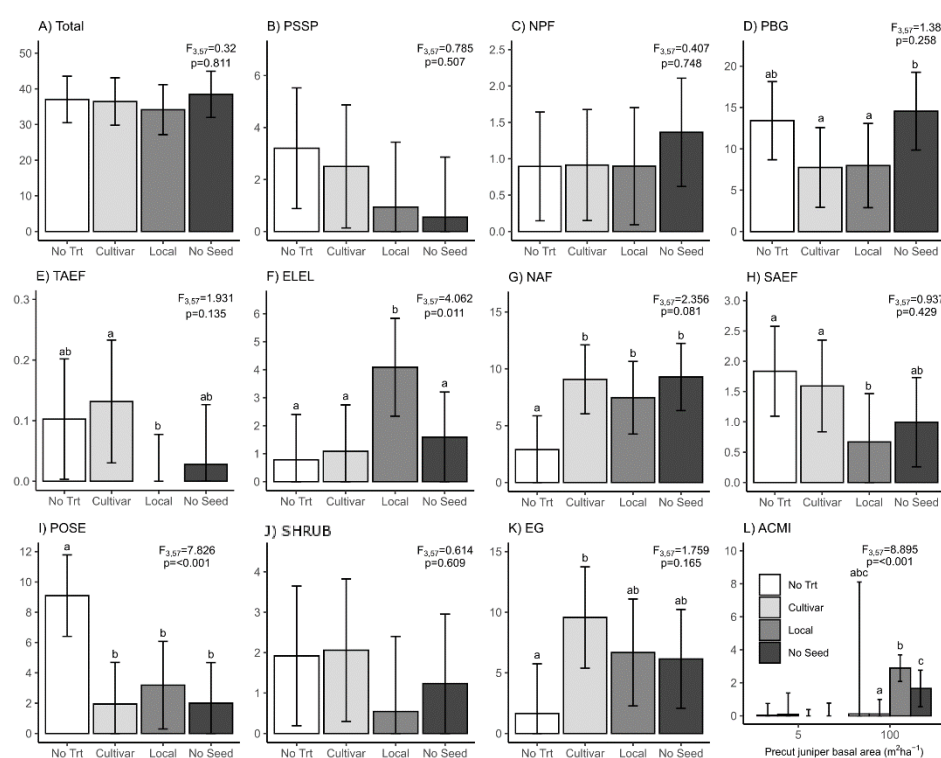


Figure 6. Functional group mean cover (%) by seeding treatment in 2017 for the skid trail experiment: least square means, 95% confidence intervals and ANOVA p -values. Lowercase letters denote significance among treatments (post-hoc contrasts, $p < 0.10$). The ACMI response variable had a significant interaction with precut juniper basal area, as shown in Panel L. (A) Total, Total plant cover; (B) SSP, *Psuedoroegneria spicata*; (C) NPF, Perennial Forbs; (D) , Perennial large bunchgrasses; (E) TAEF, Tall annual exotic forbs; (F) ELEM, *Elymus elymoides*; (G) NAF, Native annuals; (H) SAEF, Small annual exotic forbs; (I) POSE, Shallow, early season grass; (J) SHRUB, Shrubs; (K) EG, Exotic grasses; (L) ACMI, *Achillea millefolium*.

There is evidence exotic grass cover increased 4.5–8% in the skid trails as compared to no trt areas, and neither seed mix mitigated this increase. In fact, the cultivar seed mix had the highest EG cover (ca. 10%). While there was evidence of differences among the treatments, tall annual exotic forb (TAEF) cover was only observed in trace amounts across the study area, and SAEF cover was also relatively low.

In 2017, several plant cover groups still showed evidence of a relationship to PJBA. Positive relationships with PJBA were found for TAEF, SAEF, and SHRUB ($p = 0.004, 0.007, 0.04$, respectively),

while the relationship was negative for NAF ($p = 0.05$). However, PJBA did not interact with treatment response.

We found some evidence that not seeding actually increased species richness slightly in the skid trails, although not seeding slightly decreased PBG richness (Figure 7). Perennial bunchgrass (PBG) richness demonstrated a negative relationship to PJBA, while Exotic richness showed the opposite pattern ($p = 0.09, 0.007$, respectively), but the covariate did not interact with treatment for either response.

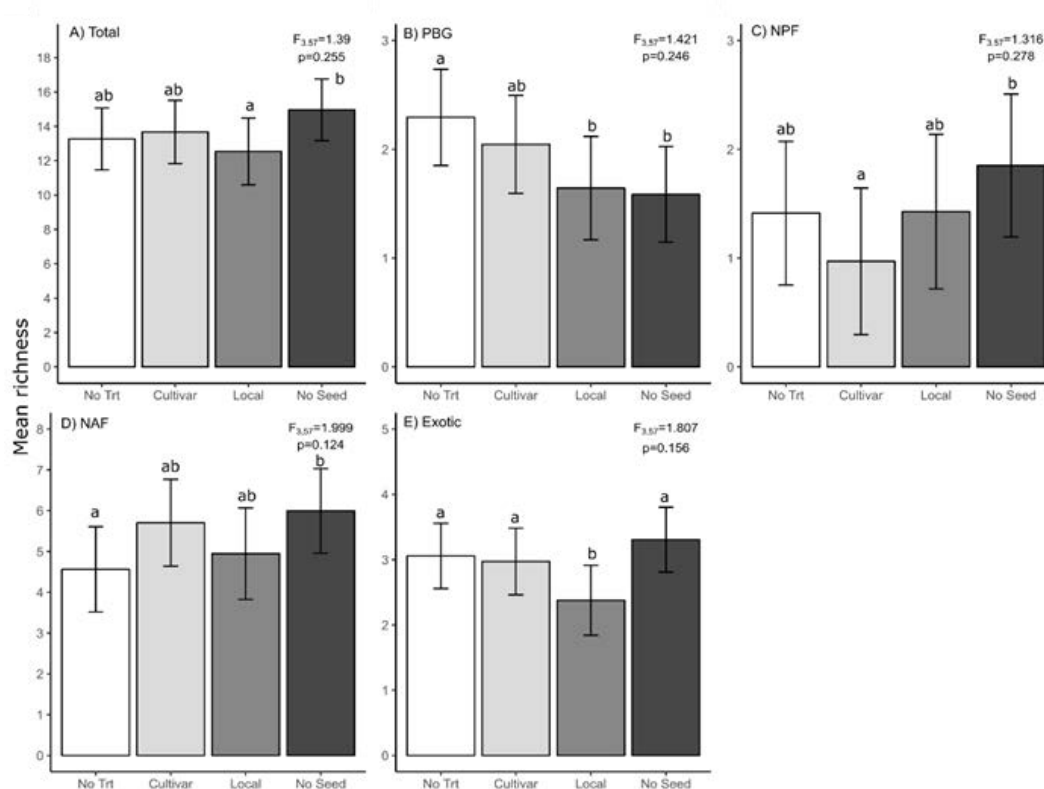


Figure 7. Skid trail functional group mean richness by seeding treatment in 2017: least square means, 95% confidence intervals and ANOVA p -values. Lowercase letters denote significance among treatments (post-hoc contrasts, $p < 0.10$). All perennial bunchgrasses (PBG), native perennial forbs (NPF), native annual forbs (NAF), all exotics (Exotic). (A) Total, Total plant cover; (B) PBG, Perennial large bunchgrasses; (C) NPF, Perennial Forbs; (D) NAF, Native annuals; (E) Exotic, all exotics.

4. Discussion

In this study, we examined the long-term effects of locally sourced and cultivar seeding in slash piles and skid trails after the mechanical removal of western juniper. Long-term effects were measured in 2017, eight growing seasons after seeding was done, in an effort to facilitate the cover of native species, suppress exotic weeds, and enhance native plant diversity after management activities. Similar to others, our findings indicate that seeded species persisted in the long term, especially after burning [33,53–55]. However, patterns and the potential importance of seeded species persistence depended on the species, seed source, and type of disturbance. Regardless of seed source, the large perennial bunchgrass, bluebunch wheatgrass, only established [30] and persisted in 2017 in the slash piles. In slash piles, available nutrient levels can be elevated for several years or longer [28,56], which may explain the more robust patterns of establishment and persistence after pile burning compared to seeding in skid trails. The other two “weedy” native seeded species bottlebrush squirreltail and common yarrow demonstrated both short-term establishment (especially yarrow) and long-term persistence in both slash piles and skid trails. In contrast to the short-term results, cover values for common yarrow in

2017 were very low, and longer-term persistence is questionable. Interestingly, seed source mattered for both bottlebrush squirreltail and common yarrow in 2017, with the local seed source demonstrating long-term persistence while the cultivar did not. This suggests that the locally derived seeds may be better adapted to the environmental conditions at this site. This seed source difference was not found for the short-term results previously reported [30].

In addition to displaying different patterns related to seeded species persistence, native plant recovery and exotic species patterns were also different for the two disturbance types. Eight growing seasons after disturbance, native perennial bunchgrasses and forbs had largely recovered in areas disturbed by skid trails. However, in slash piles, native perennial bunchgrasses and forbs still had lower or trace amounts of cover as compared to areas that had no disturbance activities, although most species groups had increased in cover as compared to short-term results [30]. Unlike skid trail formation, which can result in some plant survival, slash pile burning resulted in complete plant mortality. In contrast, we found exotic species cover was almost twice as high in slash piles that were not seeded as compared to skid trail areas. While the available nutrients in slash piles may facilitate seeded species establishment and persistence, exotic species invasion can also be problematic in these areas [27,29], which may also constrain native perennial bunchgrass and forb recovery.

Unlike our short-term results [30], we found no evidence that seeding increased total plant cover in the long term or the recovery of other plant functional groups for either disturbance type. For the skid trail study, we have some evidence that seeding may have suppressed the abundance of other perennial bunchgrass species. In addition, seeding had mixed but very small impacts on total species richness. These combined results suggest that while seeding can have long lasting impacts on seeded species cover, there is little evidence that seeding facilitated broader community recovery in the long term.

We did find evidence that seeding can suppress exotic annual grass species (largely *Bromus tectorum*) in the long term, particularly after slash pile burning. Similar to other results, this suppressive effect was not detected in the short term [30,53,57]. The long-term results found here are likely due to the increase in bluebunch wheatgrass in slash piles between 2011 and 2017, as other work has demonstrated that perennial grasses can outcompete exotic annuals [58,59]. In addition, the short-term success of seeded species common yarrow (average cover exceeding 60% on some plots, [30]) may also have facilitated a longer-term reduction in exotic grasses.

As noted above, the cultivar and local seed sources that we tested performed differently in the long term. In 2017, squirreltail showed a strong treatment effect, with cover being twice as high in locally seeded plots relative to plots seeded with the cultivar. This pattern for squirreltail held true in both skid trails and slash piles. The low cover of squirreltail for the cultivar treated skid trail plots may help explain the higher exotic grass cover for this treatment. Common yarrow also had a higher cover in slash piles for the local seed mix. While some managers suggest that cultivars are more aggressive and competitive compared to locally generated species [60], our findings do not strongly support this, particularly for the two early successional species used in our seed mix. On the other hand, cover of bluebunch wheatgrass was somewhat higher for the cultivar seed source, although evidence for this was not strong ($0.10 < p < 0.15$).

Our short-term results did not necessarily predict long-term patterns, particularly for the suppressive effect of seeding on exotic grass. In a 25-year restoration study, Copeland et al. [57] found that, five years post-treatment, the results underestimated the longer term suppressive effect of seeding on non-native exotic cover. Similarly, in a long-term remeasurement study, exotic suppression was not detected until 15 years post-seeding [53]. While short-term results were not necessarily insightful for the longer terms pattern observed in this study, the temporal patterns we can glean by comparison to our earlier work do reflect ecological succession and species life history traits. Since 2011, cover of both bluebunch wheatgrass and squirreltail doubled in seeded plots, while common yarrow decreased markedly, especially in slash piles. This is expected given that common yarrow was selected as a “weedy” native, for its ability to spread rapidly in disturbed areas, and potential to outcompete exotic annual grasses [45], whereas squirreltail and especially bluebunch wheatgrass are both slower

growing. In addition, other exotic species groups that were reported earlier were virtually nonexistent by 2017 (small annual exotics and tall exotic forbs). Lastly, we found that pre-cut juniper abundance interacted with treatment for fewer response variables in the long term. This suggests that the effect of pre-treatment differences in understory species owing to juniper abundance may wane through time.

5. Conclusions

The results of our study contribute to the growing body of literature indicating that native plant seeding can result in long-term seeded species persistence and suppression of exotic invasive species such as annual grass. Our results also highlight that short-term seeding results may not predict longer-term outcomes, but patterns through time may be somewhat predictable based on species life history traits and ecological succession. Using a mix of species with different life history traits may have helped to provide better long-term outcomes in terms of suppressing exotic grass, and locally adapted seed sources performed as well as or better than cultivars. However, the suppressive effect of seeding that we documented was not dramatic, but may have been influenced by the fact that we only seeded the study plots and not the broader landscape or all disturbed areas. That is, seeding all the skid trials might have led to a larger suppressive effect and future studies and applications may consider more extensive seeding. However, seeding large portions of the landscape may be cost prohibitive, especially when local native seed is being used. This issue may be less relevant for the slash pile experiment as piles only represented a very small portion of the landscape and were more spatially discrete.

There are several considerations we note regarding inferences and implications for our study. As reported by Kerns and Day [30], the weather patterns associated with seeding our plots were optimal for plant germination and establishment. It is likely that our results regarding long-term persistence hinged upon these ideal initial conditions. In addition, while our experiments were statistically robust and report relatively uncommon long-term outcomes, we lacked replication across the landscape. Therefore, inferences from our results should be cautiously interpreted, particularly for sites with substantially different conditions and for different types of seed mixes.

While we found that seeding reduced exotic grass cover in the long term, the invasion levels we report are still problematic and may have impacts on biodiversity, forage and patterns related to fire behavior. Exotic grass invasion after tree removal can increase the surface fire potential [61] and cover values as low as 5% have been associated with an increased fire risk in some ecosystems [62]. A loss of native biodiversity has been found after only one fire with invasive annual grasses present [63,64]. More aggressive weed treatments before and after conifer removal treatments and wider seeding application may be needed to effectively treat exotic populations after conifer removal and ground disturbance activities.

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