WALKER BASIN ALTERNATIVE AGRICULTURE AND VEGETATION MANAGEMENT PHASE III

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ABSTRACT

Restoring native shrub communities in former agricultural fields is challenging but the benefits include providing important wildlife habitat, protecting lands from erosion, and reducing populations of invasive weeds. Here, we present preliminary information on the effectiveness of seeding native shrubs in two former agricultural fields on the Rafter 7 Ranch and testing the effects of a variety of treatments. These include seeding shrubs with and without native perennial grasses, varying the seed source of shrubs and grasses, varying water amounts and timing, and using weed control efforts such as herbicide and mowing. We are also investigating how soil properties affect plant performance in these former agricultural fields. Additionally, we are gathering information on surrounding native reference communities with the goal of describing naturally occurring species composition, densities, and plant-soil relationships to provide information on realistic restoration targets in these systems. To date, the results in the restoration plots suggest strong effects of species and seed source on plant establishment, and contrasting effects of irrigation at the two field sites. In the natural reference communities, species composition and density vary across the Rafter 7 Ranch; our soil analyses are designed to provide information on species/soil associations that can be used to design seed mixes for different types of sites. After irrigation has ceased, monitoring over the next field season (2017) will be crucial for determining which restoration strategies are most effective.
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OBJECTIVE

The overall goals of this project were: (1) to experimentally determine the most effective ways to transition high-water-use agriculture systems to lower water usages by active restoration to native plant communities; (2) to consider the use of alternative, water-efficient crops; and (3) to consider how plant-soil interactions affect plant performance in arid systems. To achieve these goals, we conducted field observations and experiments, soil measurements, greenhouse work, and small plot studies. The results of Phase I and Phase II of this project are reported in detail at http://greatbasinresearch.com/walker/symposium/downloads/2016-DTLS-Program-web.pdf. Here, we focus on the results of Phase III, which addresses the following specific questions. This report serves as the final report for Phase III for PI J. Davison and as the interim report for PI E. Leger and collaborators (W. Miller, B. Sullivan, O. Baughman, S. Uselman).

QUESTIONS

1. What effects do the presence of grass seeding, timing of shrub seeding, weed control methods, irrigation regime, and seed origin have on shrub, grass, and weed establishment in former agricultural fields?
   a. Does grass seeding affect shrub establishment overall or the establishment of particular shrub species, and do these effects differ by irrigation regime, the timing of shrub seeding, or by grass or shrub seed origin?
   b. Do particular seeding strategies (presence of grass, timing of shrub seeding, weed control method), irrigation regimes, or origins of seeded species (or combinations thereof) have significant effects on weed densities?

2. How do soil characteristics affect restoration in post-agricultural fields in the salt desert?
   a. Do small spatial scale differences in soil characteristics influence shrub and grass densities in the context of our ongoing restoration study?
   b. How do soil characteristics compare in native shrublands versus our restoration study?
   c. Do soil characteristics help explain shrub composition and cover in native sites?

METHODS: AGRICULTURAL FIELD RESTORATION

SITE DESCRIPTIONS

The Rafter 7 Ranch in Lyon County, Nevada—which is located along the East Walker River, approximately 45 km upstream of Yerington, Nevada—was chosen as a representative study site for Phase III work. Irrigation infrastructure was present, and the site is a typical example of a previously cultivated area that is a candidate for transitioning to native communities. Two field sites (one north and one south, referred to as N and S) within the ranch
were identified as representative of two types of fields needing restoration. Both locations, separated by 1.4 km, were historically leveled, tilled, and cultivated for the production of alfalfa, although the sites differ in their recent history of cultivation. The N site had been irrigated and produced alfalfa for a decade or more until the summer of 2013, whereas the S site had not been in production or regularly irrigated for at least a decade (Google Earth, USGS Earth Explorer). Prior to the start of the experiment, the N site contained residual alfalfa (*Medicago sativa*) plants, whereas the S site did not. Weeds occupying one or both of the sites prior to the experiment included annuals such as flixweed (*Descurainia sophia*), tumble mustard (*Sisymbrium altissimum*), Russian thistle (*Salsola tragus*), redstem filaree (*Erodium cicutarium*), and kochia (*Bassia scoparia*), as well as perennials such as bindweed (*Convolvulus arvensis*) and perennial pepperweed (*Lepidium latifolium*).

**STUDY DESIGN**

Our previous research on restoring former agricultural fields to native vegetation in this area of the Great Basin focused on seeding grasses and transplanting shrubs while using short-term irrigation and weed control methods to increase establishment. That research found that although short-term irrigation and grass seeding decreased soil erosion and weed densities, shrub establishment was poor in areas of high grass establishment, which was likely because of competition with seeded grasses (Porensky *et al.*, 2014; Appendix A). Even though they may compete with shrubs, grasses add functional and ecological diversity as well as decrease soil erosion. Therefore, there are benefits for including grasses in the restoration process, if their competitive effects can be reduced. The present experiment incorporated five seeding strategies that were designed to determine if grasses could be incorporated into shrub-focused restoration either by establishing stands at lower densities or staggering them over time in a way that did not hinder, and perhaps even assists, shrub establishment (Table 1). The five seeding strategies were:

I) seed grasses first year, then shrubs second year

II) seed grasses and shrubs first year, then nothing second year

III) seed shrubs first year, then nothing second year

IV) herbicide fallow first year, then seed shrubs second year

V) no seeding either year (control)

Because of limited seed availability, the seed mix that we used in our previous trial was comprised of nonlocal cultivars or named varieties of perennial grasses. For this trial, we varied the seed source to determine if the origin of our seed sources would affect short- and long-term restoration outcomes. We tested the seed source by including the seed origin (local versus commercial sources for grasses, and two different but regionally local shrub sources for shrubs) as a fully crossed factor within each of these five main strategies, creating 14 unique subplots.
Table 1. The study included six experimental factors. For analysis, the design was organized into five main seeding strategies, I to V. Forward slash (/) indicates that the listed levels were analyzed separately within the given strategy. Plus signs (+) indicate that treatments were applied together within the given strategy, and dashes (-) indicate when factors were not included in a seeding strategy.

<table>
<thead>
<tr>
<th>Experimental factors</th>
<th>Seeding Strategy</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>I</td>
</tr>
<tr>
<td>Grass in year 1</td>
<td>yes</td>
</tr>
<tr>
<td>Shrub seeding year</td>
<td>2</td>
</tr>
<tr>
<td>Grass seed origin</td>
<td>local / commercial</td>
</tr>
<tr>
<td>Shrub seed origin</td>
<td>A / B</td>
</tr>
<tr>
<td>Irrigation regime</td>
<td>spring / fall+spring</td>
</tr>
<tr>
<td>Weed control</td>
<td>herbicide+mowing</td>
</tr>
</tbody>
</table>

Additionally, we manipulated the timing and amount of irrigation applied to these seeding treatments and seed sources. During our previous experiment, establishment in nonirrigated fields was near zero, so we did not include a nonirrigated treatment in this set of experiments. Instead, we opted to reduce the amount of seasonal water received on our fields (irrigation + background precipitation) from the range of 300-430 mm that we used in our previous experiment—which resulted in very high densities of perennial grasses—to a range of 150-230 mm in the present experiment. We selected two treatments: 150 mm of water in spring (spring-only), or 75 mm of water in the fall + 150 mm in the spring (fall+spring). As in our previous experiment, irrigation was applied for two years to facilitate establishment and no irrigation is planned for year three. Two groups (subblocks) of the 14 subplots were arranged side by side into a block of 28 subplots, with each group receiving one of the two irrigation regimes (spring-only, fall+spring). This design was replicated in three blocks at each of the two sites, creating a total of 168 subplots.

Within each block, a completely random arrangement of subplots was not possible because of the complexity of irrigation and mechanical seeding treatments. Therefore, each block of 28 plots was split into two side-by-side subblocks, each receiving a different irrigation regime, as described above. Within each subblock, seven plots were randomly arranged. Plots 1 and 2 were both assigned to strategy I but differed by grass origin (1=local, 2=commercial). Plots 3 and 4 were assigned to strategy II, also differing by grass origin (3=local, 4=commercial). Plots 5-7 were assigned to strategies III-V, respectively, and divided into two subplots that were each randomly assigned with a different origin of shrub seed, except for plot 7, which was unseeded.

**SPECIES SELECTION**

The species selected for this experiment are native species that occur within the region and at nearby sites, and for which seeds are generally available (Table 2). Shrub species were Wyoming big sagebrush (*Artemisia tridentata* var. *wyomingensis*), fourwing saltbush (*Atriplex*...
canescens), rubber rabbitbrush (*Ericameria nauseosa*), black greasewood (*Sarcobatus vermiculatus*), and quailbush (*Atriplex torreyi*). Grass species were Indian ricegrass (*Achnatherum hymenoides*), alkali sacaton (*Sporobolus airoides*), bottlebrush squirreltail (*Elymus elymoides*), and basin wildrye (*Leymus cinereus*). With the exception of alkali sacaton, two sources of seed were obtained for each grass species: one source being from wild collections originating from as near to the planting sites as possible (local source), and the other source representing large-scale collections and cultivars (commercial source) originating from various and often nonlocal, locations. Seed sources for shrubs were all wild collected because these species are not typically farmed and there are few to no native shrub cultivars developed. Shrubs were sourced from multiple locations and labeled as either less-local (origin A) or more-local (origin B) in our experiment (Table 2). All seed was purchased from Comstock Seed in Gardnerville, Nevada. Seed rates were established based on USDA-ARS publication AG 510 (Intermountain Planting Guide) and previous experiences of University of Nevada Cooperative Extension employees.

**PRE-SEEDING SITE PREPARATION**

Site treatments began in March 2014 (Figure 1). All plots at both sites were treated with glyphosate at 2.24 kg ai/ha (2 lbs ai/ac) by a commercial applicator on March 6, 2014, and again at the same rate on April 9, 2014. These treatments were marginally effective on annual weeds but ineffective on perennial weeds. Glyphosate was again applied to the N site at 2.24 kg ai/ha (2 lbs ai/ac) on October 3, 2014, to target remnant alfalfa, and this application was only effective on the small plants. The lack of soil moisture, which limited active plant growth, was thought to be the primary factor limiting the effectiveness of the herbicide applications. All other weedy litter and debris was mowed from all plots at both sites on September 29, 2014. Portions of the S site containing larger densities of senesced weeds were spot-treated with a handheld propane

<table>
<thead>
<tr>
<th>Shrub Species</th>
<th>Origin A (less local)</th>
<th>Origin B (more local)</th>
<th>Rate (lbsPLS/ac*)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Sagebrush</td>
<td>California</td>
<td>Humboldt/Pershing/Washoe Co., NV</td>
<td>0.5</td>
</tr>
<tr>
<td>Fourwing saltbush</td>
<td>Arizona</td>
<td>Lyon/Churchill Co., NV</td>
<td>2.0</td>
</tr>
<tr>
<td>Rubber rabbitbrush</td>
<td>Nevada</td>
<td>W. NV</td>
<td>0.05</td>
</tr>
<tr>
<td>Greasewood</td>
<td>Nevada</td>
<td>Churchill/Pershing Co., NV</td>
<td>2.0</td>
</tr>
<tr>
<td>Quailbush</td>
<td>Arizona</td>
<td>N. Central NV</td>
<td>2.0</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Grass Species</th>
<th>Commercial origin</th>
<th>Local origin</th>
<th>Rate (lbsPLS/ac*)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Indian ricegrass</td>
<td>‘Nezpar’, Idaho Co., ID</td>
<td>Mono Co., CA</td>
<td>2.0</td>
</tr>
<tr>
<td>Alkali sacaton</td>
<td>Utah (same as local)</td>
<td>Utah (same as commercial)</td>
<td>0.25</td>
</tr>
<tr>
<td>Squirreltail</td>
<td>‘Klamath’, Klamath Co., OR</td>
<td>Mono Co., CA</td>
<td>2.0</td>
</tr>
<tr>
<td>Basin wildrye</td>
<td>‘Trailhead’, Musselshell Co., MT</td>
<td>Lassen Co., CA</td>
<td>2.5</td>
</tr>
</tbody>
</table>

*Seed rates for these species are exclusively conveyed in lbsPLS/ac in this region, and therefore we present these units. For metric, 1.0 lbs/ac is equivalent to 1.12 kgPLS/ha.*
Figure 1. Timeline of precipitation (black bars, left side), added irrigation (gray bars, left side), and site activities (right side). Unless noted, all activities occurred at both N and S sites. Precipitation data was gathered from PRISM (PRISM Climate Group), which models precipitation at 4 km resolution from nearby weather stations, and adjusted for accuracy according to reported precipitation from ranch managers.
torch on October 2, 2014, to reduce biomass and scorch weed seeds on the soil surface. Seedbed preparation on all plots at both sites consisted of rototilling to a depth of 10 cm followed immediately by a ring roller cultipacker to firm the soil (Figure 2).

SEEDING AND IRRIGATION ACTIVITIES

First Year Seeding and Irrigation (Fall 2014-Spring 2015)

Irrigation throughout the project was achieved using 76 mm (3 in) hand lines off of 101 mm (4 in) mainline pipes mounted with #7 Senninger wobbler sprinkler heads on 914 mm (36 in) risers. The distance between the heads was 9.1 m to maximize the uniformity of spray patterns.

Plots in treatments receiving first-year seeding were seeded over a five-day window, between October 7 and 10, 2014 (Figure 1 and 2). The grass mix was seeded in the N-S direction, parallel to the long plot axis, with no spaces between consecutive drill passes. Shrubs were seeded in the E-W direction, perpendicular to the grasses, in 1.6 m wide single-species strips that were each separated by one 1.6 m space. Each plot was planted in the same order of shrub species—(1) fourwing saltbush, (2) greasewood, (3) quailbush, (4) rubber rabbitbrush, and (5) Wyoming big sagebrush—with the fourwing saltbush always beginning on the north side of the seeded plot. The seeding of shrubs perpendicularly to grasses was designed to allow for the spatial separation of seedlings within plots where both grasses and shrubs were seeded, which previous research suggested would improve shrub establishment. The plots were seeded at a depth of 1.3 cm using a Truax seed drill with double-disc openers at 20.3 cm row spacing followed by packing wheels. Cattle were excluded from plots to the greatest degree possible with electric fences, although some trampling in the first year did occur.

In the fall of 2014, irrigation applications occurred on October 21st and October 29th (Figure 1). Because there was trouble with the irrigation system, a planned final irrigation of 25 mm was not added, but natural precipitation was 43 mm above normal (~210 percent) for October through December. In spring 2015, approximately 100 mm of total irrigation was added on April 6th and 8th; April 16th and 17th; April 28th and 30th; May 1st, 6th, and 7th; and June 1st and 2nd.

Second Year Seeding and Irrigation (Fall 2015-Spring 2016)

Previously unseeded plots that were slated for shrub seeding in fall 2015 (strategy IV) were rototilled to 10 cm during the week of August 18th to remove any existing vegetative competition and prepare the soil for planting (Figure 1). Previously seeded plots slated for second-year shrub seeding (strategy I) were not disturbed prior to planting. Two, 25 mm pre-irrigation treatments were applied during the weeks of September 15th and 21st to add soil moisture and firm the tilled plots. The plots were drug with a lightweight rake to remove any vegetation that could impede the shrub seeding operation. The plots were seeded between September 29th and October 1st using the same methods as in the first year (Figure 2).
Figure 2. Seedbed preparation at all sites before seeding included rototilling (top, N site) and cultipacking (middle, S site). Shrubs and grasses were seeded (bottom) perpendicular to each other in both years (visible in bottom left in 2014).
In the fall of 2015, 50 mm of total irrigation was added on October 14th and 15th, and October 26th and 27th. In the spring of 2016, 13 mm of irrigation was added on April 15th. Because precipitation was ~58 mm higher than average over the winter and through the spring of 2016, this was the final irrigation of the project, as weeds were already at very high densities and large sizes by April 2016. Little precipitation occurred through the summer.

**WEED CONTROL**

To provide adequate weed management on the plots throughout the experiment, a combination of mechanical and chemical treatments was necessary (Figure 3). Plots seeded with native grasses or left unseeded in the first year and slated for second-year shrub seeding (strategies I and IV) were sprayed with broadleaf herbicides at strategic times (listed below) during the first growing season. Because plots seeded with shrubs could not be sprayed with broadleaf herbicide, repeated mechanical mowing was necessary to control weed growth in these plots, especially in irrigated treatments. Mowing was set to a height of 15.2 cm because it: (1) provided more understory light for seeded seedlings by reducing weed height and cover; (2) avoided shrub seedlings, which were substantially shorter; and (3) affected only the leaf tips of the tallest grass seedlings. No herbicide application took place on unseeded plots (strategy V) after seeding began in October 2014.

Plots assigned to strategy I and IV in the S field were sprayed with 2.3 L AI/ha (1 qt/ac) Weedmaster (2,4-d + dicamba) on May 4, 2015. No herbicide was applied to the N plots on this date because the seeded grasses were too small to withstand the spray. All plots were mowed on June 4 and 5, 2015, and July 15, 2015, after which little or no weed growth occurred because of a lack of rain. Plots assigned to strategy I and IV in both the N and S fields were sprayed on June 8, 2015, with 2.3 L AI/ha (1 qt/ac) Weedmaster because of the larger weed size. The areas immediately surrounding our experimental sites were also treated with herbicide on October 3, 2014; May 4, 2015; and June 8, 2015, to reduce the dispersal of weed seeds into the study areas.

After the second planting in fall 2015, weed control was limited to mowing rather than herbicide because all plots had been seeded with shrubs, which are sensitive to broadleaf herbicides. Mowing on all plots took place on July 5 and 6, 2016.

**MONITORING**

Densities and maximum heights of seeded species and weeds were assessed on Dec 16, 2014 (seeded species only); April 3, 2015; May 1, 2015; and October 13 and 14, 2015. At each of these sampling dates, three samples were taken randomly in every subplot using square meter sampling frames. In plots seeded with five monospecific strips of shrubs, this sampling rate was not large enough to sample every strip. However, no evidence of shrub emergence was detected at any sampling period in the first year of monitoring, so this level of sampling effort was deemed sufficient to capture emergence of seeded grasses and weeds.
Figure 3. Weedy vegetation through time at the S site. The presence of weedy vegetation became intense in the summer of the first growing season (2015, top two images). The immediate effect of mowing can be seen in the top left image and a one to two month delayed effect of herbicide application can be seen in the top right image. During the second growing season at the S site (2016, bottom two images), weeds appeared to be at benign levels early in the season (March, bottom left), but were the dominant vegetation by mid to late spring (April, bottom right).

Second-year monitoring consisted of a single sampling effort from April 29 to June 4, 2016. As shrubs emerged in this year, monitoring was more extensive than in the first year. Densities and maximum heights of seeded and nonseeded perennial native grasses, seeded and nonseeded native shrubs, exotic weeds, and other nonseeded native species were assessed in a variety of quadrat sizes. For each sample, seeded shrubs and grasses were assessed in 1 x 1 m quadrats, whereas weeds and nonseeded plants were assessed in a randomly selected 25 x 25 cm corner of the full quadrat. A reduced sampling size of 10 x 10 cm was used in cases of high weed densities (hundreds) to reduce errors caused by counting fatigue. At the S site, three quadrats were placed at random locations within each of the five monospecific shrub strips. Values were
averaged to produce one measure per strip for analysis. At the N site, one to three quadrats were randomly located in each of the five monospecific shrub strips. A reduced effort was undertaken here because of the extensive number of zeros in the sampled quadrats.

**DATA ANALYSIS**

Analyses were performed separately for each site (N and S) because the sites had different histories and clearly different establishment patterns. First- and second-year densities of seeded grasses and shrubs as well as weeds were analyzed using mixed-effect ANOVA models in JMP (SAS Institute, 2016). Four random effects were included in all models to control for the spatial patterns of potential variation associated with our design. These effects were BLOCK, SUBBLOCk (nested in BLOCK), PLOT (nested in SUBBLOCk and BLOCK), and SUBPLOT (nested in PLOT, SUBBLOCk, and BLOCK). The main model effects were IRRIGATION and STRATEGY. The appropriate comparisons between different seeding strategies were made using Tukey’s HSD (honest significant difference) or linear contrasts to examine the effects of the factors summarized by seeding strategy (see Table 1). For example, to determine the effect of the timing of shrub seeding, comparisons would be made between strategy I and II (with grasses), as well as between III and IV (without grasses). Several other model variations were used to address specific questions using a subset of data appropriate for each question. These models used seed origins—SHRUBORIGIN and/or GRASSORIGIN—as main effects and included only data from strategies in which shrubs and/or grasses were seeded (see Table 1). All models included fully factorial combinations of all main factors to identify significant interactions. The sampling date was initially included as a main model effect, but because it showed consistently significant interactions with other factors, we removed it from the model and analyzed the results separately for each sampling date.

**METHODS: PLANT-SOIL RELATIONSHIPS**

**SOIL SAMPLING IN THE RESTORATION SITES**

In the restoration study, we measured soil characteristics in “hot spots” and “cold spots” of plant density (i.e., high and low plant density). Plots with hot and cold spots of plant density were identified from the second year monitoring data, conducted in spring 2016. We identified two hot spots and two cold spots for the following species or life forms at the S site: (1) Wyoming big sagebrush; (2) rubber rabbitbrush; (3) quailbush; (4) fourwing saltbush; (5) perennial grasses, local seed sources; (6) perennial grasses, commercial seed sources; and (7) weeds. We did not sample black greasewood shrubs because emergence for that species was negligible. At the N site, we were unable to do this type of analysis because of the substantially lower shrub densities. Fortunately, this allowed us to sample more intensively at the S site. However, for comparative purposes, we measured soil characteristics in three locations in the N site, one in the center of each study block.
Specifically, hot and cold spots were chosen by ranking plots by density and selecting the two highest and two lowest density plots for each species or lifeform. In cases where there were more than two plots with the same density (e.g., when density was zero), two plots were selected randomly from among the identical values. All plot selections were from the treatments with the “more local” shrub seed source (i.e., origin B), where shrub emergence was highest (see results). For Wyoming big sagebrush, quailbush, and fourwing saltbush, plots were selected from treatments where shrubs were seeded in year 2 and where emergence was highest for these species. For rubber rabbitbrush, plots were selected from treatments where shrubs were seeded in year 1 and where emergence was highest for this species. For perennial grasses, plots were selected only from treatments where grasses were seeded. For weeds, plots were selected from any of the plots. If plots were selected more than once (e.g., for a sagebrush and fourwing hot spot), then the plot was sampled twice. This was necessary because of the spatial separation of the seeding of shrub species within a plot.

At each sampling location, we measured the following soil characteristics: (1) texture (hydrometer method, Gee and Bauder [1986]); (2) bulk density; (3) infiltration rate (disc permeameter); (4) water retention (WP4C Dew Point Potentiometer, Decagon Devices, Inc.); (5) pH (1:2 soil:water suspension); (6) salinity (electroconductivity, EC 1:2 soil:water extract); (7) total C and N (dry combustion, ECS 4010 Elemental Combustion System, Costech Analytical Technologies, Inc.); (8) soil surface cover by classes (gravel (5 mm to < 75 mm), cobble or larger rocks (≥ 75 mm), plant litter, biological crust, and physical crust; (9) N availability (nitrification potential rate [Hart et al., 1994], δ15N isotope signature); and (10) P availability (resin-extractable P, Lajtha et al. [1999]). Biological crust and physical crust did not exist within the restoration study. Soil samples were taken from a depth of 0-10 cm, except bulk density (0-15 cm). Soil measurements are currently ongoing.

Native Reference Site Selection

Based on satellite imagery searches (Google Earth, USGS Earth Explorer) and prior experience working at the ranch, we initially identified 47 sites within the Rafter 7 Ranch along the East Walker River corridor as potential candidates for sampling native vegetation and soils. Potential sites were named alphabetically, and for simplicity we have used these site designations in this report. We targeted minimally disturbed, lower-elevation sites representative of areas that would be commonly converted to agriculture. Based on previous observations of upland plant species establishing within lower-elevation old agricultural fields, we included some gently sloping upland sites where the water table was well below 1 m deep. Although these may contain species that historically did not grow in lower-elevation fields, given the altered environmental conditions because of agricultural use of ground/surface water, it may be necessary to consider using native species that are more adapted to drier conditions as a target for restoration of post-agricultural fields. Therefore, we sampled some of these higher elevation sites as well.
To encompass a range of plant and soil characteristics, we cross-referenced satellite imagery with NRCS soil maps (Soil Survey Staff, NRCS). We also considered detailed land ownership maps and other relevant environmental factors, and therefore narrowed our list down to 30 sites having the best potential for study. After ground-truthing, 15 of those sites were rejected based on current conditions, such as recent disturbance or excessive weed cover, which left us with 15 sites.

**SOIL AND VEGETATION SAMPLING IN THE NATIVE SITES**

In the 15 native shrubland reference sites, we measured both plant and soil characteristics. We conducted detailed surveys of plant community composition, density, and cover. At each site, we measured vegetation characteristics along three, 50 m long transects using standard methods (Herrick et al., 2005). Transects were spaced 20 m apart. The gap intercept method was used to measure shrub canopy gaps, shrub basal gaps, and shrub canopy cover. We modified the method to estimate shrub canopy cover by species. The belt transect method was used to measure shrub density by species, using a 3 m belt width. The same transect line from the gap intercept method was used to center the belt transect. Along each of the three transects, we placed three quadrats (1 m x 1 m) to measure the density of perennial grasses and forbs; cover of perennial grasses, annual grasses, perennial forbs, annual forbs, and shrubs (i.e., shrub canopy cover); and soil surface cover by classes (gravel, cobble or larger rocks, plant litter, biological crust, and physical crust). Quadrats were placed along each transect using systematic random sampling. The first transect was placed randomly from 1-16 m along the transect, the next was placed 17 m farther from the first, and the last was placed 17 m farther from the second. For all methods, we distinguished live from dead shrubs. Additionally, we described each site as follows: (1) GPS coordinate (latitude, longitude), (2) elevation, (3) slope, (4) aspect, and (5) landscape position (flood plain, upland).

At each of the 15 native shrubland reference sites, we measured soil characteristics at two of the nine quadrats where we surveyed vegetation. We chose the two quadrats by calculating the median shrub cover for each site and selecting the two quadrats with values closest to the median value (i.e., the value above and below the median). Soil was sampled from the center of the quadrat. Although quadrat locations were marked and relocated by GPS, we remeasured the vegetation characteristics (see methods for quadrats above) because of the accuracy of the GPS (±1 m). Shrub density was added to the data collection, and shrub density and cover were estimated by species. At each sampling location, we measured the same soil characteristics as in the restoration study (see above). These measurements are currently ongoing.

**DATA ANALYSIS**

In the restoration study, we will develop linear mixed models to predict shrub, perennial grass, and weed densities using measured soil characteristics. Our goal is to partition variation in plant density among soil predictors, restoration treatment effects (i.e., seeding strategy, irrigation treatments, etc.), and any remaining variation because of unknown or unmeasured factors.
In the native sites, we will summarize vegetation and soil characteristics by site and also across all 15 sites as baseline data to describe native salt desert shrubland communities. We will develop models to predict shrub cover using soil predictors. We will also use nonmetric multidimensional scaling (NMDS) ordination to analyze multivariate relationships between soil variables and vegetation characteristics in the 15 native sites.

FINDINGS TO DATE

AGRICULTURAL FIELD RESTORATION

Final results from the first and second growing season will be presented in our final report in late 2017. What follows is a summarized discussion of our preliminary analyses and the main results to date. Data from the third growing season (2017) are needed before final analyses and recommendations can be prepared.

At the last sampling during May 2016, densities of actively growing seeded shrubs ranged from ~0.1 to ~2.5 seedlings per square meter at the S site, and from 0 to less than 1.0 seedlings per square meter at the N site (Figures 4 and 5). These densities are similar to those in natural communities, which ranged from 0.3 to 1.2 shrubs per square meter (see the “Plant-soil relationships” findings section below). Quailbush was the most common species at both sites, and therefore dominated the observed trends described below. Fourwing saltbush, which was much less common than quailbush, was the next most common species across both sites, whereas the remaining species, rabbitbrush, sagebrush, and greasewood, demonstrated lower establishment at one or both sites. Seeded grass densities in May 2016 were generally higher than those for shrubs and did not differ as dramatically between the sites, ranging from 0.5 to 7 actively growing seedlings per square meter at the N site, and 1 to 4 seedlings per square meter at the S site (Figure 6). By all accounts, both sites were considered weed dominated during the second, very wet growing season, with densities ranging from ~100 to ~1,000 actively growing individuals per square meter (Figure 7). High densities of weeds are not unexpected in abandoned agricultural fields, especially when irrigated, as were all of our plots. All irrigation activities in our experiment have ceased as of spring 2016, and forthcoming monitoring in the third growing season will shed more light on whether or not the seeded species will continue to establish and/or weed levels will begin to recede.

Effects of seeding strategy, shrub seed origin, the presence of grass, and irrigation regime on shrub establishment

No appreciable numbers of shrubs were detected in the entire first growing season at either site. Therefore, a multi-temperature germination treatment was performed on seeds of all the shrub species during the month of August to determine if there was a problem with seed viability. Observed germination was adequate (between 70-100 percent for most species, except for *Atriplex canescens*, which demonstrated ~15-25 percent germination with increasing
germination for longer periods of cold stratification). Therefore, we hypothesized that the extremely dry and warm winter negatively affected emergence and survival of the shrubs planted in the fall 2014 in spite of the irrigation applied. In the second growing season, shrub emergence was much higher. Although densities were highest in plots that were seeded with shrubs in the second season (strategies I and IV), significant densities were also observed in first-year seeded plots. This indicates that seeds planted in the first year were still viable and able to grow in the second season (Table 3, Figures 4 and 5).

To date, seeding strategy and seed origin were the most important factors for early shrub establishment overall (Table 3, Figure 4). The origin of the shrub seed appears to be an important factor for early shrub establishment, with the B (more local) source seed producing higher densities of shrubs than the A source at both sites. Longer-term monitoring will be important to determine which shrub species and seed sources have the highest survival rates after the irrigation phase is completed.

The seeding strategies that involved broadleaf herbicide in the first year followed by shrub seeding in the second year (I and IV) produced the highest densities of seeded shrubs at both sites (Table 3, Figure 4). Broadleaf herbicides significantly reduced weeds at the N site and marginally reduced weeds at the S site (described below), which suggests that reduced weed competition may have played a role in shrub seedling recruitment. However, the higher shrub densities in strategy I and IV may also be due in part to a more favorable growing season in the second year of study than in the first year. Furthermore, densities of seeded species are commonly highest in the first year after seeding, and generally decline because of attrition in subsequent years. Our experimental design does not allow us to easily separate the effects of herbicide from the year-to-year variation in seeding success for shrubs, but future monitoring should indicate whether the current effects of seeding strategy are lasting or ephemeral. Therefore, we will interpret these results in more depth at that time.

Table 3. ANOVA results (F_{DF_Dom}) for the number of seeded shrubs encountered in May 2016. Factors included irrigation regime (IRRIGATION; fall+spring, spring only), seeding strategy (STRATEGY; I-IV), shrub seed origin (SHRUBORIGIN; A, B), and interactions. Data were log transformed, and bolded effects are significant.

<table>
<thead>
<tr>
<th>Factor</th>
<th>DF</th>
<th>N Site</th>
<th>S Site</th>
</tr>
</thead>
<tbody>
<tr>
<td>IRRIGATION</td>
<td>1</td>
<td>17.52_{(2.4)}*</td>
<td>0.05_{(2.1)}</td>
</tr>
<tr>
<td>STRATEGY</td>
<td>3</td>
<td>5.06_{(15.4)}*</td>
<td>9.19_{(24)}***</td>
</tr>
<tr>
<td>SHRUBORIGIN</td>
<td>1</td>
<td>3.05_{(22.6)}</td>
<td>41.48_{(28)}***</td>
</tr>
<tr>
<td>IRRIGATIONxSTRATEGY</td>
<td>3</td>
<td>5.52_{(15.4)}**</td>
<td>0.23_{(24)}</td>
</tr>
<tr>
<td>IRRIGATIONxSHRUBORIGIN</td>
<td>1</td>
<td>5.76_{(22.6)}*</td>
<td>0.31_{(28)}</td>
</tr>
<tr>
<td>STRATEGYxSHRUBORIGIN</td>
<td>3</td>
<td>1.88_{(22.3)}</td>
<td>17.12_{(23)}***</td>
</tr>
<tr>
<td>IRRIGATIONxSTRATEGYxSHRUBORIGIN</td>
<td>3</td>
<td>1.72_{(22.3)}</td>
<td>0.01_{(28)}</td>
</tr>
</tbody>
</table>

*P < 0.05, **P < 0.01, ***P < 0.001
Figure 4. Densities of seeded shrubs (seedlings/m²) in May 2016 sites (top two panels) for A (hashed) and B (solid) seed sources, and percent of the seeded shrub totals belonging to each species (bottom two panels). Only data from the fall+spring irrigation regime at the North site are shown here because almost no shrubs were detected in plots under the spring-only regime. Error bars are standard errors.
Previous research from Phases I and II of this project showed that early shrub establishment is hindered by high perennial grass establishment. This experiment attempted to reduce such competition by spatially separating shrubs and grasses through perpendicular seeding and by reducing the amount of irrigation applied, resulting in reduced grass densities. The current experiment lends mixed patterns regarding the interplay between shrubs and grasses. The N site showed higher seeded shrub densities when grasses were not present, but this pattern was absent at the S site, which did not show any effects of seeded grasses on shrub establishment. Because densities of both grasses and shrubs were fairly low and seedlings relatively small, it may be too early to measure the effects of any grass/shrub interactions. Further monitoring and analyses will aid us in determining if these patterns are persistent and if our techniques aimed at reducing the effects of competition were successful.
Irrigation regime was not a significant predictor of seeded shrub density at the S site, but the fall+spring regime supported higher shrub densities at the N site, which supported almost no seeded shrubs under the spring-only regime (data not shown).

**Effects of seeding strategy, irrigation regime, and seed origin on grass establishment**

Densities of actively growing seeded grasses varied through time and exhibited complex interactions with most experimental factors. The effects of seeding strategy and seed origin demonstrated fairly consistent patterns. Commercial grass seeds had higher densities than local seeds, and strategy I (which involved first-year grass seeding and herbicide application, followed by second-year shrub seeding) was marginally more successful at establishing grasses than strategy II (which involved seeding grasses and shrubs together in the first year, with no herbicide application; Table 4, Figure 6). As discussed above, the suppressive effect of first-year herbicide on weeds was significant enough to suggest that it may have reduced the competition between weeds and seeded grasses. Additional monitoring in the third growing season will help identify the importance of these patterns and will aid more conclusive interpretation. For example, in our experiment in Phase II, commercial grasses grew very well until irrigation ceased, at which point densities were reduced to near zero. Long-term monitoring will be important to determine which grass seed sources are able to persist over time and which sources are best at facilitating shrub establishment.

Table 4. ANOVA results (F_{DF_Den}) for the number of seeded grasses encountered through time by site. Too few grasses were detected in April 2015 at the N site to run analyses. Factors included irrigation regime (IRRIGATION; fall+spring, spring only), seeding strategy (STRATEGY; I, II), grass seed origin (GRASSORIGIN; commercial, local), and interactions. Data were log transformed and bolded effects are significant.

<table>
<thead>
<tr>
<th>Factor</th>
<th>Number of actively growing seeded grasses/m²</th>
<th>N SITE DF</th>
<th>Dec-14</th>
<th>Apr-15</th>
<th>May-15</th>
<th>Oct-15</th>
<th>May-16</th>
</tr>
</thead>
<tbody>
<tr>
<td>Irrigation</td>
<td></td>
<td>1</td>
<td>16.3(2)</td>
<td>3.42(2)</td>
<td>3.33(2)</td>
<td>4.80(2)</td>
<td></td>
</tr>
<tr>
<td>Strategy</td>
<td></td>
<td>1</td>
<td>3.21(12)</td>
<td>2.91(12)</td>
<td>10.4(12)</td>
<td>15.6(11.3)</td>
<td></td>
</tr>
<tr>
<td>GrassOrigin</td>
<td></td>
<td>1</td>
<td>7.32(12)*</td>
<td>4.43(12)</td>
<td>3.49(12)</td>
<td>0.78(11.3)</td>
<td></td>
</tr>
<tr>
<td>Irrigation x Strategy</td>
<td></td>
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<td>1.52(12)</td>
<td>0.00(12)</td>
<td>1.20(12)</td>
<td>1.63(11.3)</td>
<td></td>
</tr>
<tr>
<td>Irrigation x GrassOrigin</td>
<td></td>
<td>1</td>
<td>4.61(12)*</td>
<td>8.92(12)*</td>
<td>2.56(12)</td>
<td>3.56(11.3)</td>
<td></td>
</tr>
<tr>
<td>Strategy x GrassOrigin</td>
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<td>0.41(12)</td>
<td>0.82(12)</td>
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<td>0.09(12)</td>
<td>0.05(12)</td>
<td>0.00(11.3)</td>
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<table>
<thead>
<tr>
<th>Factor</th>
<th>Number of actively growing seeded grasses/m²</th>
<th>S SITE DF</th>
<th>Dec-14</th>
<th>Apr-15</th>
<th>May-15</th>
<th>Oct-15</th>
<th>May-16</th>
</tr>
</thead>
<tbody>
<tr>
<td>Irrigation</td>
<td></td>
<td>1</td>
<td>58.9(2)*</td>
<td>132.7(2)**</td>
<td>145(1.8)**</td>
<td>2.43(2.0)</td>
<td>0.65(2.0)</td>
</tr>
<tr>
<td>Strategy</td>
<td></td>
<td>1</td>
<td>1.56(9.9)</td>
<td>7.42(12.2)*</td>
<td>5.17(12.4)*</td>
<td>2.41(9.9)</td>
<td>7.33(12.4)*</td>
</tr>
<tr>
<td>GrassOrigin</td>
<td></td>
<td>1</td>
<td>30.1(11)**</td>
<td>5.2(13.6)*</td>
<td>11.4(14.0)**</td>
<td>5.07(11.3)</td>
<td>4.39(13.3)</td>
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<tr>
<td>Irrigation x Strategy</td>
<td></td>
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<td>0.02(9.9)</td>
<td>2.21(12.2)</td>
<td>5.36(12.4)*</td>
<td>4.17(9.9)</td>
<td>3.76(12.4)*</td>
</tr>
<tr>
<td>Irrigation x GrassOrigin</td>
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<td>1</td>
<td>11.7(11)**</td>
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<td>0.90(14.0)</td>
<td>3.13(11.2)</td>
<td>4.08(13.8)</td>
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<td>0.25(13.8)</td>
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<tr>
<td>Irrigation x Strategy x GrassOrigin</td>
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<td>0.53(11)</td>
<td>0.83(13.6)</td>
<td>0.52(14.0)</td>
<td>7.00(11.3)*</td>
<td>6.28(13.8)*</td>
</tr>
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</table>

*P<0.06, **P<0.05, ***P<0.01, ****P <0.001
Figure 6. Numbers of actively growing seeded grass (seedlings/m²) through time for the fall+spring (left two panels) and spring-only (right two panels) irrigation regimes at the N (top two panels) and S (bottom two panels) sites for commercial (dotted lines) and local (solid lines) material in seeding strategy I (small lines) and II (large lines). Note the difference in scale between the N and the S site panels. Error bars are standard errors.
The irrigation regime also had considerable effects on seeded grass densities through time (Table 4, Figure 6). The fall+spring regime resulted in a significant flush of seedlings early in the first growing season, particularly at the S site. Most of the resulting seedlings appear to have either senesced early or died back through the winter and early spring (likely as a result of harsh conditions and/or competition with weeds that also took advantage of the fall irrigation), until the spring irrigation beginning in April created another flush of active growth. At the N site, this apparent die-back of active growth resulted in a lack of appreciable actively growing seedlings through the winter and early spring, whereas at the S site, some active growth was maintained in most plots despite this die-back being quite dramatic. After spring of the first growing season and through the second season, the effects of irrigation regime on grass densities diminished at both sites. The S site showed only a few cases of persistent benefits of the fall+spring regime through this period. The N site did not show any further significant main or interactive effects of irrigation regime. However, after examining the graphical results (Figure 6), we conducted a linear contrast on the grass densities at the N site for May 2016. This analysis did support the result that commercial grasses in strategy I plots had higher densities under the spring-only regime than under the fall+spring regime. We speculate that the high densities of grasses evident at the N site in the second season represents delayed emergence of grasses planted in the previous year. Because the spring-only regime resulted in fewer opportunities for germination (discussed above), perhaps this led to a larger seed-bank of dormant seeds from which second-year germinants arose. Continued monitoring will aid in determining the consequences of irrigation treatments on long-term success.

Effects of Seeding Strategy, Irrigation Regime, Presence of Seeded Grass, and Grass Seed Origin on Weed Densities

The only persistent effect of seeding strategy on weed densities were directly associated with the application of first-year herbicide (strategies I and IV), which significantly reduced weed densities at both sites (Table 5, Figure 7). This is a fairly straightforward result that demonstrates the effectiveness of broadleaf herbicides on weeds.

There is much interest regarding the effect of irrigation regime on weed densities in restoration efforts that use irrigation infrastructure in an attempt to aid the establishment of seeded material in semiarid locations. This is because there is the potential to dramatically increase weed levels through irrigation, which results in higher competition between weeds and desirable seeded plants. In this experiment, post-seeding fall irrigation generally increased weed densities early in the first season across both sites (Table 5, Figure 7), which may have had competitive impacts on early establishment of seeded species through the winter and spring of the first season. This could have increased competition between weeds and seeded species and accounted for some of the die-back described above for seeded grasses. Aside from this early
effect of irrigation regime, there were no other significant effects on weed density. In other words, although the more intensive irrigation regime (spring+fall) promoted more weeds early on relative to the less intensive regime (spring-only), there were few lasting differences between the two irrigation regimes.

Comparing weed densities in plots that were seeded with grasses to those that were not seeded with grasses is of interest for determining if seeded grasses can suppress weed levels. These comparisons showed significantly lower weed densities in the presence of grasses in only one site (N) and only during April and May of the first growing season (Table 5, Figure 7). This could be interpreted as evidence of a suppressive effect, which may have faded when the grasses (mostly cool-season) began to senesce, and weeds (many warm-season) continued to thrive. However, the small size and low densities of grass seedlings in that site at that time doesn’t suggest that grasses would inflict much suppression through competition on weeds. Additionally, the lack of any other evidence of suppression at either site for the rest of the sampling dates suggests that if weed densities were being suppressed by grasses, it was a weak and/or ephemeral effect in our study. Other patterns may emerge with additional monitoring and analysis of weed height data, which is not included here. The origins of the seeded grass and shrubs had no effects on weed levels (data not shown).

Table 5. ANOVA results ($F_{(DF, Den)}$) for the number of weeds encountered through time by site. Factors included irrigation regime (IRRIGATION; fall+spring, spring only), seeding strategy (STRATEGY; I-IV), and interactions. Data were log transformed, and bolded effects are significant. The origin of grass as well as shrub seed were originally added as factors to the model, but results were consistently insignificant and therefore omitted from the final analysis.

<table>
<thead>
<tr>
<th>Factor</th>
<th>DF</th>
<th>Number of actively growing weeds/m²</th>
</tr>
</thead>
<tbody>
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<td></td>
<td></td>
<td>Apr-15</td>
</tr>
<tr>
<td>N SITE</td>
<td></td>
<td></td>
</tr>
<tr>
<td>IRRIGATION</td>
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<td>51.98**</td>
</tr>
<tr>
<td>STRATEGY</td>
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<td>0.44</td>
</tr>
<tr>
<td>IRRIGATIONxSTRATEGY</td>
<td>4</td>
<td>1.20</td>
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<tr>
<td>S SITE</td>
<td></td>
<td></td>
</tr>
<tr>
<td>IRRIGATION</td>
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<td>9.46</td>
</tr>
<tr>
<td>STRATEGY</td>
<td>4</td>
<td>0.76</td>
</tr>
<tr>
<td>IRRIGATIONxSTRATEGY</td>
<td>4</td>
<td>3.55</td>
</tr>
</tbody>
</table>

*P < 0.05, **P < 0.01, ***P < 0.001
Figure 7. Logarithmic scale of the numbers of actively growing weeds (plants/m²) through time for the fall+spring (left two panels) and spring-only (right two panels) irrigation regimes at the N (top two panels) and S (bottom two panels) sites for all five seeding strategies (I is dotted, II is solid, III is dashed, IV is dash-dot, V is double solid). Error bars are standard errors.
PLANT-SOIL RELATIONSHIPS

At the restoration S site, heat maps of plant density clearly illustrate patchiness of seeded plants, volunteer native species, and weeds at the time of the last sampling in May 2016 (Figures 8-15). It is worth restating that some of the areas with low densities of particular plants are “cold spots” because these plots were not seeded with these plants. For example, in our experimental design, some plots were not seeded with perennial grasses. Additionally, negligible shrub emergence was observed in the first year (see above), so treatments where shrubs were seeded in year one are also cold spots. In addition to the analyses above, it is possible to see some patterns of plant emergence and establishment among species and/or life-forms with these visualizations. For example, emergence of some shrubs appears to be mutually exclusive with higher densities of perennial grasses, and this is most pronounced for rabbitbrush (Figures 9 and 13). The two *Atriplex* species may be more compatible with the perennial grasses, with high emergence in some of the perennial grass “hot spots” (Figures 10, 12, and 13). Our work to better understand spatial patterns of plant emergence and establishment is ongoing.

Figure 8. Heat map showing Wyoming big sagebrush density (plants/m$^2$) at the restoration study-S site. The x-axis is oriented N-S and the y-axis is oriented E-W.
Figure 9. Heat map showing rubber rabbitbrush density (plants/m²) at the restoration study-S site. The x-axis is oriented N-S and the y-axis is oriented E-W.

Figure 10. Heat map showing quailbush density (plants/m²) at the restoration study-S site. The x-axis is oriented N-S and the y-axis is oriented E-W.
Figure 11. Heat map showing black greasewood density (plants/m²) at the restoration study-S site. The x-axis is oriented N-S and the y-axis is oriented E-W.

Figure 12. Heat map showing fourwing saltbush density (plants/m²) at the restoration study-S site. The x-axis is oriented N-S and the y-axis is oriented E-W.
Figure 13. Heat map showing perennial grass density (plants/m$^2$) at the restoration study-S site. The x-axis is oriented N-S and the y-axis is oriented E-W.

Figure 14. Heat map showing native species density (plants/m$^2$) at the restoration study-S site. The x-axis is oriented N-S and the y-axis is oriented E-W.
In the native shrubland communities, vegetation characteristics were highly variable among sites (Tables 6, 7, and 8). Shrub canopy cover averaged 28.1 ± 3.1 percent (Figure 16), but ranged from 11.4 to 51.8 percent (Figure 17). Shrub canopy gap (100 percent – shrub canopy cover), and therefore averaged 71.9 percent, and ranged from 48.2 to 88.6 percent. In contrast, shrub basal gap was consistently high, averaging 96.0 percent and ranging from 89.4 to 98.8 percent. Shrub density averaged 0.7 plants/m² (Figure 18), but there was wide variation among sites, with values ranging from 0.3 to 1.2 plants/m². It should be noted that these values include shrubs and sub-shrubs. Perennial grasses and forbs were even more variable among sites. Perennial grasses averaged 2.6 plants/m² and ranged from 0 to 12.1 plants/m², whereas perennial forbs averaged 5.2 plants/m² and ranged from 0.1 to 16.7 plants/m².

Overall, we found that shrub (and sub-shrub) abundance followed the order (listed from most to least abundant (≥0.01/m²), averaged across all 15 native sites): bud sagebrush, black greasewood, shadscale, fourwing saltbush, gray molly, quailbush, Bailey’s greasewood, Wyoming big sagebrush, rubber rabbitbrush, shortspine horsebrush, and Nevada ephedra (Table 8). However, shrub species composition was also quite variable from site to site, with sites often dominated by two to three shrub species.
Table 6. Shrub cover (mean ± SE), shown as shrub canopy gap (% cover) and shrub canopy cover (100-canopy gap, as %) for each of the 15 native sites. Shrub basal gap (% cover) is also shown for comparison, but shrub basal cover (100-basal gap) is not shown here.

<table>
<thead>
<tr>
<th>Site*</th>
<th>Cover (%) ± SE</th>
<th>Canopy gap</th>
<th>Basal gap</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Shrub canopy (live+dead)†</td>
<td></td>
<td></td>
</tr>
<tr>
<td>A</td>
<td>44.3 ± 2.0</td>
<td>55.7 ± 2.0</td>
<td>97.1 ± 0.5</td>
</tr>
<tr>
<td>B</td>
<td>29.5 ± 4.8</td>
<td>70.5 ± 4.8</td>
<td>96.6 ± 0.7</td>
</tr>
<tr>
<td>DD</td>
<td>13.8 ± 3.5</td>
<td>86.2 ± 3.5</td>
<td>97.2 ± 0.1</td>
</tr>
<tr>
<td>E</td>
<td>29.9 ± 6.6</td>
<td>70.1 ± 6.6</td>
<td>93.6 ± 1.1</td>
</tr>
<tr>
<td>F</td>
<td>27.5 ± 1.9</td>
<td>72.5 ± 1.9</td>
<td>94.3 ± 1.6</td>
</tr>
<tr>
<td>G</td>
<td>20.3 ± 0.9</td>
<td>79.7 ± 0.9</td>
<td>98.4 ± 0.6</td>
</tr>
<tr>
<td>I</td>
<td>19.1 ± 0.6</td>
<td>80.9 ± 0.6</td>
<td>94.8 ± 3.8</td>
</tr>
<tr>
<td>II</td>
<td>14.6 ± 1.6</td>
<td>85.4 ± 1.6</td>
<td>97.6 ± 0.7</td>
</tr>
<tr>
<td>K</td>
<td>11.4 ± 1.0</td>
<td>88.6 ± 1.0</td>
<td>98.8 ± 0.4</td>
</tr>
<tr>
<td>PP</td>
<td>28.8 ± 4.3</td>
<td>71.2 ± 4.3</td>
<td>97.9 ± 1.3</td>
</tr>
<tr>
<td>QQ</td>
<td>51.8 ± 8.2</td>
<td>48.2 ± 8.2</td>
<td>94.4 ± 1.0</td>
</tr>
<tr>
<td>T</td>
<td>40.9 ± 5.9</td>
<td>59.1 ± 5.9</td>
<td>96.8 ± 0.2</td>
</tr>
<tr>
<td>W</td>
<td>19.2 ± 1.4</td>
<td>80.8 ± 1.4</td>
<td>96.7 ± 0.7</td>
</tr>
<tr>
<td>X</td>
<td>42.0 ± 2.0</td>
<td>58.0 ± 2.0</td>
<td>96.2 ± 0.6</td>
</tr>
<tr>
<td>ZZ</td>
<td>28.6 ± 8.3</td>
<td>71.4 ± 8.3</td>
<td>89.4 ± 6.9</td>
</tr>
</tbody>
</table>

* Site designations are alphabetical, as described in the Methods section.
† Notes: Shrub canopy cover includes both live and dead shrub cover.

Figure 16. Shrub cover, shown as (a) shrub canopy gap (% cover) and shrub canopy cover (100-canopy gap, as %), and (b) shrub basal gap (% cover) and shrub basal cover (100-basal gap, as %), averaged across all native sites (n=15).
Soil measurements and analyses are ongoing, and therefore we present only preliminary results. At the native sites, soil pH averaged 7.6 ± 0.1 but ranged from neutral to moderately alkaline values (Table 9). In comparison, pH was lower at the restoration S and N sites, averaging 7.4 ± 0.04 and 7.2 ± 0.2, respectively (Figure 19). Soils at these sites ranged from neutral to slightly alkaline. In general, soil EC$_{1:2}$ values in the native sites were low enough for soils to be classified as non-saline, except for site ZZ (Figure 20). Soil at site ZZ would be considered saline (i.e., salt-affected). The average EC$_{1:2}$ value at the restoration N site was two to three times higher than the native sites and the restoration S site, but the restoration N site soil would still be classified as non-saline.
Table 7. Plant density (#/m²) by life-form (mean ± SE) for each of the 15 native sites.

<table>
<thead>
<tr>
<th>Site</th>
<th>Perennial grasses</th>
<th>Perennial forbs</th>
<th>Shrubs (live)†</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>0</td>
<td>0.9 ± 0.5</td>
<td>0.6 ± 0.03</td>
</tr>
<tr>
<td>B</td>
<td>2.3 ± 1.7</td>
<td>4.9 ± 1.4</td>
<td>0.7 ± 0.2</td>
</tr>
<tr>
<td>DD</td>
<td>2.7 ± 0.9</td>
<td>2.2 ± 0.3</td>
<td>0.8 ± 0.1</td>
</tr>
<tr>
<td>E</td>
<td>5.9 ± 0.7</td>
<td>2.9 ± 0.6</td>
<td>0.9 ± 0.1</td>
</tr>
<tr>
<td>F</td>
<td>0.7 ± 0.7</td>
<td>4.6 ± 1.7</td>
<td>0.8 ± 0.05</td>
</tr>
<tr>
<td>G</td>
<td>2.8 ± 1.5</td>
<td>2.7 ± 1.3</td>
<td>0.8 ± 0.2</td>
</tr>
<tr>
<td>I</td>
<td>0.7 ± 0.4</td>
<td>15.8 ± 3.3</td>
<td>0.5 ± 0.1</td>
</tr>
<tr>
<td>II</td>
<td>5.6 ± 0.8</td>
<td>5.4 ± 1.5</td>
<td>0.7 ± 0.1</td>
</tr>
<tr>
<td>K</td>
<td>0.8 ± 0.3</td>
<td>16.7 ± 2.5</td>
<td>0.3 ± 0.01</td>
</tr>
<tr>
<td>PP</td>
<td>1.3 ± 0.7</td>
<td>9.8 ± 0.9</td>
<td>0.3 ± 0.01</td>
</tr>
<tr>
<td>QQ</td>
<td>0</td>
<td>0.4 ± 0.4</td>
<td>0.7 ± 0.04</td>
</tr>
<tr>
<td>T</td>
<td>0.1 ± 0.1</td>
<td>1.8 ± 1.1</td>
<td>0.7 ± 0.1</td>
</tr>
<tr>
<td>W</td>
<td>4.1 ± 1.3</td>
<td>3.1 ± 1.3</td>
<td>1.2 ± 0.03</td>
</tr>
<tr>
<td>X</td>
<td>0.7 ± 0.7</td>
<td>7.2 ± 3.9</td>
<td>0.6 ± 0.1</td>
</tr>
<tr>
<td>ZZ</td>
<td>12.1 ± 1.6</td>
<td>0.1 ± 0.1</td>
<td>0.5 ± 0.1</td>
</tr>
</tbody>
</table>

* Site designations are alphabetical, as described in the Methods section.
† Notes: Shrub density is shown for live shrubs only.

Figure 18. Plant density (#/m²) shown by lifeform (mean ± SE), averaged across all native sites (n=15).
Table 8. Shrub species composition (listed in decreasing order of abundance), separated into most common (≥0.1/m²) and least common species (<0.1/m²), for each of the 15 native sites.

<table>
<thead>
<tr>
<th>Site*</th>
<th>Shrub / Sub-shrub species (common name)††</th>
<th>Least common (&lt;0.1/m²)††</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>Fourwing saltbush, Wyoming big sagebrush</td>
<td>Black greasewood</td>
</tr>
<tr>
<td>B</td>
<td>Black greasewood, shadscale, quailbush</td>
<td>Bud sagebrush, shortspine horsebrush, Wyoming big sagebrush, gray molly</td>
</tr>
<tr>
<td>DD</td>
<td>Bud sagebrush, Bailey’s greasewood, shadscale</td>
<td>Spiny hopsage</td>
</tr>
<tr>
<td>E</td>
<td>Shadscale, black greasewood, bud sagebrush</td>
<td>Quailbush, rubber rabbitbrush, Bailey’s greasewood, Nevada ephedra, shortspine horsebrush, Wyoming big sagebrush</td>
</tr>
<tr>
<td>F</td>
<td>Gray molly, black greasewood</td>
<td>Shadscale, bud sagebrush</td>
</tr>
<tr>
<td>G</td>
<td>Shadscale, bud sagebrush, black greasewood</td>
<td>Gray molly, Wyoming big sagebrush, shortspine horsebrush, burrobrush, prickly pear</td>
</tr>
<tr>
<td>I</td>
<td>Shadscale, shortspine horsebrush, black greasewood</td>
<td>Wyoming big sagebrush, gray molly, burrobrush</td>
</tr>
<tr>
<td>II</td>
<td>Bud sagebrush, Bailey’s greasewood, shadscale</td>
<td>Prickly pear</td>
</tr>
<tr>
<td>K</td>
<td>Bailey’s greasewood</td>
<td>Winterfat, bud sagebrush, shadscale</td>
</tr>
<tr>
<td>PP</td>
<td>Black greasewood</td>
<td>Fourwing saltbush, Nevada ephedra, shortspine horsebrush, burrobrush</td>
</tr>
<tr>
<td>QQ</td>
<td>Gray molly, black greasewood, fourwing saltbush</td>
<td>Quailbush, Wyoming big sagebrush</td>
</tr>
<tr>
<td>T</td>
<td>Fourwing saltbush</td>
<td>Black greasewood</td>
</tr>
<tr>
<td>W</td>
<td>Bud sagebrush, shadscale, black greasewood</td>
<td></td>
</tr>
<tr>
<td>X</td>
<td>Shadscale, black greasewood</td>
<td>Fourwing saltbush, shadscale</td>
</tr>
<tr>
<td>ZZ</td>
<td>Rubber rabbitbrush, shadscale, black greasewood</td>
<td></td>
</tr>
</tbody>
</table>

* Site designations are alphabetical, as described in the Methods section.
†† Species names are as follows (listed from most to least abundant, averaged across all 15 native sites): bud sagebrush (*Picrothamnus desertorum*), black greasewood (*Sarcobatus vermiculatus*), shadscale (*Atriplex confertifolia*), fourwing saltbush (*Atriplex canescens*), gray molly (*Neokochia americana*), quailbush (*Atriplex torreyi*), Bailey’s greasewood (*Sacobatus baileyi*), Wyoming big sagebrush (*Artemisia tridentata ssp. wyomingensis*), rubber rabbitbrush (*Eriogonum nauseosum*), shortspine horsebrush (*Tetradymia spinosa*), Nevada ephedra (*Ephedra nevadensis*), winterfat (*Krascheninnikovia lanata*), burrobrush (*Hymenoclea salsola*), prickly pear (*Opuntia* spp.), and spiny hopsage (*Grayia spinosa*). When averaged across all sites, the last four species were found at densities <0.01/m². Note that big sagebrush subspecies identification is preliminary and will require verification.
†† Notes: Density of ≥ 0.1 individuals/m² is equivalent to one or more individual per 10 m². Species with densities <0.01/m² are not listed in the table.
Table 9. Soil characteristics for each of the 15 native sites, including pH (1:2 soil:water suspension) and salinity (electrical conductivity (EC 1:2 soil:water extract), dS/m).

<table>
<thead>
<tr>
<th>Site*</th>
<th>Soil characteristics (mean ± SE)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>pH1:2</td>
</tr>
<tr>
<td>A</td>
<td>7.5 ± 0.6</td>
</tr>
<tr>
<td>B</td>
<td>8.0 ± 0.1</td>
</tr>
<tr>
<td>DD</td>
<td>7.4 ± 0.4</td>
</tr>
<tr>
<td>E</td>
<td>7.6 ± 0.2</td>
</tr>
<tr>
<td>F</td>
<td>8.3 ± 0.2</td>
</tr>
<tr>
<td>G</td>
<td>7.5 ± 0.1</td>
</tr>
<tr>
<td>I</td>
<td>7.9 ± 0.3</td>
</tr>
<tr>
<td>II</td>
<td>6.9 ± 0.2</td>
</tr>
<tr>
<td>K</td>
<td>7.9 ± 0.03</td>
</tr>
<tr>
<td>PP</td>
<td>7.6 ± 0.4</td>
</tr>
<tr>
<td>QQ</td>
<td>8.1 ± 0.2</td>
</tr>
<tr>
<td>T</td>
<td>7.9 ± 0.5</td>
</tr>
<tr>
<td>W</td>
<td>7.1 ± 0.1</td>
</tr>
<tr>
<td>X</td>
<td>7.1 ± 0.3</td>
</tr>
<tr>
<td>ZZ</td>
<td>8.2 ± 0.3</td>
</tr>
</tbody>
</table>

* Site designations are alphabetical, as described in the Methods section.
†Notes: The soil at this site is saline.

ONGOING WORK

Vegetation at the restoration study (N and S sites) will be sampled again in spring 2017. The final results are expected to provide information for improving restoration techniques in post-agricultural fields in salt desert ecosystems within the Walker River Basin. Seeding strategies, irrigation levels, and shrub species will be evaluated for efficacy based on the three years of results. Additionally, soil measurements and analyses are currently ongoing for both the restoration study and the native sites, and will be finished in 2017. Once completed, we will continue with data summary and analysis. As described in detail in the Methods section, we will develop mathematical models to relate shrub density (restoration S site) and shrub cover (native sites) with measured soil characteristics. For the native sites, we will continue to summarize the data by individual site and overall to provide comprehensive baseline data to describe native salt desert shrubland communities. These baseline data will be useful for improved identification of target plant species (and communities) for restoration of post-agricultural fields in salt desert ecosystems. It may also allow for the potential tailoring of plant species to existing soil conditions at prospective restoration sites in similar environments. Information gathered on plant-soil relationships may allow for the prioritization of prospective restoration sites, which would be useful information given the common limitations of time and money in restoration efforts. The final results and interpretive assessment will be contingent on our completed datasets, to be finished in 2017.
Figure 19. Boxplots showing soil pH (1:2 soil:water suspension) for the native sites (n=30), restoration study-S site (n=28), and restoration study-N site (n=3).

Figure 20. Boxplots showing soil EC (1:2 soil:water extract, dS/m) for the native sites (n=30), restoration study-S site (n=28), and restoration study-N site (n=3). Note the break in the y-axis between 0.5 and 1.5 dS/m. The saline site, ZZ, is indicated on the figure.
SUMMARY

The results of our first two years of study suggest that seeding shrubs, especially quailbush, with short-term irrigation may be an effective way to establish a shrub component in former agricultural fields. Herbicide application is very effective at reducing broadleaf weeds, but at these lower densities, there appears to be only fleeting impacts of seeding grasses on weeds at one site. On our most difficult site, the most effective treatment at this point is the fall+spring irrigation combined with a one-year herbicide fallow followed by seeding locally sourced quailbush. Long-term monitoring will be important to determine the most effective treatment overall, because the survival of shrubs after irrigation has ceased and the recruitment from natural shrub sources are factors that could dramatically change these results over time.

PUBLICATIONS AND PRESENTATIONS DURING THIS REPORTING PERIOD (2013-2016)

JOURNAL ARTICLES


PRESENTATIONS


**LITERATURE CITED**


Jensen, K., H. Horton, Reed, R., Whitesites, R. Intermountain Planting Guide. USDA-ARS Publication AG510. USDA ARS Forest and Range Research Lab, Logan, Utah and Utah State University Extension


PRISM Climate Group, Oregon State University. 38°39'47"N, 118°58'20"W and 38°29’15”N, 118°58’12”W. [http://prism.oregonstate.edu](http://prism.oregonstate.edu) [Nov 18, 2016]


APPENDIX A: ARID OLD-FIELD RESTORATION: NATIVE PERENNIAL GRASSES SUPPRESS WEEDS AND EROSION, BUT ALSO SUPPRESS NATIVE SHRUBS
Arid old-field restoration: Native perennial grasses suppress weeds and erosion, but also suppress native shrubs

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A B S T R A C T

Rates of cropland abandonment in arid regions are increasing, and abandoned fields in such regions can have low levels of ecosystem function and biodiversity. Long-lived, drought-tolerant shrubs are dominant components of many arid ecosystems, providing multiple ecosystem services such as soil stabilization, herbaceous plant facilitation, carbon storage and wildlife habitat. On abandoned agricultural fields, shrub restoration is hindered by multiple challenges, including erosion, water stress and invasive species. We hypothesized that applying short-term irrigation and seeding native perennial grasses would facilitate native shrub establishment by reducing erosion and weed abundance. Using a blocked split-plot design, we evaluated the separate and combined impacts of short-term irrigation and perennial grass seeding on five-year restoration outcomes (including direct measurements of wind erosion) at two former agricultural fields in North America’s arid Great Basin. After two years, irrigation had increased the density and biomass of seeded grasses by more than ten-fold. The combination of irrigation and seeded grasses was associated with significantly lower wind erosion, weed density and weed biomass. Three years after irrigation ended, seeded grasses remained significantly more abundant in formerly irrigated than non-irrigated plots. Formerly irrigated plots also had significantly less bare ground, annual plant cover and weed biomass than non-irrigated plots. Large plant-canopy gaps were fewer in irrigated and seeded plots. Although seeded grasses reduced erosion and invasion, they failed to facilitate native shrub establishment. Shrub cover and density were highest in plots that had been drill-seeded and irrigated, but lacked perennial grasses. Our results indicate that short-term irrigation has persistent restoration benefits, and that a tradeoff exists between the benefits and costs of seeding perennial grasses into degraded arid shrubland sites.

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1. Introduction

Shrub establishment is often a central goal of restoration in arid regions. Long-lived, drought-tolerant shrubs dominate the plant communities of many arid ecosystems (e.g., Ackery, 2004; Miller et al., 2011; Wang et al., 2012), and these shrubs provide important functional benefits. For example, shrubs can stabilize soils, facilitate the establishment of other plants, store carbon, and provide critical wildlife habitat (Garcia-Estrin et al., 2013; Fonseca et al., 2012; Miller et al., 2011; Stavi et al., 2011; van Zonneveld et al., 2012). Shrubs can also increase ecosystem-level biodiversity, both by increasing the abundance and diversity of understory plants (van Zonneveld et al., 2012) and by providing resident and transient wildlife with habitat and forage (Miller et al., 2011).

On denuded sites, native shrub restoration is hindered by both abiotic and biotic challenges. Abiotic challenges include wind erosion and water stress, which can reduce seedling survival and growth (Maestre et al., 2001; Okin et al., 2008). Climate change will likely exacerbate abiotic challenges by increasing drought frequency and intensity (IPCC, 2012). Shrubs can also be difficult to restore due to the presence of mature grasses, which can limit or reduce shrub establishment (Boyd and Svejcar, 2011; reviewed by Meyer, 1992; but see Williams et al., 2002). To improve shrub restoration in degraded drylands, it may be necessary to actively mitigate these restoration barriers.
We studied shrub restoration on abandoned agricultural fields in a cold desert ecosystem. Globally, cropland abandonment has increased exponentially since the mid-1800s (Cramer et al., 2008), and abandoned agricultural fields represent an emerging focus of restoration ecology in arid regions. In Nevada, at the heart of North America’s arid Great Basin ecoregion, the amount of actively farmed land declined by 34% between 1992 and 2011 (USDA National Agricultural Statistics Service, 2012). Although passive restoration of agricultural fields may be possible in some circumstances (e.g., Scott and Morgan, 2012), a passive approach usually leads to slow or incomplete recovery (Otto et al., 2006; Munson et al., 2012), or further degradation (Jackson and Comus, 1999).

The long-term ecological legacies of agricultural abandonment in arid regions can include altered soil properties (e.g., less organic matter, less soil carbon, nitrogen and phosphorous, higher bulk density), altered plant communities (e.g., lower plant diversity, lower native plant abundance, lower total plant cover, lower cover of dominant shrubs, less forb cover) and reduced ecological stability (e.g., larger temporal fluctuations in plant cover, density and diversity, higher probability of conversion to a degraded state) (Burke et al., 1995; Elmore et al., 2006; Kawada et al., 2011; Morris et al., 2011; Munson et al., 2012; Xu et al., 2010). Agricultural abandonment without active restoration often leads to substantial wind erosion (Kawada et al., 2011; Okin et al., 2006), and the combination of soil disturbance, loss of vegetation, reduced native propulsive pressure, increased nutrient availability, and dense weed seed-banks makes abandoned fields highly susceptible to exotic plant invasion (Cramer et al., 2008; Elmore et al., 2006; Milton and Dean, 2010; Török et al., 2012). Thus, on arid old-fields, active restoration (e.g., soil remediation, herbicide application, or planting) is often necessary to improve ecosystem stability and function (Jackson and Comus, 1999; Otto et al., 2006; Munson et al., 2012).

Planting native perennial grasses in abandoned fields can prevent or reduce weed invasion (Bugge et al., 1997; Blumenthal et al., 2005; Török et al., 2012) and reduce wind erosion (Okin et al., 2006), mitigating some of the barriers hindering shrub establishment (Maestre et al., 2001). At the same time, co-occurring grasses and shrubs will likely compete for limited resources (Maestre and Cortina, 2004). Impacts of grasses on shrub success may vary depending on grass or shrub species identity (Maestre et al., 2001; Meyer, 1992) or resource availability (Maestre and Cortina, 2004). According to the stress-gradient hypothesis, positive interspecific interactions should be more common when stress is high (Bertness and Callaway, 1994), and several studies in dryland ecosystems have found evidence for greater plant–plant facilitation at more stressful sites (e.g., Arredondo-Núñez et al., 2009; Forey et al., 2009) or at more stressful times (e.g., Veblen, 2008). In contrast, other studies suggest that stress can lead to increased competition for scarce resources (e.g., Bowker et al., 2010; Holmgren and Scheffer, 2010; Odadi et al., 2011).

Irrigation infrastructure is still present on many abandoned agricultural fields in arid regions, and this allows restoration practitioners to modify stress by irrigating seedlings during the establishment phase. However, it remains unclear whether short-term irrigation translates into longer-term restoration success (Josa et al., 2012; Roundy et al., 2001). It is also unclear whether short-term irrigation will increase or decrease the likelihood of grass–shrub facilitation (Forey et al., 2009; Jankju, 2013; Maestre and Cortina, 2004; Maestre et al., 2001).

We used a broad-scale manipulative experiment to determine the separate and combined impacts of seeded perennial grasses and short-term irrigation on 5-year restoration outcomes at abandoned agricultural sites in the Great Basin, where little previous work exists. Our study addressed three specific research questions:

1. Does short-term irrigation increase the establishment and long-term survival of shrubs or grasses, and do the impacts of irrigation depend on grass species identity?
2. Do grasses mitigate potential shrub restoration barriers by suppressing weeds or reducing erosion, and do grass impacts depend on irrigation status or grass species identity?
3. Do grasses facilitate shrubs, and does facilitation depend on irrigation status or grass species identity?

Our results provide information about the likely outcomes of passive vs. active restoration on former agricultural fields in arid shrublands, as well as what specific restoration methods succeed best in arid ecosystems.

2. Materials and methods

2.1. Study sites and species descriptions

Two study sites were located along the lower reaches of the Walker River, 11.5 km south of Mason, Nevada USA. The Valley Vista Ranch (VV) site (38°50’58”N, 119°11’04”W) was used for alfalfa production until the start of the experiment, while the 5C’s Cottonwood Ranch (5C) site (38°50’45”N, 119°11’02”W) was a denuded pasture formerly used for burro and llama grazing. Both sites are located on Malpais (loamy-skeletal, mixed, superactive, mesic Typic Haplocambids) complex soils (dominated by Malpais gravelly sandy loam and Malpais stony sandy loam) (USDA Soil Conservation Service, 1984). Soil testing indicated that the sites have a moderately alkaline pH and low salinity (saturated soil paste, Bower and Wilcox, 1965), relatively high concentrations of water extractable nitrate (saturation extract, Bower and Wilcox, 1965),

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific name and authority</th>
<th>Season; variety</th>
<th>kg/km² Sown</th>
<th>Planting date</th>
<th>Seed preparation and planting</th>
<th>Code</th>
</tr>
</thead>
<tbody>
<tr>
<td>Inland saltgrass</td>
<td>Distichlis spicata (L.) Greene</td>
<td>Warm; Wildland collection</td>
<td>0.157</td>
<td>Jul 2008</td>
<td>Only cultipacker</td>
<td>Control</td>
</tr>
<tr>
<td>Indian ricegrass</td>
<td>Achnatherum hymenoides (Roem. &amp; Schult.) Barkworth</td>
<td>Cool; Nezpar, Rimrock</td>
<td>0.896</td>
<td>Dec 2007</td>
<td>Drill + cultipacker</td>
<td>Achy</td>
</tr>
<tr>
<td>Basin wildrye</td>
<td>Leymus cinereus (Scribn. &amp; Merr.) Á. Löve</td>
<td>Cool; Trailhead</td>
<td>1.12</td>
<td>Dec 2007</td>
<td>Drill + cultipacker</td>
<td>Leci</td>
</tr>
<tr>
<td>Western wheatgrass</td>
<td>Pascopyrum smithii (Ryd.) Á. Löve</td>
<td>Cool; Arriba, Rosana</td>
<td>1.34</td>
<td>Dec 2007</td>
<td>Drill + cultipacker</td>
<td>Pasm</td>
</tr>
<tr>
<td>Beardless wheatgrass</td>
<td>Pseudoroegneria spicata (Pursh) Á. Löve</td>
<td>Cool; Whitmar</td>
<td>0.896</td>
<td>Dec 2007</td>
<td>Drill + cultipacker</td>
<td>Pssp</td>
</tr>
</tbody>
</table>

Table 1
Seeded species and management methods for different subplots.
and relatively low concentrations of water extractable phosphate and most cations (Appendix A). Annual precipitation averages 127 mm but is highly variable across years. For 2008–2011, water year precipitation (October 1–September 30) ranged from 66 to 177 mm. Natural vegetation near the sites is dominated by long-lived desert shrub species such as Atriplex canescens (Pursh) Nutt., Atriplex confertifolia (Torr. & Frém.) S. Watson, Sarcobatus vermiculatus (Hook.) Torr., Artemisia tridentata subsp. wyomingensis Beetle & A.M. Young, and Ericameria nauseosa (Pall. ex Pursh) G.L. Nesom & Baird.

We planted five native grass species at each site: four cool-season perennial bunchgrasses known to survive in Great Basin regions receiving <300 mm of annual precipitation, and one warm-season, perennial rhizomatous grass with high drought- and salt-tolerance (Table 1) (USDA NRCS, 2013). We planted four native shrub species (A. canescens, A. confertifolia, S. vermiculatus, and A. tridentata subsp. wyomingensis) grown from wildland seed collections.

2.2. Experimental design

The experiment had a blocked split-plot design with three 56 × 54 m blocks at each site. Each block was split into two 56 × 27 m plots, with one plot assigned to the “Irrigation” treatment and the other to the “No Irrigation” treatment. Within each plot, six 9.3 × 27 m subplots were assigned to different seeding treatments, which included one unseeded control (intended to represent a field receiving no active restoration planting) and five grass monocultures (one per species). Though restoration practitioners typically plant species mixtures, here we planted grasses in monocultures so that we could compare the effects of different grass species on erosion, weed invasion and shrub establishment. For Achnatherum hymenoides and Pascopyrum smithii we planted a different variety in each half of each subplot. As commercial variety did not affect long-term performance for either species, data were pooled at the species level for final analysis. In one block per site, the same subplot treatment randomization was inadvertently applied to both irrigation plots. In the other blocks, subplot treatments were randomized separately within each irrigation plot. Seven shrubs (one per species, plus three others randomly assigned with respect to species) were transplanted into each subplot planted with perennial grasses.

2.3. Management and planting

To prepare sites for planting, we applied herbicides (glyphosate and dicamba to kill alfalfa on the VV site in June and August 2007) and mechanical treatments (ripping, diskling and floating on both sites in September 2007). To reduce annual weed pressure during the first growing season, we applied 2.4-D on all plots in May 2008 (56 kg/km² 2.4-D ester in 18.7 × 10³ L/km² of water) and mowed all plots biweekly until the end of the first growing season. For Distichlis spicata subplots, we also applied glyphosate (85 kg/km², 5.69 g/L concentration, ± 0.025% NIS) in late June 2008, prior to planting. No weed control efforts were undertaken in future years.

Cool-season grasses were planted in December 2007, and warm-season D. spicata was planted in July 2008. All species were sown at recommended seeding rates (Table 1) using a Truax drill with seeds placed 1.3 cm deep and followed by press wheels. All fall-planted subplots were also rolled with a cultipacker to form a firm seed bed, but D. spicata subplots were drilled-seeded only; in July the soil was sufficiently firm for planting without the cultipacker treatment. Shrubs were grown in 1.9 L plastic pots for two years in an outdoor location in Reno, NV. Individually-marked shrubs were hand-transplanted approximately 5 m apart into subplots (excluding Control subplots) in December 2008. In total, 118 A. canescens, 77 A. confertifolia, 93 S. vermiculatus, and 132 A. tridentata were planted across both sites.

Irrigation was applied using sprinklers (set on a 9.1 × 9.1 m pattern) that delivered approximately 7.6 L/min/sprinkler. Plots assigned to the irrigation treatment were watered approximately once per week from late spring (April–May) to mid-summer (July–August). Irrigation was discontinued when approximately 85 and 30 cm of water had been applied to each irrigation plot in 2008 (year 1) and 2009 (year 2), respectively. Irrigation plots only received natural precipitation after 2009.

Cattle grazing occurred during plant dormancy at the VV site in 2009, 2010, and 2011. This unplanned treatment was the result of animals from nearby alfalfa fields entering our restoration treatments during cold winter months in spite of electric fencing. Livestock grazing did not occur at the 5C site. To account for potential effects of grazing on restoration outcomes, “site” was included as a factor in all models (see Section 2.5).

2.4. Data collection

2.4.1. Vegetation and bare ground

In August 2009 (year 2) and late July 2011 (year 4), we monitored seeded grass density (no. of individuals/m²) and wet biomass (g/m²) using 25 × 25 cm quadrats. On each sampling date, five quadrats were randomly placed in each Leymus cinereus, Pseudoroegneria spicata, D. spicata and Control subplot. Subplots planted with A. hymenoides and P. smithii were sampled using 6 quadrats (three randomly-placed quadrats within each of the two varieties). We sampled weeds (non-native plants, which were primarily annual forbs) in years 2 and 4 using methods identical to those used for seeded grasses. We monitored the survival and height of transplanted shrubs in year 2 and year 4. To estimate cover of transplanted shrubs, we also measured the length and width of each shrub, then used the formula for the area of an ellipse (π × length/2 × width/2) to calculate area in m².

In year 4, plants were large enough to contribute substantially to ground cover, and we quantified percent cover of perennial grasses, shrubs, annual plants, litter, and bare ground using point-intercept sampling, in addition to the density and biomass measurements described above. Ten evenly-spaced points were sampled along each of three 9.3 m-long, randomly located and non-overlapping transects per subplot. To quantify longer-term effects of treatments on vegetation cover, erosion potential and shrub density, we took additional measurements in December 2012 (year 5) at the 5C site; the VV site transitioned to other land uses after year 4. We determined percent bare ground and percent cover of perennial grasses, shrubs, and annual plants using line-intercept sampling. Two randomly located and non-overlapping 9.3 m-long transects were sampled in each subplot. We used line-intercept data to calculate the frequency of large canopy gaps (>2 m) within each subplot. Shrub density was measured along each transect by counting all individuals found in a 9.3 × 4 m belt.

2.4.2. Wind erosion

Dust collector nests (Appendix A, Fryrear, 1986) were used to monitor the effects of restoration treatments on wind erosion. At each site, groups of four nests were placed on one non-irrigated plot and one irrigated plot. Nests were also installed at two ‘natural’ sites (covered by less-disturbed natural vegetation) located <400 m from the experimental sites, resulting in a total of six groups of four nests. Although different sized particles may move by different mechanisms, the larger sizes typically move at lower elevations relative to the soil surface, and movement occurs further from the ground when particle sizes are smaller or erosive force (shear velocity) is greater (Okin et al., 2006). Hence, each dust collector nest was equipped with four traps set at heights of 7, 35, 60 and
100 cm above the soil surface (Appendix A) to account for potential treatment-induced differences in particulate load distribution (i.e. larger particles closer to the surface or finer particles at greater heights).

Dust data were collected from year 2 (January 2009 for 5C and June 2009 for 5W) through year 4 (December 2011). Initially, the four nests in each treatment plot were arranged in a diamond formation with each nest located 1 m from the midpoint of a different plot edge (Appendix A). Nests were moved toward the center of each plot in April 2010 to minimize edge effects. After this adjustment, each diamond-shaped group of nests had a diameter of 20 m. One of the ‘natural’ groups had a diameter of 30 m throughout the experiment. Dust collection traps were emptied following major wind events (gusts >30 mph) or when there was a substantial break in wind activity, and samples were weighed in the laboratory.

For each wind event, prevailing wind direction (based on data from an on-site weather station) was used to determine which collector nests within each nest group would be designated as incoming and outgoing (Appendix A). For each trap height, we subtracted incoming nest sediment weights from outgoing nest sediment weights to estimate overall dust generation (+) or deposition (−) amounts. Totals were divided by the distance between incoming and outgoing nests to obtain milligrams of dust generated or deposited per meter traveled (Appendix A, Fryrear et al., 1998). Multiple nest heights allowed us to characterize changes in the sediment load distribution relative to the soil surface.

2.5. Data analysis

Data were analyzed using generalized linear mixed models. Response variables included planted grass density and biomass, weed density and biomass, and dust (years 2 and 4), percent cover of different plant types and bare ground (years 4 and 5), frequency of large canopy gaps and total shrub density (year 5), and transplanted shrub survival and size (years 2 and 4). If multiple measurements were taken per subplot, data were pooled at the subplot level (one value per subplot for analysis). Variance-weighting was used when variances were not homogenous, and response values were transformed when necessary.

For vegetation and cover data from years 2 and 4, random factors included site, block nested within site, and irrigation plot nested within block and site. For year 5 data (which were only collected at one site), random factors included block and irrigation plot nested within block. In dust models, random factors included site and dust collector group nested within site.

For vegetation and cover data, fixed factors included irrigation treatment, planted species treatment, and the irrigation x species interaction. Fixed factors in dust models included irrigation treatment, trap height, and the interaction x height interaction. If interaction terms were significant (P < 0.05), simple effects tests were performed. Interaction terms were removed from models in which they had P-values >0.10. In all cases, Tukey’s Honestly Significant Difference tests (α = 0.05) were used for post-hoc means comparisons. Analyses were completed in R (package nlme, Pinheiro et al., 2013; R Development Core Team, 2011). Results are reported as means ± 1 standard error.

3. Results

3.1. Planted grasses

In year 2, planted grass density and biomass were 0 in all Control and D. spicata subplots. To avoid heteroscedasticity, these subplots were excluded from further analysis. Cool season grass densities varied based on species identity and irrigation treatments (Fig. 1a; Table 2). In non-irrigated plots, all species occurred at similar (and low) densities (Fig. 1a; simple effects F2,15 = 1.1, P = 0.4). In irrigated plots, P. smithii density was over twice as high as densities of the other three grass species (Fig. 1a; simple effects F2,15 = 8.8, P = 0.001). For all four cool-season species, densities were significantly higher in irrigated than in non-irrigated plots (Fig. 1a; all simple effects P-values <0.004). Planted grass biomass was similar across cool-season grass species and was 200 times higher in irrigated than non-irrigated plots (Fig. 1c; Table 2).

By year 4, two years after irrigation ceased, planted grass density and biomass had declined to 48% and 38% of year 2 levels, respectively (Fig. 1b and d). All values remained 0 in Control subplots, suggesting that planted grasses had not spread beyond their subplot boundaries. Control subplots were excluded from further analysis to meet model assumptions. Planted grass density and biomass values in D. spicata subplots were ≤5% of values in subplots seeded with other grass species (Fig. 1b and d; Table 2). Across species, irrigated plots had at least 3 times more planted grass than non-irrigated plots (Fig. 1b and d; Table 2).

3.2. Weeds

In year 2, the density and biomass of weeds (i.e. non-native plants, which were primarily annual forbs) were 14 and 7 times higher in irrigated than non-irrigated plots, respectively (Fig. 2a and c; Table 2). Weed density was 3.5 times higher in D. spicata subplots (where perennial grasses were absent) than in A. hymenoides and P. smithii subplots (Fig. 2a and Table 2). Weed biomass was almost 7 times higher in Control and D. spicata subplots than in L. cinereus and P. smithii subplots, and 4 times higher in D. spicata subplots than A. hymenoides subplots (Fig. 2c; Table 2). By year 4, weed densities had increased in non-irrigated plots and weed biomass had declined across all plots (Fig. 2b and d). Irrigation and planted species had no significant effects on weed density or biomass in year 4 (Fig. 2b and d; P-values >0.10).

3.3. Long-term differences in understory plant cover, bare ground and canopy gaps

In year 4, perennial grass cover was 6.7 times higher in irrigated than in non-irrigated plots (Fig. 3a; Table 2). Subplots planted with D. spicata had significantly lower cover than subplots planted with A. hymenoides, L. cinereus, or P. spicata, while Control and P. smithii subplots had intermediate cover (Fig. 3a; Table 2). Note that this analysis included both seeded perennial grasses (11 ± 4% cover) and unseeded (volunteer) perennial grasses (3 ± 1% cover). Volunteer grasses were dominated by Sporobolus airoides.

Percent cover of annual plants (mostly forbs) was 37% higher in non-irrigated than irrigated plots (Fig. 3b; Table 2). Annual cover included both non-native species (e.g., Erodium cicutarium, Salsola iberica, Sisymbrium altissimum) and native species (mostly Amsinckia tessellata). Native and non-native annuals responded similarly to treatments. Bare ground was 76% more abundant in non-irrigated than irrigated plants (Fig. 3c; Table 2). Annual plant cover and bare ground did not differ significantly based on planted species (Fig. 3b and c; Table 2). Irrigation and planted species treatments did not significantly affect litter cover (P-values >0.07). In year 5, sampling at the 5C site revealed that treatment effects on plant cover and bare ground had not changed substantially since year 4 (Appendix B).

In year 5 at the 5C site, the effects of planted species treatments on the frequency of large (>2 m) gaps between plant canopies varied based on irrigation treatment (Fig. 3d; Table 2). In non-irrigated plots, gap frequency was similar across planted species and the
Fig. 1. Impacts of irrigation and seeded species on planted grass density ((a) and (b)) and biomass ((c) and (d)) in year 2 ((a) and (c)) and year 4 ((b) and (d)). Large letters indicate significant differences among seeded species across irrigation treatments (Tukey post-hoc means comparisons for main effects). Asterisks indicate significant differences among irrigation treatments within a given seeded species treatment, and small letters indicate significant differences among seeded species within a given irrigation treatment (Tukey post-hoc means comparisons for simple effects; letters a-c are used for irrigated plots and letters x-z for non-irrigated plots). "0" indicates that no plants were observed in a given treatment, and this treatment was excluded from analysis. In all cases, treatments sharing a letter are not significantly different. Disp = Distichlis spicata, Achy = Achnatherum hymenoides, Leci = Leymus cinereus, Pasm = Pascopyrum smithii, and Pssp = Pseudoroegneria spicata subsp. inermis.

Table 2
Summary of statistical results for vegetation, bare ground, and canopy gaps. Significant main effects or interactions of grass species and irrigation treatment are presented for all response variables.

<table>
<thead>
<tr>
<th>Response variable</th>
<th>Treatment effects</th>
<th>Irrigation</th>
<th>Brief description of result</th>
</tr>
</thead>
<tbody>
<tr>
<td>Seeded grasses</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Seeded grass density, year 2&lt;sup&gt;a&lt;/sup&gt;</td>
<td>Species × irrigation: $F_{5,30} = 8.03, P = 0.0004$</td>
<td>$F_{5,5} = 510, P &lt; 0.0001$</td>
<td>Higher density in irrigated plots; within irrigated plots, higher in P. smithii subplots</td>
</tr>
<tr>
<td>Seeded grass biomass, year 2&lt;sup&gt;a&lt;/sup&gt;</td>
<td>$F_{5,5} = 0.76, P = 0.5$</td>
<td>$F_{5,5} = 29, P = 0.003$</td>
<td>More biomass in irrigated plots</td>
</tr>
<tr>
<td>Seeded grass density, year 4&lt;sup&gt;b&lt;/sup&gt;</td>
<td>$F_{4,44} = 18, P = 0.0001$</td>
<td>$F_{5,5} = 16, P = 0.008$</td>
<td>More biomass in irrigated plots; less in D. spicata subplots</td>
</tr>
<tr>
<td>Seeded grass biomass, year 4&lt;sup&gt;b&lt;/sup&gt;</td>
<td>$F_{4,44} = 7.3, P = 0.0001$</td>
<td>$F_{5,5} = 18, P = 0.008$</td>
<td>More biomass in irrigated plots; less in D. spicata subplots</td>
</tr>
<tr>
<td>Weeds</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Weed density, year 2</td>
<td>$F_{5,5} = 5.3, P &lt; 0.0001$</td>
<td>$F_{5,5} = 18, P = 0.008$</td>
<td>Higher weed density in irrigated plots; higher in Control and D. spicata subplots</td>
</tr>
<tr>
<td>Weed biomass, year 2</td>
<td>$F_{5,5} = 6.9, P &lt; 0.0001$</td>
<td>$F_{5,5} = 18, P = 0.008$</td>
<td>More weed biomass in irrigated plots; more in Control and D. spicata subplots</td>
</tr>
<tr>
<td>Understory cover, bare ground, gaps</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Perennial grass cover, year 4</td>
<td>$F_{5,5} = 4.0, P = 0.004$</td>
<td>$F_{5,5} = 78, P = 0.0003$</td>
<td>Higher cover in irrigated plots; lower in D. spicata subplots</td>
</tr>
<tr>
<td>Annual cover, year 4</td>
<td>$F_{5,5} = 1.7, P = 0.2$</td>
<td>$F_{5,5} = 8.7, P = 0.03$</td>
<td>Lower annual cover in irrigated plots</td>
</tr>
<tr>
<td>Bare ground, year 4</td>
<td>$F_{5,5} = 1.5, P = 0.2$</td>
<td>$F_{5,5} = 8.1, P = 0.03$</td>
<td>Less bare ground in irrigated plots</td>
</tr>
<tr>
<td>Canopy gaps, year 5</td>
<td>Species × irrigation: $F_{3,30} = 2.8, P = 0.04$</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Shrub density, year 5</td>
<td>$F_{3,30} = 5.9, P = 0.001$</td>
<td>$F_{3,30} = 12, P = 0.07$</td>
<td>Higher shrub density in D. spicata subplots</td>
</tr>
</tbody>
</table>

<sup>a</sup> No seeded grasses in Control or D. spicata subplots, these subplots excluded from analysis to meet model assumptions.

<sup>b</sup> No seeded grasses in Control subplots, these subplots excluded from analysis to meet model assumptions.
Control treatment (Fig. 3d; simple effects $F_{5,10} = 0.39, P = 0.8$). In irrigated plots, Control and A. hymenoides subplots had >5 large gaps per 50 m segment, while L. cinereus and P. smithii subplots had no large gaps (Fig. 3d; simple effects $F_{5,10} = 6.6, P = 0.006$). For P. smithii, gaps were significantly more abundant in non-irrigated than irrigated plots (Fig. 3d; simple effects $P < 0.0001$).

3.4. Wind erosion

In year 2, natural sites and non-irrigated plots produced dust, while irrigated plots collected dust (Fig. 4a; $F_{2,6} = 20, P = 0.002$). Dust production tended to be highest at low trap heights (i.e. close to the ground; Fig. 4a; $F_{1,35} = 4.1, P = 0.052$). In year 4, dust production at
natural sites was highest at high trap heights, but dust production in irrigated and non-irrigated plots was highest at low trap heights (Fig. 4b; treatment × height F_{2,25} = 16, P < 0.0001). Across heights, irrigated and non-irrigated plots generated significantly more dust than natural sites in year 4 (Fig. 4b).

3.5. Shrubs

In August of year 2 (nine months after transplanting), roughly 25% of transplanted shrubs were still alive (1–3 per subplot). These included 71 A. tridentata, 29 A. canescens, 6 S. vermiculatus and 0 A. confertifolia (54%, 25%, 6% and 0% survival, respectively). Neither irrigation nor planted species significantly affected the percent survival of transplanted shrubs (Table 5a; species: F_{4,40} = 1.3, P = 0.3; irrigation: F_{1,5} = 0.44, P = 0.5; species × irrigation: F_{4,40} = 2.3, P = 0.07).

By year 4, two years after irrigation was discontinued, overall shrub survival rates had dropped to about 10%. Survivors included 22 A. tridentata, 14 A. canescens, 3 S. vermiculatus, and 0 A. confertifolia (17%, 12%, 3% and 0% survival, respectively). Survival rates in D. spicata subplots were more than twice as high as survival rates in plots planted with L. cinereus, P. smithii, or P. spicata (Fig. 5b; Table 2). Irrigation did not significantly impact survival in year 4 (Table 2). Neither irrigation nor planted species significantly affected shrub height or the average area covered by a surviving transplanted shrub (m² per shrub) in year 2 or year 4 (P-values >0.10).

In year 4, shrub cover (including both planted and volunteer shrubs) was 1.4 times higher in irrigated than non-irrigated plots (Fig. 5c; Table 2). Shrub cover in D. spicata subplots was 13 times higher than cover in other subplots (Fig. 5c; Table 2). To determine whether these results were driven by perennial grasses, we included perennial grass cover as a predictor in a modified model. We found that both perennial grass cover and species identity were significant drivers of shrub cover (grass cover: F_{1,53} = 46, P < 0.0001; (grass cover)^2: F_{1,53} = 25, P < 0.0001; planted species: F_{5,53} = 5.3, P = 0.0005). After accounting for grass cover and grass species identity, shrub cover no longer differed in response to irrigation treatments (F_{1,5} = 0.4, P > 0.5). In year 5, sampling at the 5C site revealed that treatment effects on shrub cover had not changed substantially since year 4 (Appendix B).

In year 5 at the 5C site, shrub density (including both planted and volunteer shrubs) was almost 7 times higher in D. spicata subplots than in other subplots (Fig. 5d; Table 2). Shrub density was 1.3 times higher in irrigated than in non-irrigated plots (Fig. 5d; Table 2). Across irrigation and species treatments, 98% (178) of the 181 shrubs encountered in our belt transects were volunteers rather than transplants. Volunteer shrubs included 104 A. canescens (58.4%), 72 E. nauseosa (40.4%) and 2 A. tridentata (1.1%).

4. Discussion

Restoration of former agricultural areas can be challenging due to altered soil chemistry and soil structure as well as extensive weed seed banks (Elmore et al., 2006; Kawada et al., 2011; Török et al., 2012), but irrigation infrastructure provides opportunities for restoration establishment that are rarely available on more natural sites (Roundy et al., 2001). In arid systems, shrubs may be the ultimate restoration goal, but planting grasses in the first year of a restoration provides a more rapid ground cover which may suppress weeds and reduce erosion (Okin et al., 2006; Török et al., 2012), and allows the use of broadleaf herbicides to suppress common agricultural weeds. Our results support previous studies suggesting that active approaches which include herbicide and irrigation can improve restoration success and reduce further degradation (e.g., wind erosion and weed invasion) in arid old fields (Jackson and Comus, 1999; Munson et al., 2012; Otto et al., 2006).

At our sites, short-term irrigation had long-term restoration benefits. Irrigating for two years improved the long-term abundance of perennial grasses and irrigated plots had fewer large vegetation gaps, even three years after irrigation ceased. Impacts of irrigation on initial grass establishment varied based on grass species identity, with P. smithii gaining the most establishment benefit from additional water (Fig. 1). However, this species-level effect was short-lived; two years after irrigation ended, grass densities were similar across all cool-season species (Fig. 1). Irrigation marginally improved the long-term cover and density of shrubs, but did not significantly affect the survival of transplanted shrubs.

Irrigation and seeding of cool-season grasses provided substantial short-term reductions in weed invasion and wind erosion (Figs. 2 and 4). The magnitude of these functional benefits was generally similar across all of the cool-season grass species, suggesting a minor role for species identity within this guild. However, species identity did affect the ability of grasses to provide long-term benefits such as reduction in canopy gaps. Large gaps between plants are an important indicator of wind and water erosion potential (Herrick et al., 2005; Okin et al., 2009). By 2012, perennial grass cover in most subplots was below 20% (Appendix B). However, irrigated subplots planted with L. cinereus or P. smithii maintained >20% cover and had fewer canopy gaps than other subplots (Fig. 3d). These results suggest that L. cinereus and P. smithii may be worthy of attention by restoration practitioners seeking long-term reductions in erosion potential.

Although seeded grasses ameliorated several potential restoration barriers during the period of shrub establishment, seeded
Grasses did not facilitate shrubs. In years 4–5, shrub cover, density and survival were highest in plots that were drilled and irrigated but lacked seeded grasses (D. spicata subplots), and shrub outcomes did not differ among different seeded cool-season gras ses (Fig. 5). Thus, any benefits shrubs received from seeded grasses (e.g., reduced weed pressure and reduced erosion) may have been offset by the costs of competing directly with grasses. Our work supports previous studies suggesting that facilitation can give way to competition under extreme resource stress (Holmgren and Scheffer, 2010; Maestre and Cortina, 2004; Maestre et al., 2009; Odadi et al., 2011).

If grasses hinder shrub establishment, should they be included in arid old-field restoration? Our results suggest that grasses do provide important functional benefits such as reduced erosion and invasion, especially in the short-term, despite their negative effects on shrubs. Thus, restoration practitioners may want to focus on how to improve shrub establishment without losing the functional benefits produced by perennial grasses. It may be possible to improve shrub success using temporal priority (i.e. planting shrubs before grasses) (Young et al., 2005) or spatial segregation (Porensky et al., 2012). Under the latter scenario, managers would plant grasses and shrubs in separate patches or strips. Fine-scale spatial segregation would reduce grass-shrub competition, allowing grasses to accrue weed and erosion reduction benefits in the short-term while shrub establishment proceeds for long-term site stability. An added benefit of a spatial approach is that it would allow for targeted shrub- or grass-specific weed control within different patches/ strips.

While transplanted shrubs exhibited low success (survival rates of ~10%; Fig. 5), many shrubs established from seeds that were either present in the seedbank or dispersed into our sites. This was especially true in the D. spicata subplots (discussed in detail below). For arid sites where appropriate root development is critical, shrubs that grow from seeds may have higher success than shrub transplants. Natural shrub populations were present near our sites and could easily have served as a seed source, especially for wind-dispersed species. Volunteer shrubs were dominated by A. canescens and E. nauseosa, both of which are wind-dispersed and set seed in the fall. Elmore et al. (2006) also noted high abundances of these two species on previously cultivated fields, suggesting that A. canescens and E. nauseosa may be good candidates for future restoration work at similar sites. Short-lived and early successional shrubs such as E. nauseosa are not usually included as target species for restoration, but these species may be able to effectively stabilize restoration sites and facilitate the establishment of later successional shrub species (e.g., Meyer and Monsen, 1990). More generally, our results point to the importance of landscape context, and particularly proximity to local seed sources, in determining restoration outcomes. Future research could further explore how to use remnant local plant populations to improve arid old-field restoration success. For example, it may be worthwhile to gather seed from remnant shrub populations and actively plant this seed on adjacent old-fields. As many species show evidence of adaptation to local climates (Leimu and Fischer, 2008), locally-sourced seeds may have higher per-plant establishment success than non-local seeds or transplants. Compared to non-local transplants, locally gathered seeds may also be lower cost (e.g., Palmerlee and Young, 2010).

We observed a remarkable difference in shrub performance between Control subplots (neither drilled nor seeded), where shrub performance was comparable to cool-season grass subplots, and D. spicata subplots (drilled, but lacking perennial grass), where both volunteer and transplanted shrubs were the most successful (Fig. 5). This suggests that drilling itself may help facilitate shrub establishment. The furrows created by our seed drill could have increased shrub performance by capturing wind-dispersed seeds, serving as favorable microsites, or improving water infiltration and soil moisture storage (van der Merwe and Kellner, 1999). Drilling may also have helped expose shrub seeds buried deeper in the soil column. Alternatively, other management factors unique to the D. spicata subplots (e.g., the fact that D. spicata was planted in July or the fact that D. spicata subplots were sprayed with glyphosate just prior to planting) could have facilitated shrub establishment. Further studies investigating the importance of summer or early-fall furrowing, combined with seeding of native shrubs, would be valuable in these systems.

**Fig. 5.** Impacts of irrigation and seeded species on transplanted shrub survival in year 2 (a) and year 4 (b), shrub cover in year 4 (c), and shrub density in year 5 (d). Refer to Fig. 1 for explanation of letters and codes.
Cool-season grass density and biomass were high during the first two years of the experiment, and densities were especially high in irrigated *P. smithii* subplots (Fig. 1). After irrigation was terminated, however, the density and biomass of seeded grasses in irrigated plots declined precipitously, and initial differences among seeded species disappeared. To some degree, declines in grass abundance probably reflected normal self-thinning processes. However, it is also possible that commercial grasses experienced unusually high mortality because they were poorly adapted to the climate of our study sites (Leimu and Fischer, 2008). All of the commercial varieties we used were grown from seeds collected at sites experiencing at least twice as much annual precipitation as Mason Valley, Nevada, and at our very stressful sites, locally-adapted grass seeds may perform better than the commercial varieties (e.g., Rowe and Leger, 2012). We did observe some colonization of our plots by perennial grass volunteers, though these did not dominate the plots. Perennial grass volunteers were most abundant in irrigated Control and *D. spicata* subplots, suggesting competition between seeded grasses and volunteer grasses.

5. Conclusions

Future monitoring will be critical in determining whether grasses and shrubs continue to survive at our sites. Long-term monitoring is especially important in dryland ecosystems where results can be strongly influenced by year effects (e.g., Cox and Anderson, 2004; Visser et al., 2004) and initial restoration successes can fade over longer time periods (e.g., Rinella et al., 2012). Nevertheless, our results point to several concrete management actions that may improve 5-year restoration outcomes in arid old fields:

(1) Short-term irrigation can increase long-term grass establishment and reduce erosion, and has minor positive effects on shrub establishment.

(2) Seeded, non-local perennial grasses can suppress weeds and reduce erosion, especially while the grasses are being irrigated. In general, these results are not dependent on grass species identity.

(3) Restoration projects located near remnant shrub populations may be able to take advantage of volunteer shrubs, which were quite successful in our plots.

(4) Drilling appears to improve shrub establishment, but seeded grasses may compete with rather than facilitate shrubs in arid ecosystems.

(5) Restoration treatments that seed grasses and shrubs in separate strips or islands may be most effective for achieving the combined goals of reducing short-term erosion, suppressing weeds and increasing long-term shrub establishment.

As agricultural abandonment continues to accelerate, the restoration of abandoned agricultural fields in drought-prone, arid shrubland ecosystems will become increasingly relevant. In order to restore ecosystem function at such sites, ecologists and practitioners must seek innovative strategies for simultaneously suppressing weeds, reducing erosion, and facilitating shrub establishment. Successful restoration of arid old-fields may represent an opportunity to mitigate ongoing habitat fragmentation and land degradation.

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Appendices A and B. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.agee.2013.11.026.

References


