Received: 8 June 2016 | Accepted: 31 August 2017
DOI: 10.1111/avsc.12340

RESEARCH ARTICLE

Applied Vegetation Science IAVS

Seeding alters plant community trajectory: Impacts of seeding, grazing and trampling on semi-arid re-vegetation

Hannah L. Farrell | Jeffrey S. Fehmi

School of Natural Resources and the Environment, University of Arizona, Tucson, AZ, USA

Correspondence
Hannah L. Farrell, School of Natural Resources and the Environment, University of Arizona, Tucson, AZ, USA.
Email: hlfarrell@email.arizona.edu

Funding information
Sierrita Gas Pipeline LLC

Co-ordinating Editor: Norbert Hölzel

Abstract

Questions: How do seeding, cattle grazing, and vehicular use impact vegetation establishment and soil movement on a newly reclaimed pipeline right-of-way? Will these factors result in differing plant community trajectories?

Location: Southern Arizona (USA).

Methods: Within a pipeline disturbance, we randomly selected nine plots to be seeded with an 18 species mix and nine to be left unseeded. Adjacent to the disturbance, we selected nine undisturbed unseeded control plots for a total of 27 plots (30 m × 45 m each). Within each of the 27 plots, we established a grazed–trampled, grazed–untrampled and ungrazed–untrampled subplot. One year after pipeline reclamation, we analysed the impacts of seeding, grazing and trampling on native plant cover, undesirable plant cover, herbaceous biomass, species richness, soil movement and plant community trajectories in comparison to surrounding undisturbed sites.

Results: Seeding disturbed sites with a diverse seed mix resulted in greater native plant cover, higher species richness and fewer undesirable species than were found in unseeded disturbed sites. Unseeded disturbed areas were similar to the undisturbed control areas in species richness and had comparable plant community trajectories. The combined impacts of grazing and trampling reduced native plant cover and herbaceous biomass and were associated with increased soil erosion in comparison to subplots protected from grazing and trampling.

Conclusions: Natural vegetation recruitment can be a viable option in semi-arid reclamation projects when the soil seed bank is preserved and there are proximal seed sources. While seeding improved quantitative vegetation metrics, using a seed mix comprised of different species than the preexisting vegetation may set the reclaimed vegetation on a different plant community trajectory. The general prescription of protecting new reclamation sites from grazing and trampling is supported.

KEYWORDS
erosion, invasive species, land management, priority effects, reclamation, restoration ecology, seed sources, southwestern US, vegetation communities, vegetation establishment

1 | INTRODUCTION

Arid and semi-arid lands worldwide experience land degradation from natural resource extraction and the accompanying infrastructure expansion (Lambin & Geist, 2006). These industries typically have both the capability and the responsibility for at least partially reclaiming land degraded by these uses. From the industrial perspective, understanding the best practices for reclaiming degraded lands is critical to fast, efficient and publically acceptable

Nomenclature: The PLANTS Database (http://plants.usda.gov) reclamation. Arid and semi-arid grasslands have been particularly challenging for reclamation because they can be slow to recover vegetation naturally (Lathrop & Archbold, 1980), experience widely varying spatial and temporal rainfall patterns that limit vegetation establishment (Hoffman, Barr, & Cowling, 1990) and have soil fertility and moisture-holding characteristics that can further arrest plant development (Bainbridge & Virginia, 1990). Without active reclamation, some disturbed arid and semi-arid sites recover well while others may take centuries for post-disturbance recovery to resemble pre-disturbance plant communities (Abella & Newton, 2009; Berry, Weigand, Gowan, & Mack, 2016).

Direct seeding remains one of the most common arid and semi-arid land reclamation treatments because of its low cost and applicability to large-scale disturbances (Bainbridge, 2007). While common use would imply the established effectiveness of direct seeding to yield desirable vegetation communities, research has produced mixed and inconclusive results (e.g. Cox et al., 1982). In some instances, the priority of the reclamation project has been to quickly maximize vegetation cover on a disturbed site, regardless of species composition, in order to prevent soil erosion (Whisenant, 1999). While preventing erosion remains important, there is also interest in examining successful methods of more passive reclamation and understanding interactions between introduced seeded species and those species naturally colonizing from surrounding areas (Baasch, Kirmer, & Tischew, 2012).

The order and timing of species arriving to a disturbed site can have long-term impacts on the community trajectory and composition. These effects have been termed “priority effects” because the plant species that establish first can suppress later arriving plant species (Drake, 1991). Priority effects can be a factor of species’ reproductive strategies; in arid regions early germination of native or exotic plants can provide a competitive advantage (Stevens & Fehmi, 2011; Wainwright, Wolkovich, & Cleland, 2012). To forestall invasive and other undesirable species, seeded species are typically added shortly after the disturbance ends because desired species may have limited ability to quickly disperse into disturbed sites (Bossuyt & Honnay, 2008). Seeding may not entirely prevent undesirable species because the movement of people and equipment often transports exotic plants during construction, maintenance and the recreation that follows improved access (Hansen & Clevenger, 2005).

On-going land uses such as recreation and livestock production, both common uses of arid and semi-arid lands, can impact land being reclaimed. Trampling by cattle (independent of forage consumption) can reduce vegetation ground cover and biomass (Dunne, Western, & Dietrich, 2011). Livestock grazing that occurs before seedlings are well established on arid reclamation sites has been shown to reduce biomass and decrease vegetation community diversity (Whisenant & Wagstaff, 1991). Even after seedlings establish, grazing of early-seral seeded communities has slowed successional recovery, whereas grazing had few impacts on mid- and late-seral communities (Milchunas & Vandever, 2014). Conversely, ungulate grazing has also been found to positively impact some restoration sites (e.g. Martin & Wilsey, 2006). Impacts of grazing may depend on existing site conditions: grazing on poorly productive sites can decrease species richness while grazing on highly productive sites can increase species richness (Lezama et al., 2014).

We hypothesized that seeding a disturbed area with a diverse mix of desirable native species would result in greater native plant cover, less undesirable plant cover and increased species richness as compared to an unseeded area (hypothesis 1). We expected that unseeded plots would yield a different plant community assemblage than undisturbed control plots, testing the ability of native plants to migrate from the adjacent undisturbed areas (hypothesis 2). We anticipated that grazing would decrease the native vegetation cover (%) and change the plant community assemblage (hypothesis 3). Areas being trampled in this study (lumped cattle trampling and vehicle use) also inherently include the impacts of grazing; therefore we expected the impacts of grazing and trampling in combination to be greater than the impacts of grazing alone (hypothesis 4).

2 | METHODS

2.1 | Site description

During Aug–Sept 2014, a 96 km natural gas pipeline was constructed between southwest Tucson, Arizona and the border of Mexico at Sasabe, Arizona, USA (Figure 1). The study site was a 2.5-km section of the 30-m wide pipeline construction zone roughly 48 km south of Tucson (32°15′12.4560″ N, 110°54′42.4404″ W). The site was nearly flat (<5 m difference in elevation throughout the site) with silt loam soil at 775 m a.s.l. The vegetation was fairly species-poor and consisted primarily of the woody shrubs *Prosopis velutina* Wooton, *Atriplex canescens* (Pursh) Nutt, and *Vachellia constricta* (Benth.) Seigler & Ebinger in the overstorey, with sparse cacti, annual grasses and annual forbs making up the understorey (see Appendix S1 for photograph; botanical nomenclature throughout per USDA, NRCS, 2016). The study region historically had heavy over-grazing by cattle since the early 1900s. The study site continues to be moderately grazed by cattle (stocking rate of approximately 2.5 cattle 100/ha) during the winter months (December through February). The past grazing likely caused a shift from a native perennial grassland to a shrub-dominated woodland (Browning & Archer, 2011). The study site receives an average of 360 mm annual precipitation, of which approximately 53% (190 mm) is during the hot summer monsoon season, with the remainder in winter (WRCC, 2015). For the duration of the approximately year-long study (Oct 2014–Sept 2015), the study site received 390 mm of precipitation, which includes 212 mm precipitation during the summer 2015 monsoon season (NOAA, 2015).

2.2 | Experimental design

The study site was divided into three homogenous sections to avoid confounding impacts from wash drainages. In Oct 2014, nine 30 m × 45 m plots were randomly selected to remain unseeded and nine 30 m × 45 m plots to be the seeded (Figure 2). Nine control plots (30 m × 45 m) were established adjacent to the nine unseeded plots in desert areas unaltered by pipeline construction (Figure 2). Since we

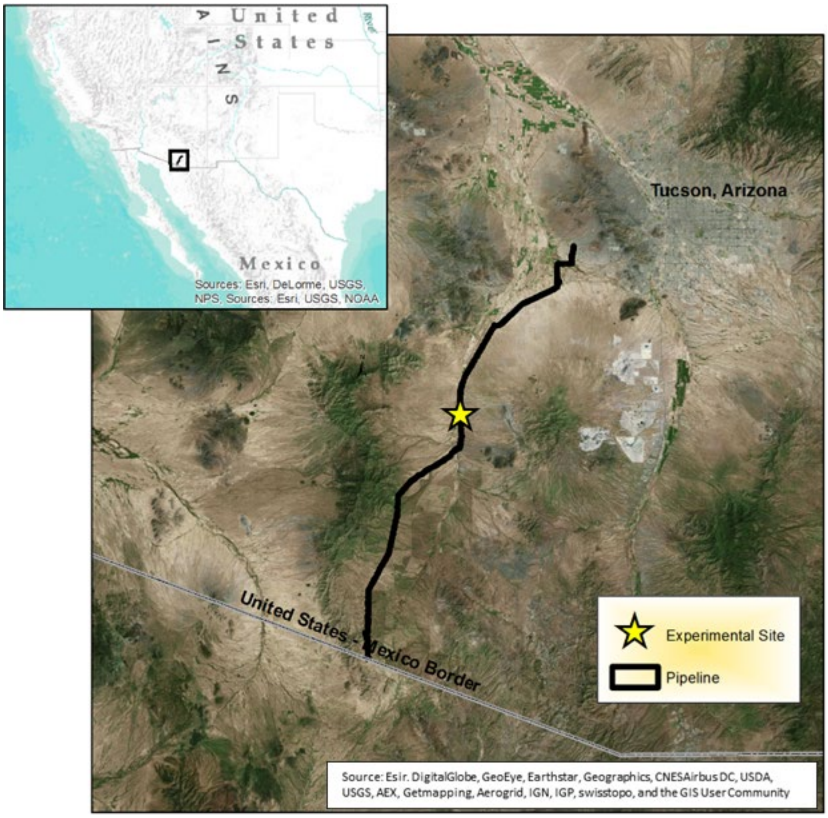

FIGURE 1 The pipeline corridor spans between Tucson, Arizona and the border of Mexico through the Sonoran Desert in Southern Arizona [Colour figure can be viewed at wileyonlinelibrary.com]

intended to compare the vegetation emerging from the pipeline reclamation treatments to the vegetation of the surrounding undisturbed desert areas, the control plots were established adjacent to unseeded plots to reduce the likelihood of unintentional seeding and to comply with land access restrictions during pipeline construction.

Within each of the 27 treatment plots (nine seeded, nine unseeded, nine control), we established three subplots: an exclosure that prevented livestock grazing (2.0 m × 1.5 m); an exclosure that prevented trampling by people, vehicles and large animals but allowed livestock grazing (2.0 m × 1.5 m); and an open subplot exposed to grazing and trampling (2.0 m × 1.5 m). The 2.0 m × 1.5 m size subplot was chosen because it was the largest size that supported both research and land-user concerns. The grazing exclosures were fenced with approximately 5-cm mesh to keep cattle and large wildlife out, whereas the trampling exclosure was a bare structure of steel poles (without fencing) that allowed cattle or other large grazing animals to extend their heads into to graze, but kept out vehicles and cattle trampling (see Appendix S2 for structure details). When the subplots were constructed, four 15 cm nails were hammered into the centre of each subplot treatment (2.0 m × 1.5 m) at an interval of 0.3 m. The nails were inserted to the depth at which the nail heads were flush with the ground surface, allowing us to approximate whether the soil had accumulated or eroded from the initial surface level.

2.3 | Reclamation and seeding assumptions

The natural gas pipeline construction activity was confined to a 30–40-m wide construction zone termed the right-of-way (ROW). The reclamation practices used on the pipeline ROW were chosen to fulfill the US Federal Energy Regulatory Commission requirements. The main concerns addressed were soil erosion, vegetation impacts (including potential forage loss for on-going cattle grazing), loss of wildlife habitat, impacts to endangered species and increased unauthorized access. In Aug and Sept 2014, the sequence of construction for the pipeline ROW was: clearing vegetation, scraping the upper 10 cm of topsoil and segregating it on the east side of the ROW, excavation of a 3-m wide trench in the centre of the ROW, laying a 91-cm diameter pipe, backfilling the trench and spreading topsoil back across the entire ROW. For the seeded plots, seeds were drilled into 19-mm deep furrows at a rate of 4.01 kg/ha pure live seed during the first 2 weeks of Sept 2014. The seed mix species were selected by the pipeline company and included 18 species of grasses, forbs and shrubs native to the southwestern USA and sourced from regional native plant stock (USDA, NRCS, 2016; Table 1). The functional groups present in the seed mix by seed weight were: 40.8% woody species, 28.6% perennial grasses, 8.1% perennial forbs and 21.9% annual forbs. The unseeded plots were subjected to the same construction processes as the seeded portions of the ROW, except that no seeding occurred.

2.4 | Sampling methods

Following monthly site visits sampling was conducted between 28 Sept 2015 and 12 Oct 2015 (post-2015 monsoon peak biomass production), which allowed a full growing season for both the winter annual species and the summer perennial species. A 0.16-m^2 quadrat

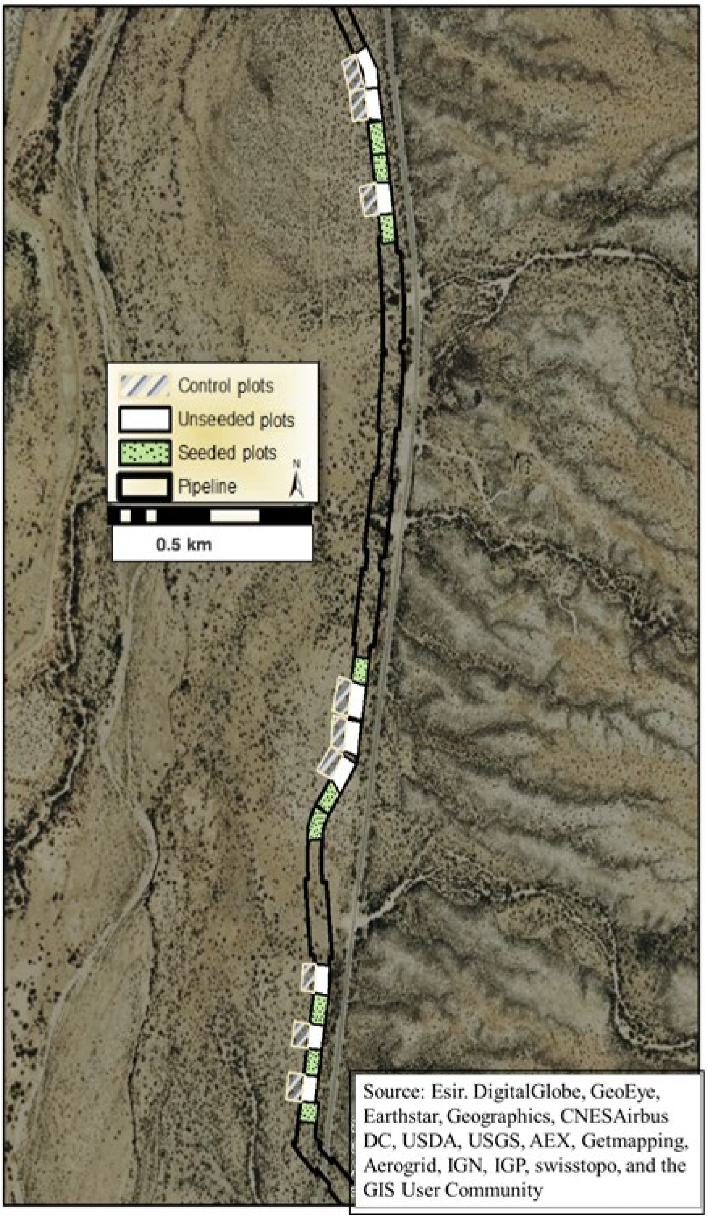

FIGURE 2 Experimental design along the 30-m wide pipeline corridor. The entirety of the study site runs approximately 2.5 km in length and is located on public land that is leased for grazing by local ranchers [Colour figure can be viewed at wileyonlinelibrary.com]

was used to estimate total percentage aerial cover by species, density by species and above-ground herbaceous biomass for each subplot treatment (three subplot treatments per 30 m × 45 m treatment area; Figure 2; N = 80; one subplot discarded due to rodent damage). A pilot study was conducted to ensure the 0.16-m^2 quadrat size was appropriate to measure density at this site and this quadrat size matched the standard for the region (Coulloudon et al., 1999). Above-ground herbaceous biomass (dead and live) was clipped to the soil surface and dried for 48 hr at a 70°C. The vertical difference between the soil surface and the head of each erosion nail was estimated (to the nearest mm).

Species designated as "native species" were native to the southwestern US region (USDA, NRCS, 2016) and included all seed mix species (Table 1) as well as all species naturally occurring in undisturbed areas of the research site (Appendix S3). Species designated as "undesirable species" were those not present on the site before the disturbance (Appendix S3) nor were they a part of the seed mix (laboratory verified). The "undesirable species" designation included common ruderal species and noxious weeds not consistent with site goals and that arrived to the site unintended.

2.5 | Data analysis

Total vegetation cover (%), total biomass (kg/ha), species richness (species 0.16/m^2) and soil movement (mm) were log-transformed to meet normality assumptions (Shapiro–Wilk p = .09, p = .15, p = .14, p = .09, respectively). The data were back-transformed for presentation. The four vegetation response variables (native species cover [%], undesirable species cover [%], species richness [species 0.16/m^2], total herbaceous biomass [kg/ha]) and soil movement (mm) were analysed with a linear mixed-effects nested ANOVA to determine the significance of differences among treatments and interactions.

TABLE 1 Percentage pure live seed (PLS) by seed mass of species found in the commercial seed mix applied to the study as estimated from a seed mix sample in the lab is shown. Species are shown with the designated plant functional group it was classified into for the purposes of this study. As the seeds varied greatly in size and mass, the number of seeds per gram is included for reference. The annual forb species in the seed mix were confirmed to have been growing in the right-of-way (ROW) during the winter season immediately following ROW seeding, but were no longer present during the autumn data collection period

Semi-desert grassland seed mix	PLS (%)	Seeds/g	Functional group
Acacia greggeii	8.2	9.5	Shrub
Lycium andersonii	23.4	88.3	Shrub
Acacia constricta	10.7	6.0	Shrub
Digitaria californica[a,b]	<1	2,163.4	Perennial grass
Setaria macrostachya[a,b]	4.5	673.3	Perennial grass
Bouteloua curtipendula[a,b]	19.2	1,821.2	Perennial grass
Sporobolus cryptanthus[a,b]	<1	11,695.4	Perennial grass
Bouteloua eriopoda	<1	1,278.1	Perennial grass
Leptochloa dubia[a,b]	2.4	1,187.6	Perennial grass
Sphaeralcea ambigua[a,b]	<1	17,660.0	Perennial forb
Baileya multiradiata[a,b]	3.8	1,354.5	Perennial forb
Penstemon sp.[a,b]	3.4	1,324.7	Perennial forb
Salvia columbariae[a]	2.9	946.2	Annual forb
Escholzia californica[a]	3.0	646.8	Annual forb
Plantago ovata[a]	3.8	717.4	Annual forb
Lupinus (2 spp.)[a]	11.6	187.6	Annual forb
Chamaecrista fasciculata[a]	2.0	143.5	Annual forb

[a]Denotes species confirmed to be growing in the seeded portion of the study site within the first year of seeding based on monthly surveys.
[b]Denotes seed mix species that were recorded during the post-monsoon season data collection period.

Once treatment significance was determined, a Tukey–Kramer multiple comparisons test was used to isolate effects. Each plot assigned a reclamation treatment (n = 27) was used as a random effect and the management treatments were nested within them (n = 80 due to one lost subplot). The same analysis was applied to the soil movement data (n = 78 due to three unrecoverable soil nail locations). The data analysis was completed in R v 3.2.3 (R Foundation for Statistical Computing, Vienna, Austria).

Non-metric multidimensional scaling (NMDS) was used to identify trends in vegetation communities among treatments. NMDS simplifies elements of a community into fewer dimensions to allow interpretation and communication of system changes (Gauch, Whittaker, & Singer, 1981). We split the total vegetation cover (%) into six functional groups for NMDS analysis: undesirable species, native perennial grasses, native annual grasses, native perennial forbs, native annual forbs and native woody species. The reclamation treatments (seeded, unseeded, control plots) were analysed and plotted separately from the land management treatments (grazed–trampled, grazed–untrampled, ungrazed–untrampled subplots). The NMDS data analysis was completed in SAS 9.4 (SAS Institute, Cary, NC, USA) and plotted in R v 3.2.3.

3 | RESULTS

The reclamation treatments (seeded, unseeded, undisturbed control) showed a significant response for native plant cover (%), undesirable plant cover (%), species richness and herbaceous biomass (Table 2). The land management treatments (grazed–trampled, grazed–untrampled, ungrazed–untrampled) showed a significant response for native plant cover (%), herbaceous biomass and soil movement (mm) (Table 2). The interaction between the reclamation treatments and the land management treatments were not significant (p > .12) for any vegetation or soil response variables (Table 2).

Seeding the reclamation site resulted in greater native plant cover (%), higher presence of the perennial grass and perennial forb functional groups than the undisturbed desert control (Figure 3a). Of the total native plant cover within the seeded sites, approximately 64% were seed mix species; the remaining 36% were species naturally recruited from the seed bank and surrounding areas. Seeding also resulted in significantly higher species richness than both the unseeded areas and the undisturbed control sites (Figure 3c).

The unseeded reclamation sites resulted in native plant cover (%), function group composition (Figure 3a) and species richness (Figure 3c) similar to the undisturbed control sites. Reclamation without seeding resulted in significantly higher cover (%) of undesirable species (Appendix S3 for species list) than the undisturbed desert areas (Figure 3b). Seeding reduced the cover of undesirable species, but was not significant. Both seeded and unseeded reclaimed sites produced significantly more biomass than the undisturbed control desert sites (Figure 3d).

Excluding both grazing and trampling resulted in increased native plant cover (%; Figure 3a), higher species richness (Figure 3c) and larger herbaceous biomass (Figure 3d) compared to areas exposed to both grazing and trampling. However, excluding both grazing and trampling did not significantly alter the functional group composition (Figure 3a) or reduce the cover of undesirable species (Figure 3b). Grazing alone (without trampling) did not result in any statistical differences, as the results tended to be intermediate between protected sites and exposed sites.

Seeding did not significantly impact soil movement (p > .75 for all combinations); seeded areas lost an average of 0.1 ± 1.4 mm of soil, unseeded areas accumulated an average 0.5 ± 0.7 mm of soil, and control plots accumulated an average 1.2 ± 0.4 mm of soil. Areas exposed to combined grazing and trampling experienced higher soil erosion than areas protected from both grazing and trampling; exposed areas lost an average of 1.1 ± 0.6 mm of soil whereas areas protected from grazing and trampling accumulated an average of 1.7 ± 1.1 mm of soil (p = .006; average 2.8 mm difference). Grazing alone resulted in an average of 0.9 ± 0.9 mm of soil accumulation, which was not different from exposed areas (p = .08) or protected areas (p = .53).

TABLE 2 ANOVA *F*- and *p*-values from the linear mixed-effects analysis. Reclamation treatments were seeded, unseeded and control plots. Management treatments, nested within the reclamation treatments were grazed–trampled, grazed–untrampled and ungrazed–untrampled subplots

	Native plant cover		Undesirable plant cover		Species richness		Herbaceous biomass		Erosion	
	$F_{(df)}$	*p*	$F_{(df)}$	*p*	$F_{(df)}$	*p*	$F_{(df)}$	*p*	$F_{(df)}$	*p*
Reclamation	$6.38_{(2,24)}$	**.031**	$3.56_{(2,24)}$	**.044**	$10.61_{(2,24)}$	**<.001**	$20.45_{(2,24)}$	**<.001**	$0.1.5_{(2,24)}$	.743
Management	$3.23_{(2,47)}$	**.049**	$0.541_{(2,47)}$	.586	$3.06_{(2,47)}$	.056	$3.89_{(2,47)}$	**.027**	$5.69_{(2,45)}$	**.006**
Interaction	$0.971_{(4,47)}$	.432	$0.742_{(4,47)}$	.568	$1.96_{(4,47)}$	.116	$0.50_{(4,47)}$	.74	$1.82_{(4,45)}$	.141

Significnat *P*-values less than 0.05 are shown in bold.

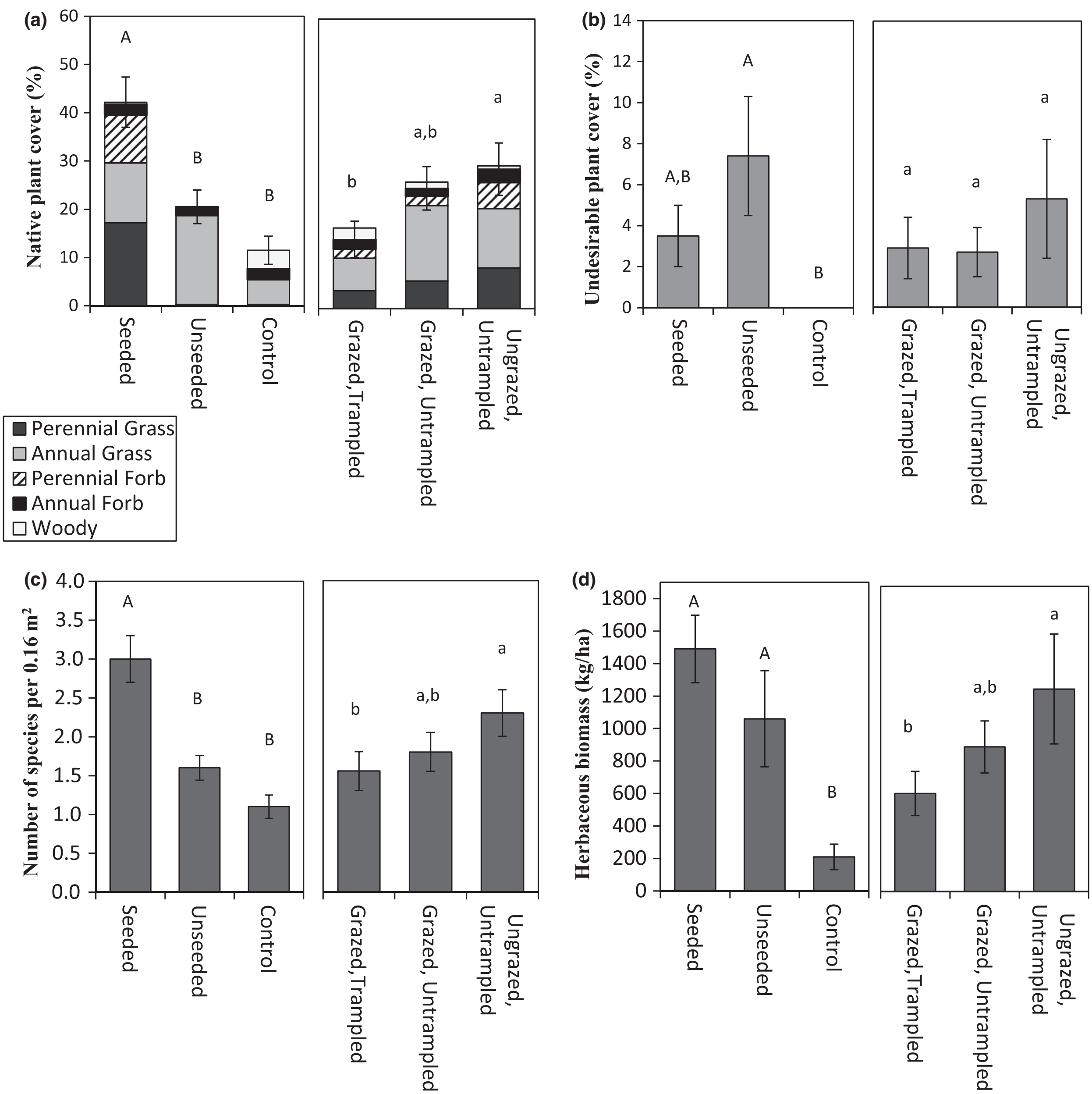

FIGURE 3 Back transformed data (mean ± *SE*) (a) native species cover, (b) undesirable species cover, (c) species richness, (d) total herbaceous biomass. Reclamation treatments significantly different from each other are indicated by uppercase A or B (p = .05); land management treatments significantly different from each other are indicated by lowercase a or b (p = .05)

The NMDS ordination analysis on the plant community trajectories that emerged 1 year after reclamation shows that unseeded reclaimed areas developed an assemblage similar to the undisturbed desert control, whereas seeded reclaimed areas clumped into a different assemblage (Figure 4). The dominant functional group in both the unseeded reclaimed areas and the undisturbed control areas was

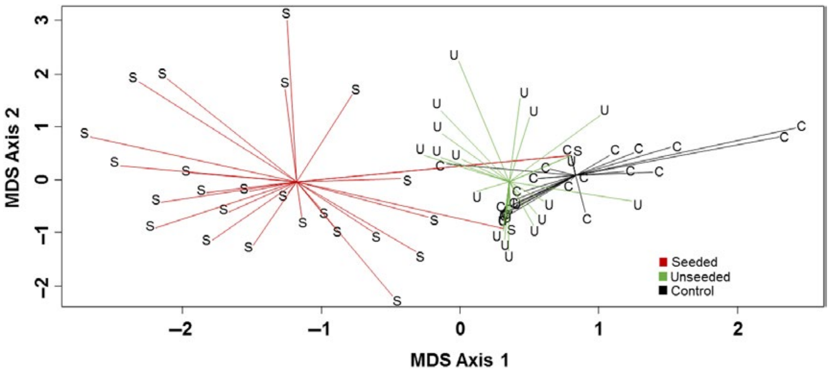

FIGURE 4 Results of the two-dimensional non-metric multidimensional scaling (NMDS) comparing plant assemblages based on ecological functional groups using Euclidean distance of the log +1 transformed data. NMDS was performed with R on the vegetation functional groups disregarding whether the species was naturally recruited or originated from the seed mix. Lines originate from the treatment group centroid to all points within that group (ordispider). Figure shows reclamation treatments (stress = 0.17) with S = seeded plots, U = unseeded plots, C = control plots [Colour figure can be viewed at wileyonlinelibrary.com]

annual grasses (primarily *Bouteloua aristidoides*, an annual grass originating on site), followed by annual forbs to a lesser degree (Figure 3). No seeded species were found in unseeded or control areas. While the seeded areas naturally recruited annual grasses as well (12.4% annual grass cover; primarily *B. aristidoides*), the perennial grass (16.1% seeded perennial grass cover; 1.1% naturally recruited perennial grass cover) and perennial forb (10.9% perennial forb cover; all seeded) functional groups dominated (Figure 3). Both seeded and unseeded reclamation areas had presence of undesirable species, albeit seeded areas had this to a lesser extent. Trampling and grazing did not alter the vegetation community assemblage enough to be distinguishable in NMDS analysis (Figure 4).

4 | DISCUSSION

4.1 | Seeding and vegetation community development

After the first growing season, the vegetation in the unseeded reclamation areas returned to very similar species richness, species composition, native plant cover and functional group assemblage as the undisturbed control plots. Based on other studies of restoration in semi-arid regions (e.g. Stylinski & Allen, 1999; Waller, Anderson, Holmes, & Allsopp, 2016), we had expected our short-term study would result in unseeded areas being worse off in all aspects (hypothesis 1). Passive re-vegetation (not seeding) has been found to be a viable option in arid and semi-arid grasslands throughout the world in several recent longer-term studies. Martinez-Ruiz, Fernandez-Santos, Putwain, and Fernandez-Gomez (2007), in a study on reclaimed mine tailings in mediterranean grasslands, found that the native vegetation from adjacent undisturbed areas showed high capacity to colonize in disturbed areas and maintained or increased presence over 4 years of research. They found significant differences in species composition between seeded and unseeded areas during the first 2 years of research, but the differences were no longer significant after year 2 (Martinez-Ruiz et al., 2007). Fensham, Butler, Fairfax, Quintin, and Dwyer (2016) executed a chronosequenced study on old agriculture fields in Australian grasslands that revealed native annual grasses and forbs quickly established cover on disturbed sites without seeding due to high dispersal capabilities and proximal seed sources. Target perennial grasses had less effective dispersal strategies and recovered more slowly than annuals but nonetheless showed linear recovery patterns (Fensham et al., 2016). Similarly, Deák et al. (2015) found that highly diverse vegetation assemblages naturally colonized a grassland in Hungary within the first year of new soil installation. They compared older soil-filled channels to 1-year old channels and found that older channels developed to be dominated by perennial species rather than ruderal annual species over time. A common theme of successful natural vegetation establishment occurs with the disturbed sites having nearby seed sources; our narrow, linear pipeline corridor provided ideal proximity between the disturbance and the adjacent undisturbed vegetation.

We attribute the high degree of natural colonization measured in unseeded areas to the viable seed persistence of native species in the stockpiled surface soil and the ability of these species to readily disperse onto the narrow disturbance corridor from adjacent intact desert. Scott and Morgan (2011) found that seed-rain recruitment was a more important mechanism than the seed bank for recolonizing disturbed sites in disturbed semi-arid grasslands of Australia. They additionally found that ruderal, early-successional annual grassland species generally dominated the seed bank. Similarly, Bertiller and Aloia (1997) found that perennial grasses had transient seed banks while annuals (grasses and forbs) had persistent seed banks in semi-arid grasslands in Patagonia. In response to disturbance, the annual species with persistent seed banks were able to colonize more successfully (Bertiller & Aloia, 1997), perhaps due to more opportunistic reproductive strategies (Dyer, Hardison, & Rice, 2012). Both our seeded and unseeded reclaimed areas produced more herbaceous biomass and total plant cover than the undisturbed desert areas, which at least implies that the site was in poor condition prior to the pipeline construction. The study site likely had low ground cover and productivity compared to its potential because this region, as with much of southern Arizona, has experienced altered fire and grazing regimes that caused historic semi-desert grasslands to be converted into shrub-dominated vegetation with increased bare ground (Browning & Archer, 2011). Mesquite (*Prosopis* spp.) encroachment has resulted in uneven distribution of topsoil and soil nutrients where patches of vegetation can survive under the shrub canopy, but the interspace between shrubs becomes increasingly barren and devoid of topsoil and nutrients (Yavitt & Smith, 1983). We hypothesize that the new vegetation on the ROW could have been stimulated by the removal of competing mesquite and cactus species during construction, similar to the findings of Hessing and Johnson (1982) who studied the recovery of a different Sonoran Desert pipeline corridor. Improved vegetation biomass in reclaimed areas compared to undisturbed desert areas could also have been due to the soil disturbance that occurred during construction and reclamation, which has been shown to increase soil infiltration and plant productivity in arid grasslands (Miyamoto, Olson, & Schuman, 2004). Another possible mechanism to explain the reclaimed areas producing significantly more biomass than the undisturbed desert areas is that the soil disturbance could have stimulated increased availability of soil nutrients from the soil microbial pools, similar to the results of soil tilling (Kristensen, Kasia, & McCarty, 2003).

In our study, while seeding produced higher ground cover and species richness than not seeding, the species composition and community assemblage more closely resembling that of the seed mix rather than that of adjacent undisturbed areas (approximately 2/3 or the ground cover was from seed mix species and 1/3 was naturally recruited). This supports our hypothesis that seeding would recover towards a different plant community trajectory than not seeding (hypothesis 2). While the species making up the seed mix are all native to the southwestern US region as well as being desirable rangeland plants, they do not match the local species at the research site. The priority effects of seeded species establishing first may lead to long-term differences in vegetation communities compared to the surrounding desert communities (e.g. Belyea & Lancaster, 1999). In further support of priority effects being a useful principle in drylands, Walker and Powell (1999) measured the differences between seeded and unseeded roadsides in the Mojave Desert after 4 years and found that unseeded areas recovered to similar species richness and community composition as undisturbed areas, whereas seeded areas were dominated by seed mix species. Their community composition results closely resemble our initial outcomes and suggest that the fast-establishing seeded species excluded natural establishment of on-site species through resource competition, thus seeding may be less effective for replicating pre-disturbance plant communities (Walker & Powell, 1999).

An additional consideration is how the seed mix species might impact the species-poor adjacent undisturbed desert control plots that averaged 1.1 species 0.16/m^2 while seeded plots averaged 3.0 species 0.16/m^2. Baasch et al. (2012), looking at restoration of disturbed post-mining sites in dry grasslands of Central Europe, found that desirable species from seeded sites spontaneously migrated into adjacent unseeded control sites. It is probable that the seeded species from the ROW will migrate into both unseeded ROW areas as well as adjacent undisturbed control areas, but our study was not long enough to detect migration of seeded species. The effect of spontaneous migration might be considered beneficial in this case for increasing species richness and vegetation cover in the adjacent desert areas around our site that had been impacted by historic over-grazing.

4.2 | Impacts of grazing and trampling

Contrary to our expectation (hypothesis 3), grazing alone did not negatively impact native vegetation establishment, soil movement or lead to an increase in undesirable weedy plant presence even though there was moderate cattle use in the study area during the winter months. Davies, Boyd, Bates, and Hulet (2015) similarly found that dormant season grazing on dry shrub grasslands of eastern Oregon only temporarily reduced forb and grass cover and had no negative impact on desirable perennial grass cover or production, nor did dormant season grazing increase invasive species presence. Gornish and Ambrozio dos Santos (2016) examined a California annual grassland seeded with native species 10 years prior and found that grazing led to decreased desirable seeded native plant cover and dominance of invasive species. They hypothesized that drought conditions played a role in the outcome, as the invasive species compete strongly for moisture availability, and the authors suggested that low intensity grazing may be appropriate for seeded sites in non-drought years. Our study period had higher than average rainfall, which may have enabled hardy vegetation growth in the face of grazing; for example, Fynn and O'Connor (2000) found that precipitation was more influential than grazing intensity on semi-arid rangeland productivity in South Africa over a 10-year study period.

Emerging prior to, as well as during, the grazing period of our study (Dec 2014 through Feb 2015), we observed seeded reclaimed areas densely colonized by three rapid establishing winter annual forb species from the seed mix (*Salvia columbariae*, *Escholzia californica*, *Plantago ovata*; Table 1). These seeded annual forbs were no longer present on the ROW during the data collection period (Sept and Oct 2015) due to their phenology. The perennial forbs and grasses that dominated seeded portions of the ROW during our data collection period germinate in the summer in response to monsoon rains, therefore were not exposed to the impacts of winter season grazing. We partially attribute the lack of grazing impacts to this phenological timing.

Our treatment of the combined impacts of trampling and grazing reduced native plant cover, reduced biomass and caused increased soil erosion as compared to the areas that were protected from both grazing and trampling (as expected in hypothesis 4). This finding is supported by numerous other studies. Salihi and Norton (1987) observed that grazing increased perennial grass seedling mortality in semi-arid rangelands of the southwestern US, which the authors primarily attribute to livestock trampling. In a different arid grassland of Arizona, Allington and Valone (2011) documented that cattle grazing and trampling reduced native plant cover and increased soil compaction (bulk density) over the long term. Lezama et al. (2014) had results showing that grazing reduced species richness in grass and shrub lands in study sites ranging across Argentina and Uruguay. In a study simulating cattle trampling on a dry rangeland of Kenya, Dunne et al. (2011) found that trampling reduced plant cover, biomass and plant regeneration.

They also found that cattle trampling increased soil loss, which they attributed to reduced vegetation cover and disturbed topsoil structure.

4.3 | Management implications

Seeding can only be assessed in relation to the goals for the site. The seeded areas could be considered better than unseeded areas from a management or regulatory perspective because seeding produced more ground cover, suppressed undesirable plants and increased species richness. But from an economic or conservation perspective, not seeding and allowing natural recruitment similarly provided adequate ground cover to protect the soil from erosion as well as having species richness, native plant cover and species composition similar to the undisturbed desert.

Our research offers support for more passive forms of restoration as a viable and possibly preferable alternative in arid grasslands. Pilot studies could verify seed banks and dispersal abilities of the existing vegetation to lower the risks of a passive approach and to verify that invasive or noxious weeds with potential to colonize a disturbed site are not present in adjacent areas. If a site requires seeding, local ecotypes and site-specific species should be used if at all possible.

The contrast between a recently disturbed site and an undisturbed site usually shows the undisturbed site as better in all respects. In our comparison, even in the unseeded areas, the reclamation areas showed improvements to vegetation metrics that are used by land managers to gauge the success of a reclamation project or management action (i.e. forage production and ground cover increased in all reclamation treatments compared to surrounding undisturbed desert). Our study design cannot resolve whether the mechanism behind these improvements was removing woody species and cacti vs soil disturbance increasing infiltration, reducing compaction and increasing nutrient availability. We also partially attribute the robust vegetation establishment at our reclamation project to adequate precipitation, which is essential for successful reclamation projects in arid and semi-arid regions.

Dormant-season cattle grazing did not significantly alter the community assemblage based on functional groups. However, the combination of grazing and trampling reduced production, reduced native plant cover and caused soil erosion. The general prescription of keeping cattle and vehicles off reclaimed sites for at least two full growing seasons (e.g. Stevens, 2004) seems warranted if at all possible.

ACKNOWLEDGEMENTS

This work was supported by Sierrita Gas Pipeline LLC. We thank field assistants J. Hanson, R. Turner, T. Chilcote, E. Pecilunas, Z. Winston, J. Sutter and G. Martin. We also acknowledge the contributions/assistance from the Altar Valley Conservation Alliance, the Arizona State Land Department, and the Buenos Aires National Wildlife Refuge.

AUTHOR CONTRIBUTIONS

JF and HF designed the research, HF collected the data, JF and HF analysed the data and wrote the manuscript.

ORCID

Hannah L. Farrell http://orcid.org/0000-0001-8740-8436

REFERENCES

Abella, S. R., & Newton, A. C. (2009). A systematic review of species performance and treatment effectiveness for revegetation in the Mojave Desert, USA. In A. Fernandez-Bernal & M. A. De La Rosa (Eds.), *Arid environments and wind erosion* (pp. 45–74). Hauppauge, NY: Nova Science.

Allington, G. R. H., & Valone, T. J. (2011). Long-term livestock exclusion in an arid grassland alters vegetation and soil. *Rangeland Ecology & Management*, *64*, 424–428. https://doi.org/10.2111/REM-D-10-00098.1

Baasch, A., Kirmer, A., & Tischew, S. (2012). Nine years of vegetation development in a postmining site: Effects of spontaneous and assisted site recovery. *Journal of Applied Ecology*, *49*, 251–260. https://doi.org/10.1111/j.1365-2664.2011.02086.x

Bainbridge, D. A. (2007). *New hope for arid lands: A guide for desert and dryland restoration*. Washington, DC: Island Press.

Bainbridge, D. A., & Virginia, R. A. (1990). Restoration in the Sonoran Desert of California. *Restoration and Management Notes*, *8*, 3–14.

Belyea, L. R., & Lancaster, J. (1999). Assembly rules within a contingent ecology. *Oikos*, *86*, 402–416. https://doi.org/10.2307/3546646

Berry, K. H., Weigand, J. F., Gowan, T. A., & Mack, J. S. (2016). Bidirectional recovery patterns of Mojave Desert vegetation in an aqueduct pipeline corridor after 36 years: I. Perennial shrubs and grasses. *Journal of Arid Environments*, *124*, 413–425. https://doi.org/10.1016/j.jaridenv.2015.03.004

Bertiller, M. B., & Aloia, D. A. (1997). Seed bank strategies in Patagonian semi-arid grasslands in relation to their management and conservation. *Biodiversity and Conservation*, *6*, 639–650. https://doi.org/10.1023/A:1018397615476

Bossuyt, B., & Honnay, O. (2008). Can the seed bank be used for ecological restoration? An overview of seed bank characteristics in European communities. *Journal of Vegetation Science*, *19*, 875–884. https://doi.org/10.3170/2008-8-18462

Browning, D. M., & Archer, S. R. (2011). Protection from livestock fails to deter shrub proliferation in a desert landscape with a history of heavy grazing. *Ecological Applications*, *21*, 1629–1642. https://doi.org/10.1890/10-0542.1

Coulloudon, B., Eshelman, K., Gianola, J., Habich, N., Hughes, L., Johnson, C., … Willoughby, J. (1999). *Sampling vegetation attributes*. Technical Reference 1734–4. Denver, CO: Bureau of Land Management, National Business Center.

Cox, J. R., Morton, H. L., Johnson, T. N., Jordan, G. L., Martin, S. C., & Fierro, L. C. (1982). *Vegetation restoration in the Chihuahuan and Sonoran Deserts of North America*. Agriculture Reviews and Manuals #28. Tucson, AZ: U.S.D.A. Agricultural Research Service.

Davies, K. W., Boyd, C. S., Bates, J. D., & Hulet, A. (2015). Dormant season grazing may decrease wildfire probability by increasing fuel moisture and reducing fuel amount and continuity. *International Journal of Wildland Fire*, *24*, 849–856. https://doi.org/10.1071/WF14209

Deák, B., Valkó, O., Török, P., Kelemen, A., Miglécz, T., Szabó, S., … Tóthmérész, B. (2015). Micro-topographic heterogeneity increases plant diversity in old stages of restored grasslands. *Basic and Applied Ecology*, *16*, 291–299. https://doi.org/10.1016/j.baae.2015.02.008

Drake, J. A. (1991). The mechanisms of community assembly and succession. *Journal of Theoretical Biology*, *147*, 213–233.

Dunne, T., Western, D., & Dietrich, W. E. (2011). Effects of cattle trampling on vegetation, infiltration, and erosion in a tropical rangeland. *Journal of Arid Environments*, *75*, 58–69. https://doi.org/10.1016/j.jaridenv.2010.09.001

Dyer, A. R., Hardison, J. L., & Rice, K. J. (2012). Phenology constrains opportunistic growth response in *Bromus tectorum* L. *Plant Ecology*, *213*, 103–112. https://doi.org/10.1007/s11258-011-0010-4

Fensham, R. J., Butler, D. W., Fairfax, R. J., Quintin, A. R., & Dwyer, J. M. (2016). Passive restoration of subtropical grassland after abandonment of cultivation. *Journal of Applied Ecology*, *53*, 274–283. https://doi.org/10.1111/1365-2664.12551

Fynn, R. W. S., & O'Connor, T. G. (2000). Effect of stocking rate and rainfall on rangeland dynamics and cattle performance in a semi-arid savanna, South Africa. *Journal of Applied Ecology*, *37*, 491–507. https://doi.org/10.1046/j.1365-2664.2000.00513.x

Gauch, H. G., Whittaker, R. H., & Singer, S. B. (1981). A comparative study of nonmetric ordinations. *Journal of Ecology*, *69*, 135–152. https://doi.org/10.2307/2259821

Gornish, E. S., & Ambrozio dos Santos, A. P. (2016). Invasive species cover, soil type, and grazing interact to predict long-term grassland restoration success. *Restoration Ecology*, *24*, 222–229. https://doi.org/10.1111/rec.12308

Hansen, M. J., & Clevenger, A. P. (2005). The influence of disturbance and habitat on the presence of non-native plant species along transport corridors. *Biological Conservation*, *125*, 249–259. https://doi.org/10.1016/j.biocon.2005.03.024

Hessing, M. B., & Johnson, C. D. (1982). Disturbance and revegetation of Sonoran Desert vegetation in an Arizona powerline corridor. *Journal of Range Management*, *35*, 254–258. https://doi.org/10.2307/3898405

Hoffman, M. T., Barr, G. D., & Cowling, R. M. (1990). Vegetation dynamics in the semi-arid eastern Karoo, South Africa—The effect of seasonal rainfall and competition on grass and shrub basal cover. *South African Journal of Science*, *86*, 462–463.

Kristensen, H. L., Kasia, D., & McCarty, G. W. (2003). Short-term effects of tillage on mineralization of nitrogen and carbon in soil. *Soil Biology and Biochemistry*, *35*, 979–986. https://doi.org/10.1016/S0038-0717(03)00159-7

Lambin, E. F., & Geist, H. J. (2006). *Land-use and land-cover change: Local processes and global impacts*. Berlin, Germany: Springer. https://doi.org/10.1007/3-540-32202-7

Lathrop, E. W., & Archbold, E. F. (1980). Plant response to utility right of way construction in the Mojave Desert. *Environmental Management*, *4*, 215–226. https://doi.org/10.1007/BF01866455

Lezama, F., Baeza, S., Altesor, A., Enrique, A. C., Chaneton, J., & Paruelo, J. M. (2014). Variation of grazing-induced vegetation changes across a large-scale productivity gradient. *Journal of Vegetation Science*, *25*, 8–21. https://doi.org/10.1111/jvs.12053

Martin, L. M., & Wilsey, B. J. (2006). Assessing grassland restoration success: Relative roles of seed additions and native ungulate activities. *Journal of Applied Ecology*, *43*, 1098–1109. https://doi.org/10.1111/j.1365-2664.2006.01211.x

Martinez-Ruiz, C., Fernandez-Santos, B., Putwain, P. D., & Fernandez-Gomez, M. J. (2007). Natural and man-induced revegetation on mining wastes: Changes in the floristic composition during early succession. *Ecological Engineering*, *30*, 286–294. https://doi.org/10.1016/j.ecoleng.2007.01.014

Milchunas, D. G., & Vandever, M. W. (2014). Grazing effects on plant community succession of early- and mid-seral seeded grassland compared to shortgrass steppe. *Journal of Vegetation Science*, *25*, 22–35. https://doi.org/10.1111/jvs.12049

Miyamoto, D. L., Olson, R. A., & Schuman, G. E. (2004). Long-term effects of mechanical renovation of a mixed-grass prairie: I. plant production. *Arid Land Research and Management*, *18*, 93–101. https://doi.org/10.1080/15324980490279601

NOAA (2015). *National Oceanic and Atmospheric Administration: Monthly summary observations*. Retrieved from https://gis.ncdc.noaa.gov/maps/ncei/summaries/monthly

Salihi, D. O., & Norton, B. E. (1987). Survival of perennial grass seedlings under intense grazing in semiarid rangelands. *Journal of Applied Ecology*, *24*, 145–151. https://doi.org/10.2307/2403793

Scott, A. J., & Morgan, J. W. (2011). Resilience, persistence and relationship to standing vegetation in soil seed banks of semi-arid Australian old fields. *Applied Vegetation Science*, *15*, 48–61.

Stevens, R. (2004). Management of restored and revegetated sites. In S. B. Monsen, R. Stevens, & N. L. Shaw (Eds.), *Restoring western ranges and wildlands* (pp. 193–198). Gen. Tech. Rep. RMRS-GTR-136. Fort Collins, CO, US: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Stevens, J. M., & Fehmi, J. S. (2011). Early establishment of a native grass reduces the competitive effect of a non-native grass. *Restoration Ecology*, *19*, 399–406. https://doi.org/10.1111/j.1526-100X.2009.00565.x

Stylinski, C. D., & Allen, E. B. (1999). Lack of native species recovery following severe exotic disturbance in southern Californian shrublands. *Journal of Applied Ecology*, *36*, 544–554. https://doi.org/10.1046/j.1365-2664.1999.00423.x

USDA, NRCS (2016). *NRCS: The PLANTS Database*. Retrieved from http://plants.usda.gov

Wainwright, C. E., Wolkovich, E. M., & Cleland, E. E. (2012). Seasonal priority effects: Implications for invasion and restoration in a semi-arid system. *Journal of Applied Ecology*, *49*, 234–241. https://doi.org/10.1111/j.1365-2664.2011.02088.x

Walker, L. R., & Powell, E. A. (1999). Effects of seeding on road revegetation in the Mojave Desert, Southern Nevada. *Ecological Restoration*, *17*, 150–155. https://doi.org/10.3368/er.17.3.150

Waller, P. A., Anderson, P. M. L., Holmes, P. M., & Allsopp, N. (2016). Seedling recruitment responses to interventions in seed-based ecological restoration of Peninsula Shale Renosterveld, Cape Town. *South African Journal of Botany*, *103*, 193–209. https://doi.org/10.1016/j.sajb.2015.09.009

Whisenant, S. G. (1999). *Repairing damaged wildlands: A process-oriented, landscape-scale approach*. Cambridge, UK: Cambridge University Press. https://doi.org/10.1017/CBO9780511612565

Whisenant, S. G., & Wagstaff, F. J. (1991). Successional trajectories of a grazed salt desert shrubland. *Vegetatio*, *94*, 133–140. https://doi.org/10.1007/BF00032627

WRCC (2015). *Western Regional Climate Center: Cooperative climatological data summaries*. Retrieved from http://www.wrcc.dri.edu/climatedata/climsum

Yavitt, J., & Smith, E. (1983). Spatial patterns of mesquite and associated herbaceous species in an Arizona desert grassland. *The American Midland Naturalist*, *109*, 89–93. https://doi.org/10.2307/2425519

SUPPORTING INFORMATION

Additional Supporting Information may be found online in the supporting information tab for this article.

APPENDIX S1 Representative photographs of the undisturbed desert area adjacent to the pipeline

APPENDIX S2 Photographs of the subplot structures

APPENDIX S3 Table of "Native Species" and "Undesirable Species"

How to cite this article: Farrell HL, Fehmi JS. Seeding alters plant community trajectory: Impacts of seeding, grazing and trampling on semi-arid re-vegetation. *Appl Veg Sci*. 2018;21:240–249. https://doi.org/10.1111/avsc.12340