Assessment of Post-Fire Rehabilitation of Desert Tortoise Habitat
Final Project Report

Contributions by

Lesley A. DeFalco[1], Sara J. Scoles-Sciulla[1], Cayenne Engel[2], Nathan A. Custer[1], Olivia K. Niziolek[1], and Todd C. Esque[1]

[1]Las Vegas Field Station
USGS Western Ecological Research Center
160 N. Stephanie Street
Henderson, Nevada 89074

[2]Department of Environmental and Occupational Health
University of Nevada Las Vegas
Las Vegas, Nevada 89154-3064

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For additional information, contact:

Steven Schwarzbach
Western Ecological Research Center
U.S. Geological Survey
7801 Folsom Blvd., Suite 101
Sacramento, CA 95826

Executive Summary

Landscape-scale disturbances such as wildfires fueled primarily by invasive annual grasses are becoming more common in the Mojave Desert and impacting sensitive wildlife habitat, particularly for the desert tortoise. Recent wildfires have prompted managers to take an active role in rehabilitating damaged shrublands. In an effort to reduce the frequency of fire occurrence and accelerate the recovery of burned desert tortoise habitat, preemptive actions are needed to identify areas with high fine fuel production and to develop approaches for reducing exotic annual grasses that fuel desert wildfires. At the same time, managers need a rigorous assessment of methods to actively rehabilitate disturbed tortoise habitat as well as determine which species have the most potential for success.

We developed a spatial model of the abundance and variability of annual plant production in desert tortoise habitat for prioritizing the reduction of fine fuels in Clark County, Nevada. We developed a set of linear models and compared them using a model selection approach to predict annual plant production based on published literature and expert opinion. These models included variables such as satellite-derived data (Normalized Difference Vegetation Index, NDVI; Enhanced Vegetation Index, EVI), interpolated data from weather stations (PRISM and NOAA precipitation and temperature models), and topographic digital data (elevation and aspect) as correlates for estimating annual plant biomass. We harvested peak annual plant production in 2010 and 2011 and compared these data with the models' estimates of biomass. Using Akaike's Information Criterion, we determined that the best prediction of annual plant biomass was derived from a fine fuels model originally developed for estimating fire risk in the Mojave Desert. Using this best model, we developed our final maps (average annual plant biomass and coefficient of variation during the period 2001 to 2012) for prioritizing areas for fine fuels management. Used in combination with our maps to identify and prioritize high fuel areas, site visits will help differentiate between areas that are dominated by exotic annual grasses and therefore likely to burn or re-burn (priority for aggressive fuels reduction) and those that are unburned and dominated by native annual species (priority for protection as high quality desert tortoise habitat).

We evaluated several rehabilitation approaches for re-vegetating burned desert tortoise habitat including seeding (native seed broadcast by aircraft or by hand), herbicide application (pre- and post-emergent), and outplanting greenhouse-raised seedlings of native species in areas burned by wildfires in 2005 and 2006. Hand seeding typically performed better than the one fire location that was seeded by aircraft. The native perennial forbs in the seed mixes—desert marigold (*Baileya multiradiata*) and desert globemallow (*Sphaeralcea ambigua*)—responded to seeding, while shrubs, perennial grasses, and annuals that were seeded at lower rates performed poorly. Herbicides successfully reduced shoot biomass of exotic annual grasses and forbs by >50% up to three years after treatment and promoted perennial plant establishment from the seed mix. Seeding at high rates in combination with herbicides, particularly those containing the active ingredient imazapic (Journey™ and Plateau™), substantially enhanced the density of seeded perennials. The reduction in exotic biomass coincided with their reduction in the soil seed bank two years after herbicide application of Journey™, Oust™ and Plateau™, and the increase in native annual biomass with Plateau™ was consistent with an increase in native annual grasses, particularly *Vulpia octoflora*, in the seed bank.

Outplanting of natives demonstrated that seedling survival of Nevada joint fir (*Ephedra nevadensis*) was 61% after 3.5 yrs, while survival of blackbrush (*Coleogyne ramosissima*) was only 4%. Soil moisture amendments and volunteer watering unexpectedly did not enhance long-term survival. Instead, joint fir survival was dependent upon unique site characteristics, and was decreased by the herbicide Journey™, which was applied one year following outplanting to reduce exotic annual plants. Blackbrush survival also was influenced by site, along with the type of herbivore protection (mesh cage better than plastic cone) and greenhouse source.

Few seeded species persisted in the seed bank, thereby limiting our recommendation of species for future seeding efforts. We found that the perennial herb, *Baileya multiradiata,* was consistently increased in the seed bank by seeding compared with all other seeded species, whether measured in the soil surface at 5 years (initially seeded at 268 seeds/m^2) or at 2 years post-seeding (1,559 seeds/m^2). However, this species occurred at seed densities that were considerably lower than the seeding rate, suggesting that we do not know the fate of many of the seeds broadcast in burned areas. Another seeded perennial herb *Sphaeralcea ambigua* had marginally greater densities in seeded areas, but also lower than the seeding rate of 356 seeds/m^2. The annual grass *Vulpia octoflora* was higher in seeded areas when seeded at 317 seeds/m^2 but not at 30 seeds/m^2. Seeding of this species was particularly successful in combination with the herbicide Plateau™, which suppressed competitive exotic annuals and promoted native grass seed production. While these results suggest that extremely high seeding rates are necessary to achieve elevated densities in the seed bank, other species seeded at high rates did not have elevated densities in the seed bank including the summer annual *Bouteloua aristidoides* (181 seeds/m^2 and 4,068 seeds/m^2), the winter annual *Malacothrix glabrata* (104 seeds/m^2), the perennial grass *Achnatherum hymenoides* (288 seeds/m^2), and the shrubs *Atriplex canescens* (169 seeds/m^2) and *Hymenoclea salsola* (116 seeds/m^2). All other species in seed mixtures for these two experiments were seeded at rates <100 seeds/m^2 and were typically undetected in the seed bank. Whereas these studies indicate that fast-growing herbaceous perennials are suitable for re-vegetating burned areas, optimal seed germination conditions, activity of seed harvesting ants and rodents, and the facilitative and competitive interactions between plants are important considerations when planning to re-seed burned Mojave Desert shrubland habitats. The large loss of seeds we measured during our re-seeding efforts emphasizes the importance of selecting less-palatable, early-colonizing species to stabilize burned sites, while simultaneously optimizing seeding rates and protecting seeds from granivores.

Introduction

Background and Need for Project

Landscape-scale disturbances in the Mojave Desert result in the loss of native vegetation and have increased in frequency and extent during recent decades (Lovich and Bainbridge 1999, Brooks and Matchett 2006, Webb et al. 2009). Most notably, wildfires in the eastern Mojave Desert burned an unprecedented one million acres during the summers of 2005 and 2006 (Brooks and Matchett 2006). In addition, the footprint of alternative energy development—such as solar arrays and wind farms—is expected to grow and encompass vast areas of low and middle elevation shrublands in the coming decades. Although Mojave Desert shrub species can resprout from mechanical damage to the root crown (Gibson et al. 2004), many species are poorly adapted to burn injury and do not predictably recover following wildfires (Brown and Minnich 1986, Webb et al. 2003, Brooks et al. 2007, Esque et al. 2003, DeFalco et al. 2009). Low-elevation (creosotebush/bursage/Joshua tree) and mid-elevation (blackbrush) shrublands of the Mojave Desert are poorly adapted to wildfires. Historically rare, desert wildfires are now fueled by abundant exotic annual grasses that have invaded and dominated shrublands in recent decades. Exotic annual grasses such as red brome (*Bromus madritensis* ssp. *rubens*), cheatgrass (*Bromus tectorum*), and Mediterranean grass (*Schismus* spp.) arrived in California during the late 1800s and spread throughout the Mojave Desert during several decades of increased winter precipitation coinciding with Pacific Decadal Oscillations (Salo 2002). By providing continuous ground cover between shrubs, exotic annuals fuel wildfires that consume shrubs and perennial grasses leaving a landscape vulnerable to re-invasion by exotic annual grasses (Brooks et al. 2007). The accumulation of exotic grasses perpetuates the loss of native shrubs, bunchgrasses and annuals by fueling successive fires thereby degrading the seed bank and shrublands that support native plant and wildlife habitat (Esque et al. 2003, 2010).

Growing concern for landscape-level disturbances such as desert wildfires has spurred the need to predict the spatial and temporal distribution and production of exotic annual species in particular. While direct ground-level harvests of production provide accurate estimates of fine fuels, they are time-consuming, labor-intensive, and cost-prohibitive when characterizing broad desert landscapes (Miller and Yool 2002). Remote sensing technology has potential for providing resource managers a time-efficient and cost-effective alternative. Recent remote sensing efforts in the Mojave and Sonoran deserts have estimated spring growth potential and biomass production of annual plants by correlating ground-based estimates of canopy cover with satellite-based imagery (Wallace and Thomas 2008, Casady et al. 2013). In addition, fire risk in the northeast Mojave Desert was recently predicted by indirectly estimating peak spring production of annual species through the cover-based fuel model FIREMON (Van Linn 2013). Fine fuel production can vary greatly for a given unit area of canopy cover due to variation in plant heights and the differing allometries among annual species (Casady et al. 2013, DeFalco unpublished data). The high variability in canopy architecture of desert plants may diminish the effectiveness of satellite-based predictions of annual plant production (Casady et al. 2013). Alternatively, because annual plant production is correlated with seasonal and interannual rainfall (Beatley 1974, Bowers 1987), spatial interpolations of environmental variables, such as monthly precipitation and temperature minima and maxima, also have potential for predicting peak annual plant production. Such spatially-explicit climate data are becoming more publicly available through the internet; however, their resolution (e.g., 4 km scale) may be too coarse to capture the dynamics in weather patterns that influence germination, emergence and growth patterns of annual plant communities. A comparison between approaches that use remote sensing and interpolated climate data for estimating annual plant biomass would be informative, but has never been attempted.

Much of the low elevation desert shrubland that has burned is habitat for sensitive wildlife species including the Mojave desert tortoise (Nussear et al. 2009). Plant structure provides thermal cover, protection from predators, nest sites, and hunting perches to desert wildlife in general (Miller and Stebbins 1963), and for the Mojave desert tortoise in particular (Nussear 2004, Nussear and Tuberville, *In Press*). The largest herbivorous reptile in the Mojave Desert, the desert tortoise is intimately tied to plants in its environment for food (Esque 1994, DeFalco 1995, Esque et al. *In Press*). Shrub and perennial grass roots also armor soils against erosion and stabilize soils for tortoise burrow construction (Soulard et al. 2011). While the loss of important habitat components for the tortoise is a major concern for recovering populations, burned Mojave shrublands are slow to recover naturally and have proven difficult to repair using conventional restoration methods (Abella and Newton 2009).

Arid land restoration is an ancient concept (Bainbridge 2007), but rehabilitation of North American deserts damaged by wildfires is in its infancy, in part due to the complex relationships among exotic annual grasses, precipitation pulses/interpulse droughts, and wildfire. The limited studies on desert rehabilitation to date typically focus on a single climate zone and/or are short-term (1-3 yrs); thus, the mechanisms driving restoration success and failure are poorly understood. Furthermore, surface disturbances diminish native plant seed banks, which reside in the top several centimeters of soil in the Mojave Desert and represent the regeneration potential for native shrublands (Esque et al. 2010, DeFalco et al. 2009). The recovery of native vegetation in the Mojave Desert following large disturbances is also hindered by the characteristic lack of rainfall, broad temperature fluctuations, and extreme summer temperatures that create an inherently challenging environment for plants to reproduce, germinate, establish and develop (Webb et al. 2003, Turner 2004). The Mojave Desert is a pulse-driven system with brief, but intense precipitation events followed by long seasonal and even decadal interpulse periods with fluctuating resources available to plants (Schwinning et al. 2004). In addition, climate scenarios portend a future with hotter, drier, and more extreme swings between pulse and interpulse periods in the desert southwest (Seager et al. 2007). Collectively, these challenges make restoration planning and implementation difficult at best and underscore the importance of selecting and adequately vetting restoration methods and native species that can adapt to an uncertain future.

Description of the Project

We developed a network of long-term monitoring sites in the northeast Mojave Desert following the 2005 and 2006 wildfire seasons in collaboration U.S. Bureau of Land Management and U.S. Fish and Wildlife Service. An evaluation of rehabilitation treatments began in fall 2006 following the broadcast seeding of native annual and perennial species either by hand (Bonnie Springs, Dry Middle, Dry Rock, Garnet, Tramp, Halfway, and Meadow Valley fires), and in fall 2008 by aircraft (Goodsprings Fire). During fall 2008, four herbicides in combination with seeding were applied to experimental plots associated with the original Dry Rock, Dry Middle, Garnet, Tramp, Halfway, and Meadow Valley fires. Finally, during fall 2007, several thousand native seedlings were transplanted into the Fork, Tramp, Goodsprings, and Loop fires. Although located outside of Clark County, standardized monitoring at sites in the 2005 Meadow Valley and Halfway fires (Lincoln County) help provide a broad perspective across the northeast Mojave Desert on single-occurrence wildfires. We monitored native and exotic annual plant production, density and canopy cover of establishing perennials, and the soil seed bank associated with the herbicide treatments (2 yrs post treatment) and the seeding treatments (5 yrs post treatment). Collectively, these "layers of rehabilitation methods" across a broad weather gradient, provide insights into the best approaches for managers facing continuously changing disturbance regimes in Mojave Desert shrublands with emphasis on desert tortoise habitats in particular.

The first aspect of this project, hereafter referred to as Biomass Prediction, evaluated a set of potential models for estimating peak spring annual plant production, which represents fine fuels, in order to identify areas that could be the focus for fuels reduction in Clark County desert tortoise habitat. By identifying areas with high average annual plant production and low interannual variability, managers can proactively identify areas for fine fuels control using such methods as herbicide application. Instead of focusing on a single predictor for estimating production, we used a model selection approach to determine the most plausible statistical model among a set of candidate models that represent remote sensing and spatially interpolated environmental predictors.

The second aspect of this project, hereafter referred to as Rehabilitation Treatment Assessment, evaluated the success of broadcast seeding (by hand and by aircraft), herbicide application in combination with seeding, and seedling outplanting as methods for rehabilitating burned desert tortoise habitat. The success of these rehabilitation techniques was examined in light of environmental conditions (i.e., precipitation and temperature) to better understand the scenarios where each method would have the greatest chance of success.

The third aspect of this project, hereafter referred to as Evaluation of Native Species, evaluated the relative performance of the individual native species used during seeding and seeding/herbicide treatments. We document the species that persisted in the soil seed banks of burned areas that were seeded with a native seed mix in 2005/06 (5-yrs post-treatment), and areas that were seeded in combination with herbicide in 2008 (2-yrs post-treatment). We use the results of our seeding/herbicide study, which had high rates of seeding for a variety of species (ranging from 30 to >4,000 seeds/m^2), to understand how abiotic factors (winter/spring minimum temperatures, spring maximum temperatures, precipitation), plant competition (annual plant biomass, density of perennial plants), and facilitation (canopy cover of perennials) influence recruitment and mortality for species in the seed mixture. Collectively, we discuss the suitability of species that have been used for post-fire rehabilitation in tortoise habitat and provide insight into future seeding efforts.

Management Actions Addressed

This project benefits several MSHCP Permit Conditions. Specifically, “The Permittees, in cooperation with the AMP contractor and the Service, shall participate with the land management agencies in development and/or revision of conservation management plans that identify the management and monitoring actions needed for the following areas or Covered Species:

- MSHCP(2-4) J1b-d. 1. Low Elevation Uplands: b) Coyote Springs Valley DWMA; c) Mormon Mesa DWMA; d) Gold Butte DWMA
- MSHCP(11)J4. Low Elevation Plant Species

The following actions will benefit the desert tortoise by reducing the conversion of burned Blackbrush and Mojave Desert Scrub to a fire regime due to the introduction of exotic annuals. These actions will also reduce the risk of fire due to exotic plants (such as red brome):

- Conservation/Management action BLM(207), which includes implementing inventory, monitoring and research projects dealing with management issues within desert tortoise areas of critical environmental concern, will be implemented in the Coyote Springs Valley ACEC, Red Rock Canyon National Conservation Area, Bird Springs ACEC, Whitney Pocket’s ACEC, Devil’s Throat ACEC, Gold Butte Part B ACEC, and Gold Butte Townsite ACEC.

- Conservation/Management action BLM(38), which includes establishing and reading vegetation trend monitoring plots in desert tortoise Critical Habitat, will be implemented in Mormon Mesa and Gold Butte-Pakoon Critical Habitat units. Results of this project will also contribute to Conservation/Management action BLM(135) by testing the efficacy of, and determining the best plant species for, reseeding with native plant species and other soil stabilization and habitat restoration actions following wildfires within areas important for the conservation of covered species such as the desert tortoise.

- Results of this project will also contribute to Conservation/Management action NDOW(23) as we cooperated with USFWS and others to restore native vegetation communities on public and private lands.

- Although this project was not specifically conducted on NPS or USFWS lands, results of this project have implications for Conservation/Management action NPS(43) and USFWS(38) by defining if reseeding with native plant species and other soil stabilization and habitat restoration actions following fires within the LMNRA and DNWR, respectively, are appropriate to implement.

Goals and Objectives

Project goals and specific objectives to meet these goals were as follows:

Goal 1, Biomass Prediction: Predict areas with high fine fuel production in desert tortoise habitat.

Objective 1a: Collect plot-level rainfall data to validate use of spatially-explicit precipitation and temperature models.
Objective 1b: Quantify peak production of annual plants across multiple field sites as a measure of fine fuels.
Objective 1c: Correlate fine fuel production with model output of precipitation (soils, if available).
Objective 1d: Identify areas for fine fuel management based on model.

Goal 2, Rehabilitation Treatment Assessment: Determine recommended rehabilitation treatments that are appropriate for burned tortoise habitat.

Objective 2a: Monitor native and exotic plant establishment in areas treated after the wildfires.
Objective 2b: Evaluate influence of climate (and soils, if available) on plant establishment in treated areas.
Objective 2c: Evaluate appropriate rehabilitation treatments.

Goal 3, Evaluation of Native Species: Identify appropriate Mojave Desert species for rehabilitating burned desert tortoise habitat.

Objective 3a: Establishment of seed bank.
Objective 3b: Determine persistence of species in the seed bank of seeded areas.
Objective 3c: Evaluate adequacy of selected species and seeding rates on plant establishment.
Objective 3d: Estimate recovery times for plant cover and community composition.

Materials and Methods

Goal 1, Biomass Prediction

Study sites and ground-based environmental data

Our study occurred in and adjacent to areas burned by the Southern Nevada Fire Complex in summer 2005 (Fig. 1). All fires were ignited by lightning strikes and fueled by abundant annual plant production resulting from above average winter rainfall during 2005. The study area encompasses low to mid elevation shrublands dominated by creosote bush (*Larrea tridentata*)/bur-sage (*Ambrosia dumosa*)/Joshua tree (*Yucca brevifolia*) or blackbrush (*Coleogyne ramossisima*) (Brooks and Matchett 2006). The study area occurs on Federal land managed by the Bureau of Land Management (BLM) and within designated critical habitat for the threatened desert tortoise.

Seventeen long-term monitoring sites were established in early 2006. At each site, two 40-acre monitoring plots (400 m × 400 m) were positioned in burned desert tortoise critical habitat and assigned either the seeded or the untreated control treatment (see below for full details of seeding study); a third monitoring plot with similar topography was positioned in each adjacent unburned area to serve as a long-term reference site (total monitoring plots = 51). The seventeen sites vary in precipitation, temperature and elevation and represent a broad range of variability in desert tortoise habitat in the Dry Middle (N=3 sites), Dry Rock (N=6 sites), Garnet (N=1 site), Halfway (N=2 sites), Meadow Valley (N=2 sites), Bunkerville (N=1 site), and Tramp fires (N=2 sites). Although funded by other sources, we included data from the Bunkerville (Clark County), and Meadow Valley and Halfway fires (Lincoln County) to represent a broad range of annual plant production. Comparison between burned and adjacent unburned areas suggests these sites have no indications of burning within past decades, except the Bunkerville Fire site. The Bunkerville Fire has signs of repeated burning including absence of long-lived species, lack of canopy cover development, and dominance by ruderal annual grasses and forbs in the adjacent unburned area.

Monthly precipitation data were collected using manual rain gauges initially installed in 2006 for long-term monitoring of the post-fire rehabilitation treatments. Manual gauges were installed at the northwest corner of each monitoring plot and visited to record monthly precipitation (Tru-Chek® rain gauge, Edwards Manufacturing Company). In addition to manual gauges, we established one self-logging HOBO® weather station (Onset Computer Corporation, Pocasset, MA) near the center of each burned area. The weather stations logged rainfall and air temperature every 30 minutes. We used weather data from November 2007 through May 2013 for comparison with spatially-explicit models.

Peak production of annual plants

The spring peak production for annual grasses and forbs was sampled from each seeded, unseeded, and unburned monitoring plot during April 5 to May 11, 2010, and March 28 to May 30, 2011. During each year, twenty-seven to thirty sampling quadrats (20 cm × 50 cm = 0.1 m^2) were randomly placed within each monitoring plot. The number of quadrats sampled in each year was based on adequacy of sampling calculations (Elzinga et al. 2001). Shoots were cut at ground level, placed in a paper bag, and transferred to the laboratory where samples were dried in a convection oven (50 °C to a constant mass) and weighed. Shoot biomass was averaged over all quadrats to obtain a representative value for each monitoring plot.

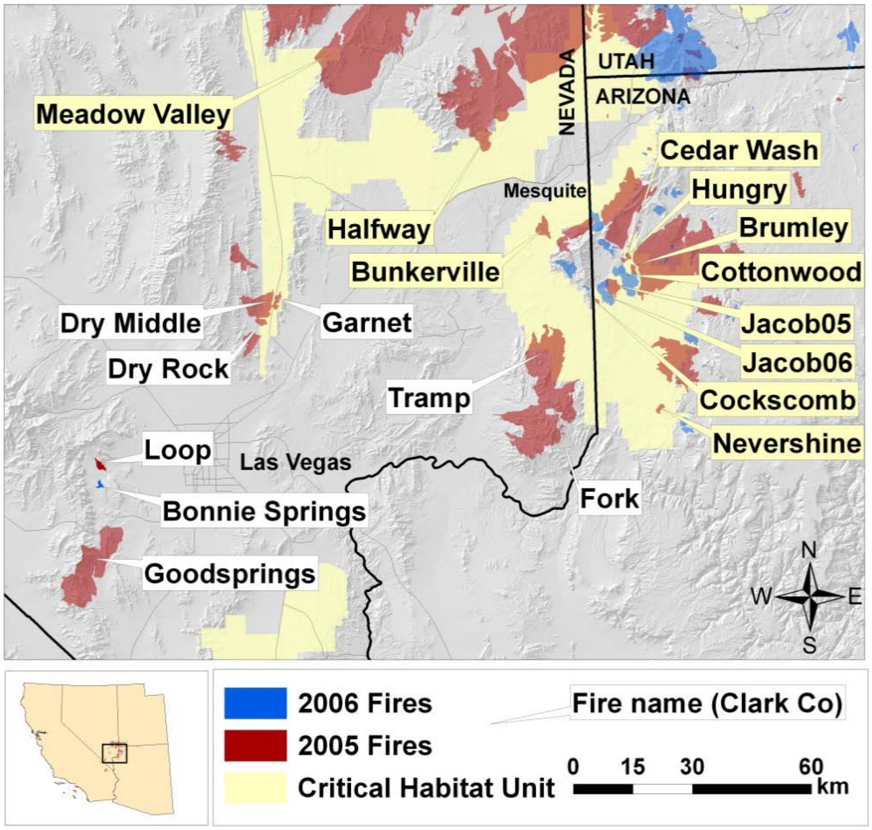

Figure 1. Mojave Desert long-term monitoring sites located within fires burned during the summers of 2005 and 2006. Sites represent once-burned habitat for the Mojave desert tortoise (Nussear et al. 2009). Effectiveness monitoring for seeding and herbicide/seeding treatments reported here occurred at the Dry Middle, Dry Rock, Garnet, Tramp (Clark Co.), and Meadow Valley and Halfway fire locations (Lincoln Co., funding provided by BLM and USGS). Other sites are located in Goodsprings and Bonnie Springs fires (seeding effectiveness) and Loop, Tramp, Fork and Goodsprings fires (outplanting effectiveness). Other sites are mapped to illustrate extent of monitoring network, but are not reported here.

Development of candidate models for estimating peak annual plant production

We created 51 separate linear regression models, including an intercept-only model, for estimating annual plant production, and similar models were grouped into "families" of candidate models. These models were based on Mojave Desert annual plant literature and on expert opinion of the authors. Due to the lack of heteroscedasticity in the residuals, we log-transformed each value before relating annual plant biomass in our candidate models. See Deliverable 31 for the full candidate set of models we used to relate environmental variables to peak annual plant biomass. Model variables representing spatial environmental data and remote sensing data were derived as detailed in the following sections.

Spatial environmental data

The Near Real Time Monthly High Resolution Precipitation, average monthly maximum (T_{max}) and minimum temperatures (T_{min}) Climate Data Set for the Conterminous States were downloaded from the PRISM (Parameter-elevation Regression on Independent Slopes Model) database (ftp://prism.oregonstate.edu//pub/prism/us/grids/, PRISM Climate Group, 2004). ASCII files were extracted and imported as raster files into ArcGIS 10.0 with a sampling resolution of 4 km and units of mm×100 for precipitation and C°×100 for temperature. Comparable precipitation data were downloaded from the NOAA database (http://www.noaa.gov/), and similarly, ASCII files were extracted and imported into ArcGIS 10.0 as raster files with a sampling resolution of 4 km and units in inches. Meta-data downloaded from the databases were attached within the ArcGIS 10.1 interface. Modeled monthly temperature and precipitation data from PRISM, and precipitation data from NOAA, were compared with data from our weather stations to determine how well the models correlated with the plot-specific data.

We obtained elevation and aspect values using the 30 m resolution Digital Elevation Model (DEM) for Clark County and surrounding counties from the National Map Viewer and Download Platform (www.nationalmap.gov/viewer.html). Elevation values were directly extracted from the DEM model while aspect was extrapolated using ArcGIS 10.0 “3DAnalyst Tools, Raster Surface Aspect Tool.” The resulting raster was in arc degrees (0-360°), and a sine-transformation converted values to a linear scale representing degrees from true north (-1.0 to +1.0). Values were then extracted for the center of each monitoring plot.

Remote sensing data

To obtain Normalized Difference Vegetation Index (NDVI) and Enhanced Vegetation Index (EVI) remote sensing data, the 250-m resolution, 16-day L3 Global MOD13Q1 product was downloaded from (http://reverb.echo.nasa.gov/reverb/). NDVI and EVI images from within the time period when biomass sampling occurred (April 5 to May 11, 2010 and March 28 to May 30 2011) were examined to determine which 16-day product represented peak annual plant production. One MODIS13Q1 raster representing peak production (starting April 23, 2010 and April 7, 2011) for each year was designated as “SpringNDVI/SpringEVI”.

Model selection procedure for estimating peak annual plant biomass

We used an information-theoretic approach to determine the most plausible model that explained peak annual plant biomass across both years and all sites in tortoise habitat from our families of models that combined precipitation, aspect, elevation, T_{max}, T_{min} and/or remote sensing data. The use of Akaike’s Information Criterion corrected for small sample size (AICc) is favored over traditional methods such as stepwise, backward or forward selection because it considers existing knowledge about the biology or ecology of the system when developing the set of candidate models (Burnham and Anderson 2002).

Extracted point values for all model input variables were uploaded into R (R version 2.15.1 (2012-06-22, Copyright © 2012 The R Foundation for Statistical Computing)) where each linear model was built using the “ln” function or “quadln.” The lowest AICc value denotes the “best” model, or most plausible explanation, out of the set of models that were considered. The models were ranked by rescaling AICc values so that the model with the minimum AICc had a value of 0. The difference calculated from this best model, ΔAIC, allows comparisons with the remaining models: $\Delta AIC < 2$ suggest substantial support for the model, ΔAIC between 4 and 7 suggest considerably less support, and $\Delta AIC > 10$ suggest essentially no support for the model (Burnham and Anderson 2002). AICc, ΔAICc, k parameters, and cumulative weights (w_i) were obtained using the “AICcmodavg” package within the R interface while adjusted r^2 values and intercepts were calculated using the summary ln statistic. The importance of each plot attribute (a value ranging between 0 and 1 for least and most important, respectively) was derived from all the candidate models by summing the Akaike weights across all the models where the attribute occurred (Burnham and Anderson 2002).

Spatial representation of fine fuels in Clark County

We portrayed "fine fuels" in two maps: average annual plant biomass map (abundance) and a coefficient of variation map (variability) estimated during 2001-2012. The monthly climate grids of precipitation, minimum and maximum temperature were statistically downscaled from their original spatial resolution of 4 km to 250 m using a modified gradient-inverse-distance-squared approach (Flint and Flint 2012). We used slope in addition to elevation as environmental gradients, and used generalized additive models rather than linear regression to perform climate downscaling. Using Model Builder Raster Math within the ArcGIS interface, composite raster files representing annual plant production based on the selected model were created for each year. Again, using Model Builder Raster Math, resulting annual plant biomass estimates were averaged across the 12 year period to create the final maps. Rasters were clipped to Clark County and the desert tortoise habitat model mask composed of values ranging from 0.5-1.0 (Nussear et al. 2009). Values of average and coefficient of variation were binned into seven categories based on quantiles. Major roads, Clark County land ownership, and an impervious surface layer (ie., human populated areas) were overlaid on the final maps.

Goal 2, Rehabilitation Treatment Assessment

Part I. Monitoring seeding effectiveness for augmenting native species establishment on the Goodsprings and Bonnie Springs fires, Red Rock Canyon National Conservation Area

Seeding

The Bureau of Land Management (BLM) conducted a seeding by aircraft in early 2006 on the 2005 Goodsprings Fire (32,000 acres in size) in Red Rock Canyon National Conservation Area (Fig. 2). 2,500 acres of the northern extent of the burn was seeded in six different seeding units (ranging in size from 311 to 1,114 acres). The following species were seeded with a thermic blackbrush mix: *Achnatherum speciosum* (desert needlegrass), *Atriplex canescens* (fourwing saltbush), *Coleogyne ramosissima* (blackbrush), *Ephedra nevadensis* (Mormon tea), *Eriogonum fasiculatum* (California buckwheat), *Krascheninnikovia lanata* (winterfat), *Purshia tridentata* (Antelopebush), and *Sphaeralcea ambigua* (desert globemallow; Table 1).

BLM organized a hand-seeding effort in 2008 on the 2006 Bonnie Springs fire in Red Rock Canyon National Conservation Area (Fig. 3). Seeded species include *Aristida purpurea* (purple threeawn), *Baileya multiradiata* (desert marigold), *Sphaeralcea ambigua* (globemallow), *Salvia columbariae* (chia), and *Hymenoclea salsola* (cheesebush; Table 2).

Because seeding was conducted in multiple units within the burn, monitoring specifically compared seeding effectiveness across the landscape to see if seeding is preferentially effective across sites that may have different vegetative and geologic attributes.

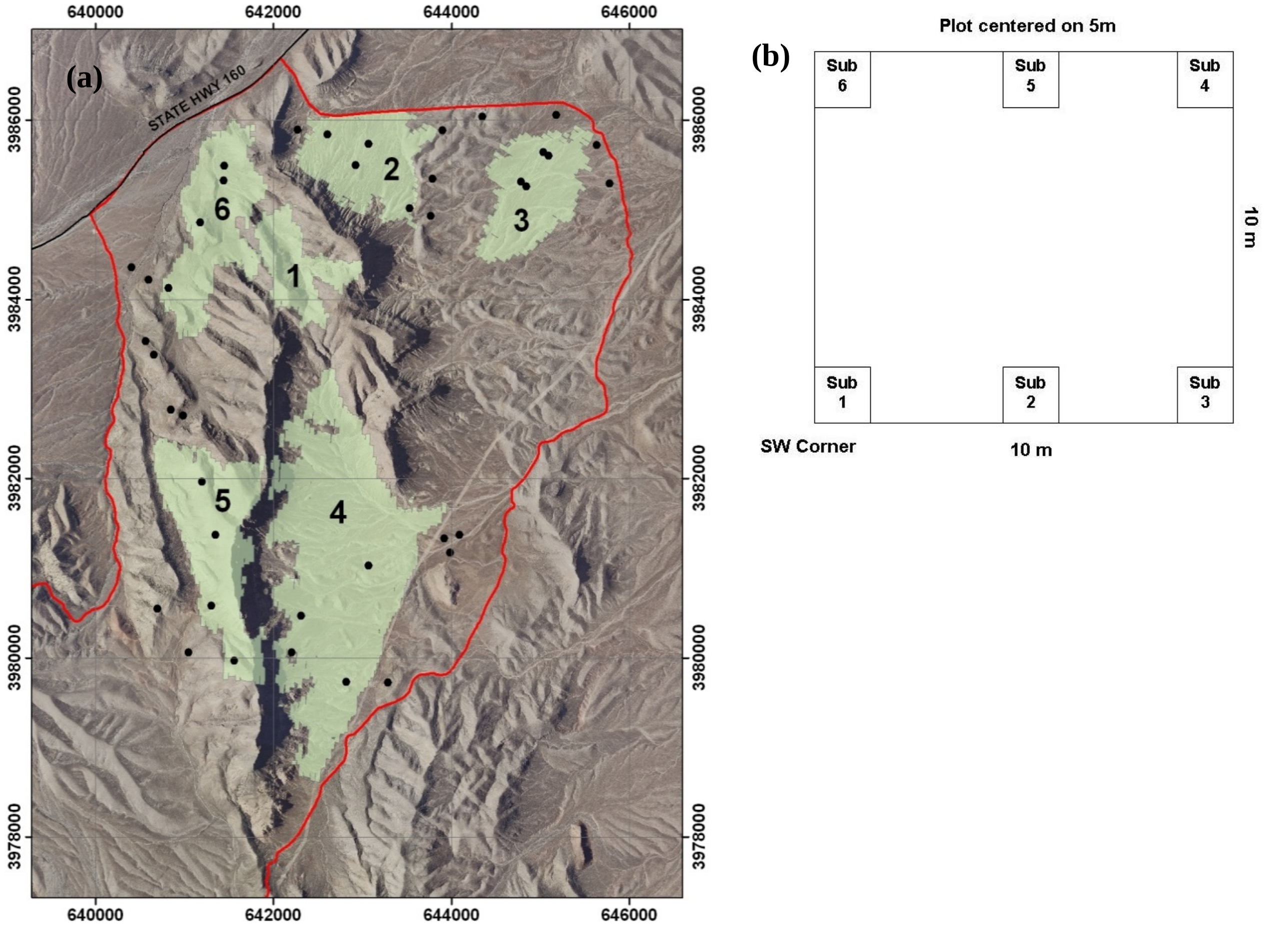

Figure 2. (a) Map with sampled plot layout (black dots) for each of six seeded sites (green shading represents the seeded areas) within the Goodsprings Fire (red outline). UTMs around the perimeter of the map are a 2 km grid in NAD 83 zone 11 projection. Site 1 was excluded from sampling due to lack of appropriate control sites. **(b)** Schematic of 10 m × 10 m plot setup with 1 m^2 subplots used to sample at both the Goddsprings and Bonnie Springs fires. Four plots were sampled within each of five seeding sites throughout the northeastern section of the Goodsprings Fire, with four adjacent unseeded control sites.

Table 1. Properties of the thermic blackbrush mix distributed by aerial seeding across 2,500 acres of the Goodsprings Fire. PLS = Pure Live Seed

Seeded Species	Common Name	Final Mix (Bulk lbs)	PLS lbs	Bulk lbs/acre	PLS lbs/acre	Bulk seeds/acre	PLS seeds/acre
Achnatherum speciosum	Desert Needlegrass	100	77.14	0.04	0.03	3	2
Atriplex canescens	Four wing saltbush	15,100	7,021.75	6.04	2.81	42,411	19,722
Coleogyne ramosissima	Blackbrush	350	338.35	0.14	0.14	47	46
Ephedra nevadensis	Nevada Ephedra	1,100	632.69	0.44	0.25	278	160
Eriogonum fasciculatum	California Buckwheat	750	123.20	0.3	0.05	37	6
Krascheninnikovia lanata	Winterfat	4,000	1,848.93	1.6	0.74	2,958	1,367
Purshia tridentata	Antelope Bitterbrush	2,100	1,699.62	0.84	0.68	1,428	1,155
Sphaeralcea ambigua	Desert Globemallow	1,200	961.39	0.48	0.38	461	370

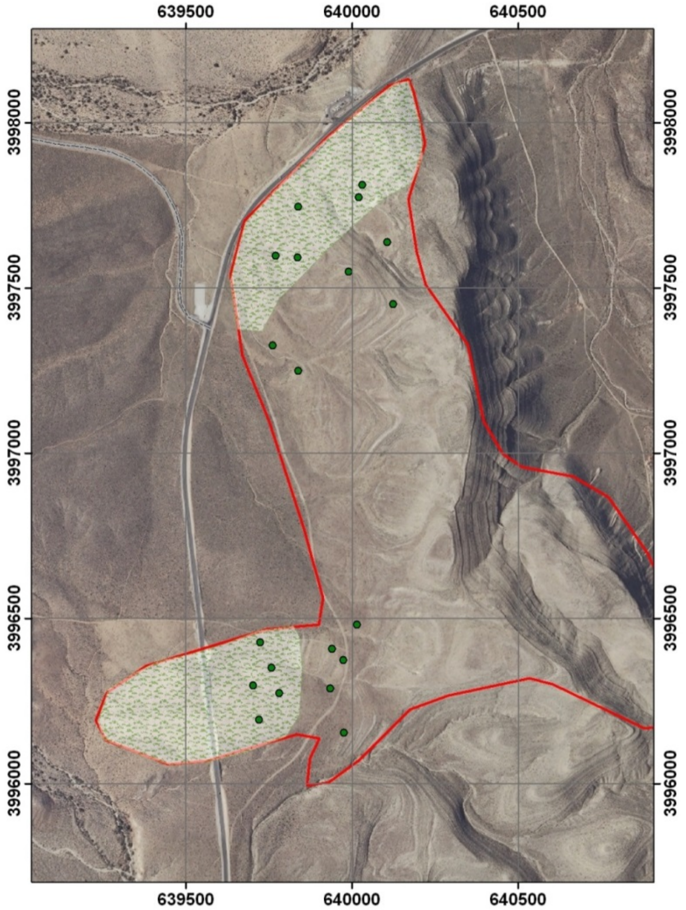

Figure 3. Plot placement (green dots) within the seeded (patterned polygons) and control sites of the Bonnie Springs Fire (red outline). Grid is 500 m UTM, NAD 83, zone 11.

Table 2. Properties of the seeding mix hand distributed across 100 acres of the Bonnie Springs Fire. PLS = Pure Live Seed.

Seeded Species	Common Name	Final Mix (Bulk lbs)	PLS lbs	Bulk lbs/acre	PLS lbs/acre	Bulk seeds/acre	PLS seeds/acre
Sphaeralcea ambigua	Desert Globemallow	250	158	2.5	1.58	1,250,000	496,125
Baileya multiradiata	Desert Marigold	100	72	1	0.72	1,060,000	549,504
Aristida purpurea	Purple Three-awn	165	99	1.65	0.99	495,000	178,200
Hymenoclea salsola	Cheesebush	200	128	2	1.28	220,000	90,112
Salvia columbariae	Chia	250	214	2.5	2.14	916,240	669,794

Monitoring methods

Four plots were randomly located using the random plots function in the NPS AlaskaPak in ArcMap v. 9.3 within five of the six seeded units (sites) at the Goodsprings Fire, along with four unseeded control plots that were established outside each site (40 sampling plots total). One of the seeded units was excluded from the sampling because the seeding was applied to a valley with steep walls and there were no adjacent unseeded areas of similar topographical and vegetative composition to serve as control sites. Control plots (unseeded) were placed 100 - 500 m from the edge of the accompanying seeded site, and are located in sites of similar topography as the seeded plots in the paired site. Ten plots within the seeded region of the Bonnie Springs Fire were randomly selected along with 10 paired unseeded control sites (burned but unseeded). The plots external to the seeded sites are located in sites of similar topography as the associated seeded plots, and are located within 300 m of the seeded region. Each plot is 10 m × 10 m (0.01 hectare; 0.025 acre; layout identical to that of Goodsprings seeding monitoring).

Each 10 m × 10 m (0.01 hectare; 0.025 acre) plot was sampled during the period of greatest active vegetative growth March – May 2010 and 2011. We recorded the density (counted individuals) of seedlings of the seeded species and recorded all plant species cover and richness within the entire plot. We used a sampling scheme where we positioned six 1 m^2 subplots within each plot where we recorded seedling counts of seeded species as well as annual and perennial plant identity and foliar cover (Fig. 2b). Cover assessments were recorded a on a numerical scale where 1 = trace (0.1%), 2 = 0-1%, 3 = 1-2%, 4 = 2-5%, 5 = 5-10%, 6 = 10-15%, 7 = 15-25%, 8 = 25-50%, 9 = 50-75%, 10 = 75-95%, 11 = >95%. Identity and abundance (cover) of species that were not captured in the subplot sampling were recorded in the whole plot. Subplots are used to capture species that may be missed with a general ocular site survey. Species captured at the subplot are averaged and scaled to the plot level cover assessments. Live and dead *Bromus rubens* were recorded separately to account for current and previous year's brome grass abundance.

Soil seed bank samples were collected from interspace soils from a 0-5 cm depth (7 cm diameter cores, 193 cm^3) at each of the four whole-plot corners, approximately 1 m inside the plot boundary (precise location dependent on perennial plant distribution) in October 2010. The four samples from each plot were thoroughly homogenized, and 400 cm^3 was placed atop commercial potting soil in a 15 cm × 15 cm square pot and grown out in the UNLV greenhouse for a period of approximately 6 months. Two emergence methods were conducted for each sample. First, we provided ample water daily (two sessions of 5 minute misting per day throughout the length of the experiment). Second, we utilized a methodology described by DeFalco et al. (2009), which cycles through periods of wetting, drying, and additions of gibberrelic acid and KNO_3 to stimulate emergence of individuals with complex dormancy mechanisms. Samples were monitored for emergence 2-3 times per week and irrigated with an automatic watering system twice daily. Seedlings were identified, recorded, and removed from pots weekly or bi-weekly as appropriate.

Data analyses

Cover classes were converted to the mean value for their class (e.g., class 3 is given a value of 1.5%). These converted cover classes were then used to perform statistical analyses. Data from subplots were averaged within each plot and species only present in the whole plot but missed in subplot sampling were added to form a matrix of species cover values.

For the Goodsprings Fire, seedling abundance per plot was compared among treatments (control and seeding) and year (2010 and 2011) using a partially hierarchical model in which site was the subject, and seeding and sampling year were within-subject effects. Data were analyzed in SAS software, using the Proc Mixed statement (SAS v. 9.2, SAS Institute 2002-2003). Data were checked for normality and any post-hoc analyses were interpreted using a Tukey's procedure. Due to a sparsity of seedling observations, only one species, *Sphaeralcea ambigua* (desert globemallow) had sufficient observations to conduct an ANOVA analysis. Data from all other species are presented descriptively.

For the Bonnie Springs Fire, seedling abundance per plot was compared among treatments (control and seeding) and year (2010 and 2011) using repeated measures analysis. Data were analyzed in SAS software, using the Proc GLM statement, with seeding treatment as the model fixed effect and sample year (2010 or 2011) as the repeated measures effect (SAS v. 9.2, SAS Institute 2002-2003). Data were checked for normality using the univariate procedure normality curves and Shapiro-Wilk W statistic. ANOVA analyses were performed on *Baileya multiradiata* and *Sphaeralcea ambigua*, which were present in nearly every plot. The frequency of occurrence of other species resulted in a majority of zeroes in the dataset, and are therefore interpreted descriptively. *Baileya multiradiata* was log transformed to meet assumptions of normality. Post-hoc analyses were interpreted using a Tukey's procedure.

Community composition (abundance of all species within a plot) between seeded and unseeded sites was explored using non-metric multidimensional scaling (NMS) ordination with a multiple response permutation procedure (MRPP) test for examining differences among groups using PC-Ord software (McCune and Mefford 1999). A significant p-value within the MRPP represents differences among groups (in this case, control vs. seeded). The NMS ordination allows a visual representation of community composition at the plot level. Data were converted to relative cover for all community composition analyses.

Community level metrics of richness (number of species per square meter) and diversity (calculated as Shannon's diversity index) were analyzed for treatment effect (control and seeding) and year (2010 and 2011) using a partially hierarchical model in which site was the subject, and seeding and sampling year were within-subject effects. Data were analyzed in SAS software, using the Proc Mixed statement (SAS v. 9.3, SAS Institute 2008).

Seeds were summed across each of the emergence techniques for each plot, resulting in a single density count for each plot, one from the consistent water emergence method, one from the DeFalco method. Data from both emergence techniques were summed before scaling up from the two-225 cm^2 growout pots to 1 m^2 for more effective interpretation. No seeded species were found within the soil seed bank, therefore statistical analyses were not performed on the data.

Part II. Evaluation of hand seeding and herbicide application for rehabilitation of Mojave Desert tortoise habitat

Seeding and herbicide application

Rehabilitation treatments occurred at sites burned in 2005 (Southern Nevada Fire Complex) on federal land managed by the Bureau of Land Management (BLM). These sites represent "single, recent" burns with no evidence of repeated plant injury (i.e., recently burned skeletons of late-successional species still dominate the plant community). The study sites occur within designated critical habitat for the threatened Mojave desert tortoise (Fig. 1). All fires occurred in low- to mid-elevation habitat dominated by creosotebush-bursage-Joshua tree or mixed blackbrush shrublands. In 2006, USGS worked in concert with BLM to establish paired seeded and unseeded plots in multiple fire locations for a total of 16 monitoring locations, each with a seeded and unseeded monitoring plot. Each monitoring plot was 400 m × 400 m (40 acre) for evaluating the effectiveness of hand seeding desert tortoise critical habitat. A native seed mix was applied to seeded treatment plots in December 2005; however, due to the limited availability of diverse species in this seeding, a second seeding occurred in October 2006 once additional seed could be obtained (Table 3). To evaluate the effectiveness of herbicide application in combination with hand seeding, ten 10 m × 10 m experimental plots were established in 2008 adjacent to each 40-acre seeded plot that had been established in 2006 (Fig. 4). Seeding (2 levels) and herbicide (5 levels) treatments were randomly assigned to the ten experimental plots. Seeding treatments included a native seed mix hand-broadcast in October 2007 or a non-seeded control (Table 3). The five herbicide treatments included: non-herbicide control, 100% plot coverage of 3.2 oz/acre of Oust™ (active ingredient sulfometron methyl) or Plateau™ (imazapic) applied as pre-emergent herbicides in October 2008, and RoundUp™ (glyphosate) or Journey™ (imazapic+glyphosate) spot-applied to *Bromus* rosettes in February 2009.

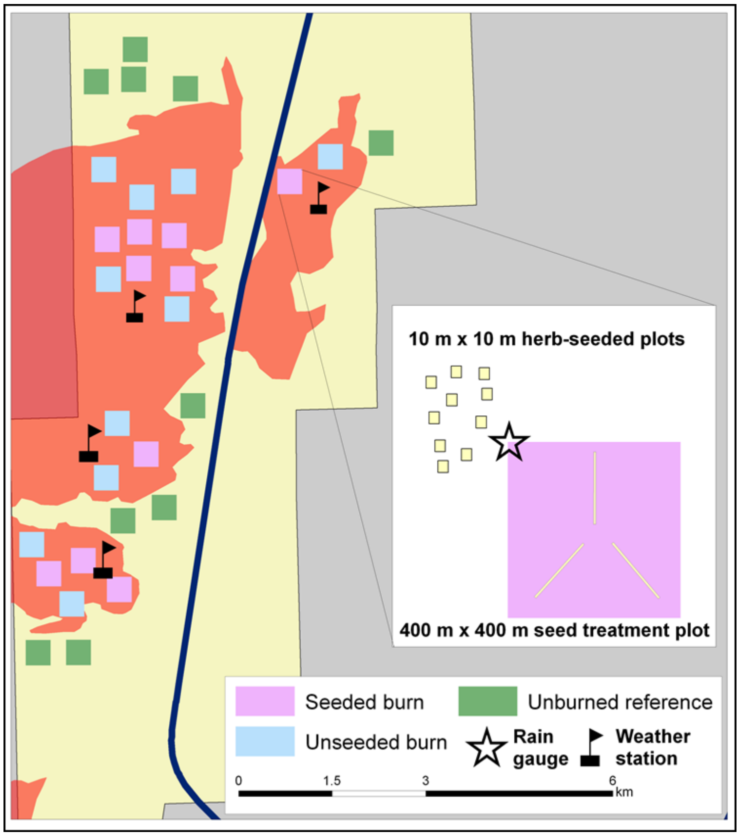

Figure 4. Close-up of Dry Middle, Dry Rock, and Garnet fire locations illustrating the monitoring design for evaluating seeding treatments (400 m × 400 m plots) and herbicide/seeding treatments (10 m × 10 m experimental plots) associated with the main seeding plots.

Monitoring methods

Monthly precipitation data were collected using manual rain gauges installed in 2005/2006 for the long-term monitoring sites at Dry Middle, Dry Rock, Garnet, Tramp, Halfway, and Meadow Valley fires. Manual gauges were installed at the northwest corner of each seeding monitoring plot (Fig. 4), and adjacent to the herbicide/seeding experimental plots, to record monthly precipitation (Tru-Chek® rain gauge, Edwards Manufacturing Company). Weather stations were also installed at a subset of the total monitoring plots to characterize precipitation patterns and temperatures.

Table 3. Properties of low-desert species mixes that were hand broadcast on seeding plots (Seeding Effectiveness) and experimental plots (Herbicide/Seeding Effectiveness) at the Dry Middle, Dry Rock, Garnet, Tramp, Meadow Valley, and Halfway fires in fall of 2005, 2006, and 2007. Seed mixes (PLS=pure live seed) were applied during two periods for the seeding effectiveness study because only perennial species were commercially available in 2005; additional seeding in 2006 included annual species important as tortoise forage. The herbicide/seeding effectiveness mix was broadcast one time only.

Seeding Effectiveness Seed Mix

	Species	Common name	Habit	PLS lbs/acre	PLS seeds/m^2
2005	*Sphaeralcea ambigua*	Desert globemallow	Per forb	1.4	175
	Baileya multiradiata	Desert marigold	Per forb	0.5	122
	Ambrosia dumosa	Bursage	Shrub	0.4	9
	Atriplex canescens	Four-wing saltbush	Shrub	0.7	8
2006	*Bouteloua aristoides*	Needle grama	Ann grass	3.7	381
	Sphaeralcea ambigua	Desert globemallow	Per forb	1.5	181
	Baileya multiradiata	Desert marigold	Per forb	0.6	146
	Vulpia octoflora	Six-week fescue	Ann grass	0.1	30
	Achnathereum hymenoides	Indian ricegrass	Per grass	0.7	27
	Atriplex canescens	Four-wing saltbush	Shrub	1.2	16
	Hymenoclea salsola	Cheesebush	Shrub	0.4	11
	Malacothrix glabrata	Desert dandelion	Ann forb	0.02	10
	Camissonia claviformis	Brown-eyed primrose	Ann forb	0.03	4
	Larrea tridentata	Creosote bush	Shrub	0.2	4
	Eschscholtzia minutiflora	Pygmy goldenpoppy	Ann forb	0.02	2
	Ambrosia dumosa	Bursage	Shrub	0.1	2
	Encelia virginensis	Brittlebush	Shrub	0.02	< 1

Herbicide/Seeding Effectiveness Seed Mix

	Species	Common name	Habit	PLS lbs/acre	PLS seeds/m^2
2007	*Bouteloua aristoides*	Needle grama	Ann grass	39.8	4,068
	Sphaeralcea ambigua	Desert globemallow	Per forb	15.6	1,926
	Baileya multiradiata	Desert marigold	Per forb	6.0	1,559
	Vulpia octoflora	Six-week fescue	Ann grass	1.3	317
	Achnathereum hymenoides	Indian ricegrass	Per grass	7.2	288
	Atriplex canescens	Four-wing saltbush	Shrub	13.2	169
	Hymenoclea salsola	Cheesebush	Shrub	4.3	116
	Malacothrix glabrata	Desert dandelion	Ann forb	0.2	104
	Camissonia claviformis	Brown-eyed primrose	Ann forb	0.3	44
	Larrea tridentata	Creosote bush	Shrub	2.0	40
	Eschscholtzia minutiflora	Pygmy goldenpoppy	Ann forb	0.2	19
	Ambrosia dumosa	Bursage	Shrub	0.9	18
	Encelia virginensis	Brittlebush	Shrub	0.3	3

Biomass of native and non-native annual plants was measured at peak production during April-May of 2010 and 2011. Each year, 1 m^2 sampling quadrats were randomly placed within each 40-acre monitoring plot. We identified all species in each 1 m^2 quadrat and harvested annual plant shoot biomass within a nested 20 cm × 50 cm quadrat. The number of quadrats to be sampled per plot in each year was determined based on adequacy of sampling calculations. Samples were dried in a convection oven at 50 °C to a constant mass and weighed. Within each 10 m × 10 m herbicide/seeding treatment plot, ten 1 m × 1 m quadrats were regularly arranged to sample annual plants. Annual plant sampling was conducted as described for the seeding monitoring plots, except that no destructive harvesting occurred. Instead of clipping shoots within the nested 0.1 m^2 quadrats, biomass was estimated as volume of shoot tissue (i.e., total cover × average height for each quadrat). Actual biomass was estimated from this volume using biomass-volume relationships from quadrats harvested off-plot. The same two observers conducted all estimates to reduce measurement variability (r^2 >0.80).

Perennial canopy cover and density of seeded species were measured on three permanent 100-m line transects located within each 40-acre monitoring plot arranged in a configuration based on Herrick et al. (2005; Fig. 4). Perennial plant cover by species was measured as canopy intercept along the transect lines. Density of seeded species was measured within a 2 m × 100 m belt centered along each transect line (600 m^2 per plot), with individuals classified as either adult or seedling depending on size and reproductive maturity. Perennial plant measurements were conducted May-June each year following annual plant sampling. In the 10 m × 10 m herbicide/seeding treatment plots, we measured perennial plant cover as canopy intercept on six 10-m long transects oriented west to east. All perennial seedlings rooted within the plot were identified to species and counted over the entire herbicide plot (100% coverage).

We evaluated seeding effectiveness on soil seed bank by collecting at random five soil cores (each 10.5 cm × 10.5 cm × 2.5 cm deep = 275.625 cm^3) adjacent to the perennial transects in burned-unseeded and burned-seeded monitoring plots in October 2011 (N=16 monitoring plot pairs), 5-6 yrs after the initial 2005/06 seedings. The five cores from each plot were pooled (5 cores × 275.625 cm^3 = 1378.125 cm^3), and this entire sample was spread to a depth of 2 cm in multiple bulb pots. All pots were arranged at random on four benches in the College of Southern Nevada (CSN) greenhouse and watered using a series of wetting-drying-chemical treatments for a period of eight months (DeFalco et al. 2009). Samples were monitored daily, and seedlings were identified to species, counted and removed.

In October 2010, four replicate soil cores (each 10.5 cm × 10.5 cm × 2.5 cm deep = 275.625 cm^3) were collected from plots previously treated with Plateau™, Oust™, Journey™ or no herbicide within the Clark County fire locations (N=14 locations, each with eight plots representing the herbicide×seeding treatment combinations). The four replicate cores were pooled for a total sample volume of 1102.5 cm^3, and samples were prepared and treated as for the previously-described seed bank analysis at the CSN greenhouse. Samples were monitored daily, and seedlings were identified to species, counted and removed.

Data analyses

Data from multiple monitoring sites were analyzed using mixed models in SAS (SAS software v. 9.3, SAS Institute, 2002-2010, Cary, NC). Perennial plant responses were compared between seeded and unseeded treatments (SEED) in a random complete block ANOVA blocked by site, with year designated as a repeated measure (YR, 2006-2011), and October-April precipitation prior to census (PRECIP) as an environmental covariate. This repeated measures ANOVA for analyzing long-term experimental data uses environmental covariates to explain random effects of YR and YR × SEED while incorporating models for correlation structure

(Loughin 2006). Seeded plant density (plants/100 m^2) was analyzed using the GLIMMIX procedure with a lognormal distribution and random residuals designated for overdispersion. Canopy cover (%), species richness, Shannon diversity and annual plant biomass (g/0.10 m^2) were analyzed using the MIXED procedure in SAS after arcsine-square root and $\log_{10}$ transformations, respectively. Perennial responses measured for evaluating herbicide (HERB) and seeding (SEED) treatments were similarly analyzed in a random complete block ANOVA blocked by site, YR as a repeated measure (2008-2012), and PRECIP as a covariate. After comparing models with different correlation structures, we selected the best structure based on the lowest AIC value. Comparisons of lsmeans of treatment effects were tested and back-transformed for illustration.

Because we monitored plant densities across multiple years and multiple sites in unseeded and seeded areas, we explored the effects of abiotic and biotic factors in explaining recruitment (yearly net increases in density) and mortality (yearly net decreases in density). We used an information-theoretic approach by first developing a set of linear regression models (explanations) that combined rainfall, temperature, competition and facilitation variables (Table 4). For each regression, we used Akaike's Information Criterion corrected for small sample size (AICc), which is favored over traditional methods such as stepwise, backward or forward selection because it considers existing knowledge about the biology or ecology of the system (Burnham and Anderson 2002). The lowest AICc value denotes the "best" model, or most plausible explanation of recruitment or mortality patterns, out of the set of models that were considered. The models were ranked by rescaling AICc values so that the model with the minimum AICc had a value of 0. The difference calculated from this best model, ΔAIC, allows comparisons with the remaining models: ΔAIC < 2 suggest substantial support for the model, and ΔAIC between 4 and 7 suggest considerably less support (Burnham and Anderson 2002).

Table 4. Candidate models tested to understand the influence of continuous abiotic (rain, temperature) and biotic variables (competition, facilitation) on perennial plant recruitment and mortality of seeded species. Twenty models each were tested with and without seeding (SEED) as a discrete variable. Recruitment and mortality were $\log_{10}$-transformed before analysis.

Models for Recruitment (Recr)	Models for Mortality (Mort)	Rationale for Recr/Mort (expected + or – relationship)
1.Intercept only	1.Intercept only	No relationship exists
2.SEED	2.SEED	Reduce seed limitation (+)/increase competition (-)
3.Oct-Mar rain (ODmm)	3.Oct-Mar rain (ODmm)	Stimulate germination (+)/reduce water stress (+)
4.Oct-Mar mean T_{min} (ODT_{min})	4.Apr-Jun mean T_{max} (AJT_{max})	Stimulate germination (+)/stress seedlings (-)
5.ODmm + ODT_{min}	5.ODmm + AJT_{min}	Stimulate germination (+)/stress seedlings (-,-)
6.Annual biomass (BMS)	6.Annual biomass (BMS)	Competition at emergence(-)/competition at development(-)
7.Live canopy cover (COV)	7.Live canopy cover (COV)	Facilitation (+/-) or competition(-/+)
8.Plant density (DENS)	8.Plant density (DENS)	Competition (-/+)
*9.NMS1	*9.NMS1	Combination of ODmm, AJT_{max}, COV, DENS
*10.NMS2	*10.NMS2	Combination of ODT_{min} and BMS
11.NMS3	11.NMS3	Dead cover releases nutrients (+)/(-)
12-20.SEED added to #3-11	12-20.SEED added to #3-11	Rationale

*Linear combinations of environmental variables were developed among sites using non-metric multidimensional scaling (McCune and Grace 2002) which resulted in the first axis correlated with ODmm, AJTmas, COV, and DENS; the second axis was correlated with ODT_{min} and BMS. A third axis NMS3, was correlated with dead canopy cover. These variables were included to determine if unique combinations of variables performed better at explaining recruitment and mortality than the separate variables.

Seed bank data were compared between seeding and herbicide treatments (SEED and/or HERB) using the GLMMIX procedure in SAS (SAS software v. 9.3, SAS Institute, 2002-2010, Cary, NC). For evaluating seeding effectiveness, densities of seeds in the soil seed bank were compared using site as a random blocking factor (BLK) and seeding (SEED) treatment as a fixed effect (dist=Poisson, and overdispersion accounted for in model with random residual). For evaluating herbicide/seeding effectiveness, herbicide (HERB) was also added to the model as a fixed effect. Significant main effects and interactions are reported. Least square means comparisons were tested, and back-transformed lsmeans and 95% confidence intervals were calculated.

Part III. Contrasting survival of two outplanted Mojave Desert perennials

Study sites

The study was conducted in four burned areas within Clark County, Nevada. The Goodsprings (35°55′N, 115°26′W), Loop (36°09′N, 115°26′W), Fork (36°15′N, 114°13′W), and Tramp (36°18′N, 114°10′W) wildfires occurred in summer 2005. These four fire sites are in previously unburned blackbrush/mixed-shrub communities and vary in elevation and precipitation (Fig. 5). All sites have shallow soils with lithic or paralithic contact, a duripan, or a calcic or petrocalcic horizon within 100 cm of the surface.

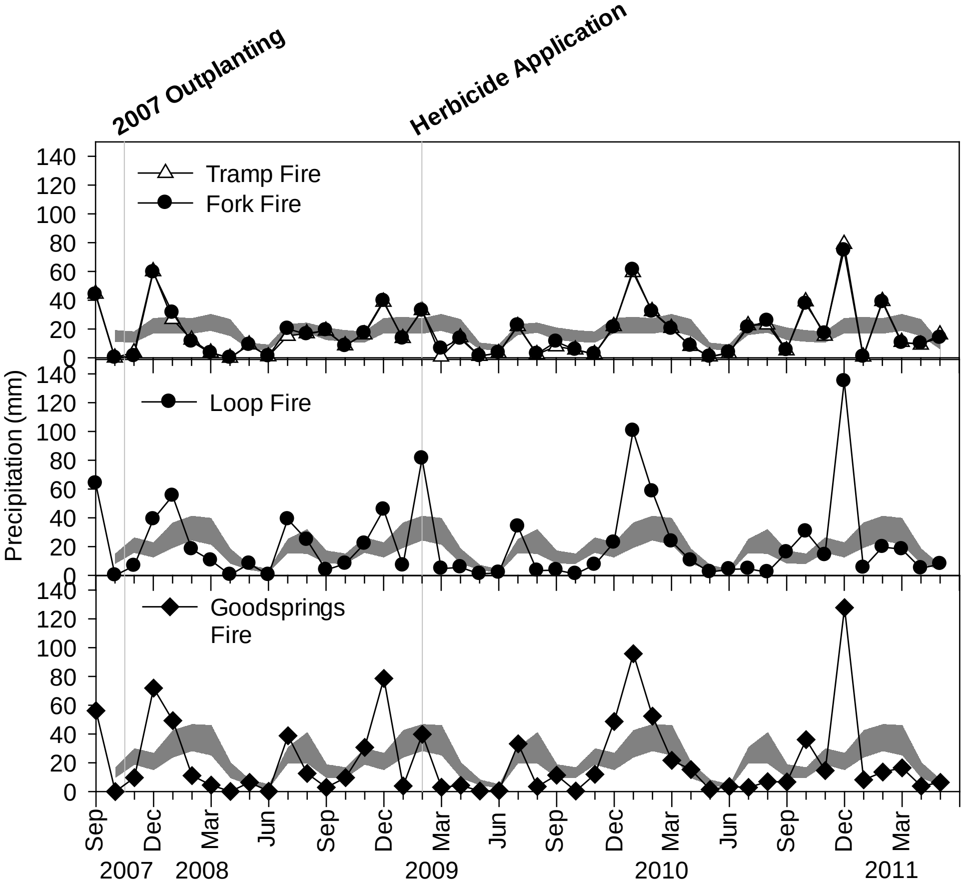

Figure 5. Monthly precipitation for lower-elevation Tramp (1030-1210 m a.s.l.) and Fork (1045-1275 m) sites (upper panel), mid-elevation Loop (1190-1305 m) site (middle panel), and higher-elevation Goodsprings (1280-1465 m) site (lower panel) in Clark County, Nevada. The 95% confidence interval around the 1945 – 2005 monthly averages (dark gray shading) were derived from the Fork, Loop, and Goodsprings sites, respectively. Monthly precipitation was derived from the PRISM Climate Mapping Program, Oregon State University (http://www.prismclimate.org).

Study species

The study species were selected based on pre-burn vegetation at the study sites. Nevada jointfir (*Ephedra nevadensis* S. Watson) is a fast-growing gymnosperm that can re-sprout after fires, and is found in both recently disturbed and undisturbed Mojave Desert plant communities from 0 m -1100 m elevation. Blackbrush (*Coleogyne ramosissima* Torr.), in contrast, is a slower-growing shrub in the Rosaceae that is found almost exclusively in undisturbed plant communities at 600 m -1600 m elevation.

Experimental design

We outplanted 3,101 blackbrush and 510 Nevada jointfir seedlings into fifteen 14 × 14–m plots at each of the Loop, Goodsprings, Tramp, and Fork fires between 29 October and 11 November 2007. Seedlings were grown in two local greenhouses: 4-month old and 1-year old blackbrush and 9-month old jointfir at one; 1-year old blackbrush at the second. Approximately equal numbers of seedlings representing the greenhouses, ages, and species were stratified among the four sites, four soil moisture amendment treatments, and two watering treatments (monthly watering or no watering). The soil amendment treatments were (i) Rain Bird Irrigation Supplement (Rain Bird IS, DriWater, Inc., Santa Rosa, CA), (ii) Zeba (Absorbent Technologies, Inc., Beaverton, OR), (iii) weed-free garden mulch, and (iv) no amendment. Plots were randomly assigned an herbicide treatment of RoundUp™ (Monsanto Company, St. Louis, MO), Journey™ (BASF Corporation, Research Triangle Park, NC), or no herbicide.

Outplanting procedure

We created a 30 × 30 × 30–cm hole for each seedling with a spacing of 1 plant per 2–4 m^2. For seedlings receiving Zeba, we mixed 15 g of granules into the bottom of the hole and 15 g of granules with soil from the hole. For Rain Bird IS, a 20 x 10–cm diameter PVC tube containing Rain Bird IS was installed adjacent to the seedling. Gel packs were replaced in late June 2008 and mid-March 2009. The mulch treatment entailed 1 L of weed-free organic garden mulch mixed with the native soil. All treatments, including controls, received 2 L water mixed into the native soil profile during planting. For the watering treatment, seedlings received a 0.5 L closed-top plastic reservoir with a 30-cm wick buried below rooting depth (adapted from Bainbridge 2007). Reservoirs were refilled monthly from December 2007 until October 2008. After planting, each seedling was given an additional 2 L of water, tagged with a unique number, and protected from herbivores with a 17 cm diameter × 46 cm tall plastic cone (Tree Sentry, Summit Environmental Group, Toledo, OH). We observed in early 2008 that air temperatures inside the cones were warmer than those outside, and replaced cones on half the number of surviving plants with closed-top 6-mm mesh cages. In mid-October 2009 we removed cones and cages from seedlings outplanted in 2007.

Herbicide application

Herbicides were applied to the pre-assigned plots in February 2009 with a 2-m buffer (18 × 18–m). Based on manufacture recommendations, RoundUp™ (active ingredient glyphosate) was applied as a 1% solution without surfactant, and Journey™ (active ingredients imazapic and glyphosate) was applied as a 2% solution with 0.25% methylated seed oil surfactant (C. Deuser 2009, Lake Mead NRA, pers. comm.). We applied herbicides using backpack sprayers, selectively targeting emerging non-native brome plants (*e.g., Bromus madritensis*) while avoiding seedlings. The outplanted seedlings were protected from herbicide contact by their protective cones or by plastic bags temporarily placed over the cages.

Environmental conditions

We monitored precipitation monthly using manual rain gauges at each plot and established two self-logging HOBO weather stations (Onset Computer Corporation, Pocasset, MA) at opposite ends of each site. The weather stations logged rainfall and temperature every 30 minutes. In February 2008, we placed HOBO Pro Series Temp/Ext Temp (H08–031–08) or StowAway Temperature (STEB16) loggers (Onset Computer Corporation) in 26 protective cones to record interior and exterior temperatures every 30 min until November 2008.

Seedling survival

We assessed seedlings monthly January–November 2008, and annually thereafter. Blackbrush were rated as alive (green leaves), dormant (dying leaves still attached, stems flexible), or dead (without leaves, stems grey and brittle). Nevada jointfir were rated as alive (green joints), dormant (joints red or brown but flexible), or dead (joints tan or grey and brittle). Mammalian herbivore damage was noted as trimmed twigs or cut stems and signs of tooth marks.

Statistical analyses

We used an information-theoretic approach to determine the most plausible model for seedling survival (Burnham and Anderson 2002). Potential explanations for seedling survival were developed into separate accelerated failure time models (Wei 1992) based on two guidelines: 1) we expected that processes influencing survival would differ between species; therefore, jointfir and blackbrush plants were analyzed separately, and 2) prior to model development, the most appropriate distribution type (*e.g.*, Weibull) for the failure model was selected by comparing Akaike's Information Criterion with correction for small sample size (AICc) values of intercept-only models for each species. A log-likelihood was calculated for each survival model using the LIFEREG procedure in SAS (SAS, Cary, NC). From the log-likelihood, we calculated Akaike's Information Criterion with correction for small sample size, AICc (Burnham and Anderson 2002). The lowest AICc value denotes the most plausible explanation out of the set of models that were considered. The differences from this best model, ΔAICc, were compared for remaining models: ΔAICc <2 suggests substantial support for the model; ΔAICc between 4 and 7, considerably less support; and ΔAICc >10, essentially no support (Burnham and Anderson 2002).

For each species, we developed a set of candidate models combining variables representing seedling age and management actions (*e.g.*, soil amendment, watering). For jointfir, analysis of survival after herbicide application included only those plants alive at the time of herbicide application (442 plants). All main effects models for blackbrush that included herbicide treatment performed worse than an intercept only model (ΔAICc >5); therefore, herbicide treatment was eliminated from the models, and all plants alive at the time of cage placement were included in the analysis (2513 plants). Multicollinearity did not occur among variables based on Pearson's |r| <0.75 and variation inflation factors <10 (Neter et al. 1996). The importance of each attribute (a value ranging from 0 to 1 for least to most important) was derived by summing the Akaike weights (w_is) across all candidate models where the attribute occurred (Burnham and Anderson 2002).

Goal 3, Evaluation of Native Species

See Goal 2, Rehabilitation Treatment Assessment, Part II for a description of field and greenhouse methods.

Statistical analysis

Due to the different sampling efforts for the seed bank samples, we conducted separate analyses on the seed bank communities between seeding treatments (i.e., seeding study on 40-acre plots) and herbicide/seeding treatments (i.e., experimental treatments on 10 m ×10 m plots). Seed bank data were analyzed using the GLMMIX procedure in SAS (SAS software v. 9.3, SAS Institute, 2002-2010, Cary, NC). To evaluate seeding effectiveness, densities of seeds in the soil seed bank were compared with seeding (SEED) treatment as fixed effect (for non-normal data, we specified dist=negbin, and overdispersion accounted for in model with random residual). To evaluate herbicide/seeding effectiveness, herbicide (HERB) was also added to the model as a fixed effect. Significant main effects and interactions are reported. Least square means comparisons were tested, and back-transformed lsmeans and 95% confidence intervals were calculated.

Separate analyses of community composition were conducted for the seeding effectiveness and herbicide/seeding effectiveness studies. Seed densities were $\log_{10}+1$-transformed before analysis. Seed bank densities were represented by species or by genus when species could not be determined. Differences in community composition between unseeded, seeded, and unburned treatments were tested with the multiple response permutation procedure (MRPP) in PCOrd (McCune and Grace 2002). Similarly, seed densities were used to test differences in community compositions for the herbicide/seeding study using a two-factor (SEED and HERB effects and interaction) in PerMANOVA (McCune and Grace 2002). When significant, compositions of the different treatment levels were compared after a Bonferroni correction for multiple comparisons. When significant treatment effects occurred, differences in communities were illustrated using non-metric multidimensional scaling (NMS), and association of species with different treatment groups were identified using indicator species analysis (McCune and Grace 2002).

To evaluate the influence of our treatments on plant establishment, we explored the effects of abiotic and biotic factors in explaining recruitment (yearly net increases in density) and mortality (yearly net decreases in density) for each of the seeded species using an information-theoretic approach. We first developed a set of linear regression models (explanations) that combined rainfall, temperature, competition and facilitation variables (Table 5). For each regression, we used Akaike's Information Criterion corrected for small sample size (AICc), which is favored over traditional methods such as stepwise, backward or forward selection because it considers existing knowledge about the biology or ecology of the system (Burnham and Anderson 2002). The lowest AICc value denotes the "best" model, or most plausible explanation of recruitment or mortality patterns, out of the set of models that were considered. The models were ranked by rescaling AICc values so that the model with the minimum AICc had a value of 0. The difference calculated from this best model, ΔAIC, allows comparisons with the remaining models: $\Delta\text{AIC} < 2$ suggest substantial support for the model, and ΔAIC between 4 and 7 suggest considerably less support (Burnham and Anderson 2002).

Table 5. Candidate models tested to understand the influence of abiotic (rain, temperature) and biotic variables (competition, facilitation) on perennial plant recruitment and mortality for each seeded species. Twenty models each were tested with and without seeding (SEED) as a discrete variable. Recruitment and mortality were $\log_{10}$-transformed before analysis.

Models for Recruitment (Recr)	Models for Mortality (Mort)	Rationale for Recr/Mort (expected + or – relationship)
1.Intercept only	1.Intercept only	No relationship exists
2.SEED	2.SEED	Reduce seed limitation (+)/increase competition (-)
3.Oct-Mar rain (OMmm)	3.Oct-Mar rain (OMmm)	Stimulate germination (+)/reduce water stress (+)
4.Oct-Mar mean T_{min} (OMT_{min})	4.Apr-Jun mean T_{max} (AJT_{max})	Stimulate germination (+)/stress seedlings (-)
5.OMmm + OMT_{min}	5.OMmm + AJT_{min}	Stimulate germination (+)/stress seedlings (-,-)
6.Annual biomass (BMS)	6.Annual biomass (BMS)	Competition at emergence(-)/competition at development(-)
7.Live canopy cover (COV)	7.Live canopy cover (COV)	Facilitation (+/-) or competition(-/+)
8.Plant density (DENS)	8.Plant density (DENS)	Competition (-/+)
*9.NMS1	*9.NMS1	Combination of AJT_{max}, BMS, and OMmm
*10.NMS2	*10.NMS2	Combination of COV and DENS
11.NMS3	11.NMS3	OMT_{min}
12-20.SEED added to #3-11	12-20.SEED added to #3-11	Seeding will increase recruitment and mortality at high rate

*Linear combinations of environmental variables were developed among sites using non-metric multidimensional scaling (McCune and Grace 2002) which resulted in the first axis correlated with AJTmax, BMS, and OMmm; the second axis was correlated with COV and DENS. A third axis NMS3, was correlated with OMT_{min}. These variables were included to determine if unique combinations of variables performed better at explaining recruitment and mortality better than the separate variables.

Results

Goal 1, Biomass Prediction

Correspondence between weather station data and spatial environmental models

Modeled monthly T_{min}, T_{max}, and precipitation data from PRISM, and precipitation data from NOAA, corresponded well with actual readings from our weather stations. Monthly T_{max} and T_{min} across the more than five year time span were significantly and positively related to values derived from PRISM (Fig. 6). Overall, the PRISM model only underestimated average maximum temperature by 0.9 °C and overestimated average minimum temperature by approximately 2.0 °C at our study sites.

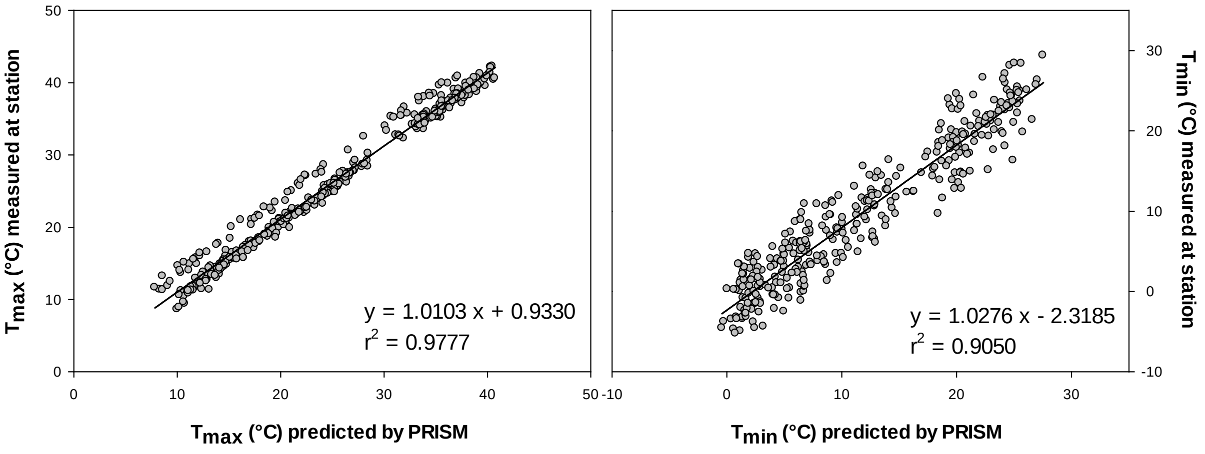

Figure 6. Relationship between actual station measurements and modeled maximum (left panel) and minimum (right panel) average monthly temperature from November 2007 – May 2013 at Southern Nevada Fire Complex sites. Modeled data were derived from the PRISM Climate Mapping Program, Oregon State University (http://www.prismclimate.org). Weather stations measured temperature every 30 min at 1 m above the soil surface at the Dry Middle, Dry Rock, Garnet, Halfway, Meadow Valley, Bunkerville, and Tramp fires.

Monthly precipitation values derived from both the NOAA and PRISM models were correlated with measured precipitation (Fig. 7). The correlation between measured monthly precipitation and data derived from the NOAA model was higher than the correlation with data derived from the PRISM model.

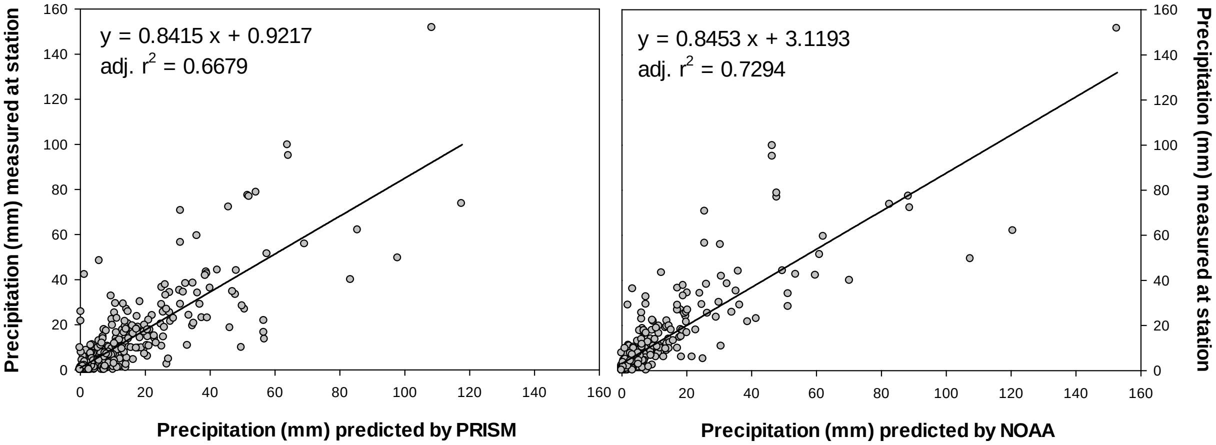

Figure 7. Relationship between measured and modeled monthly precipitation for the PRISM model (left panel) and the NOAA model (right panel) from November 2007 – December 2011 at Southern Nevada Fire Complex sites. Modeled data were derived from the PRISM Climate Mapping Program, Oregon State University (http://www.prismclimate.org) and from the NOAA ftp database (http://www.noaa.gov/), respectively. Precipitation values were summed by month from weather stations located at the Dry Middle, Dry Rock, Garnet, Halfway, Meadow Valley, Bunkerville, and Tramp fires.

Selection of best model for estimating annual plant production

One family of models, those related to the environmental and remote sensing data described in Van Linn et al. (2013), had the lowest AICc values, and therefore had the best correspondence with annual plant production (Table 6, adjusted $r^2 = 0.59$). Two models within this family had ΔAICc < 2 and differed only by the period that maximum and minimum temperatures were used (i.e., best model during October through March period; second best model during September through March). We used the best model with ΔAICc= 0 as described by the following equation:

$$\text{Biomass } (g/0.1\ m^2) = -2.2266+0.0009*\text{Elev}+3.181*\text{EVI}+0.0054*\text{Aspect}+0.00045*T_{min(Oct\text{-}Mar)}+0.00067T_{max(Oct\text{-}Mar)}$$

Where elevation is in meters, T_{max} and T_{min} are in C°*100, Aspect is sine (arc degrees), and peak productivity EVI ranges from -1.0 to +1.0 (variation inflation factors for all variables < 10, adj r^2=0.5932, df=96).

Table 6. Best models relating environmental attributes to annual plant biomass. Most plausible models (ΔAIC$_c$ <10), and intercept-only model for comparison, are presented. See Deliverable 31 for explanation of all candidate models included in model selection.

Model_Name	AICc	ΔAICc	w_i	Cum w_i	K	Adj-r2
Elev+EVI+Aspect+Tmin$_{\text{(Oct-March)+}}$Tmax$_{\text{(Oct-March)}}$	-144.92	0.00	0.45	0.45	7	0.58
Elev+EVI+Aspect+Tmin$_{\text{(Sep-March)+}}$Tmax$_{\text{(Sep-March)}}$	-144.38	0.54	0.34	0.79	7	0.59
Elev+EVI+Aspect+Tmin$_{\text{(Sep-Dec)+}}$Tmax$_{\text{(Sep-Dec)}}$	-142.13	2.80	0.11	0.90	7	0.58
Elev+EVI+Aspect+T$_{\text{min(Oct-Dec)+}}$T$_{\text{max(Oct-Dec)}}$	-140.95	3.97	0.06	0.96	7	0.58
Elev+NDVI+Aspect+T$_{\text{min(Oct-March)+}}$Tmax$_{\text{(Oct-March)}}$	-138.72	6.20	0.02	0.98	7	0.54
Elev+NDVI+Aspect+T$_{\text{min(Sep-March)+}}T_{\text{max(Sep-March)}}$	-137.99	6.94	0.01	0.99	7	0.56
Elev+NDVI+Aspect+T$_{\text{min(Sep-Dec)+}}$T$_{\text{max(Sep-Dec)}}$	-136.50	8.43	0.01	1.00	7	0.54
Intercept only	-59.06	85.86	0.00	1.00	2	

Map of "fine fuels" derived from best annual plant biomass model

The areas that indicate high average fuel production (Fig. 8, left panel; dark green) and the lowest variation (Fig. 8, right panel; light red) during the past 12 years include: south and west of Searchlight; west of Las Vegas and north and south of Hwy 160; and the west-central portion of the Gold Butte region. Lands in tortoise habitat with Clark County ownership generally have low to medium fuel production, and medium to high variability. One exception is the parcel at the extreme south of Clark County that had high average fuel production; however, high variability implies that high production is infrequent, and interannual monitoring may be necessary to identify years that require fuels management.

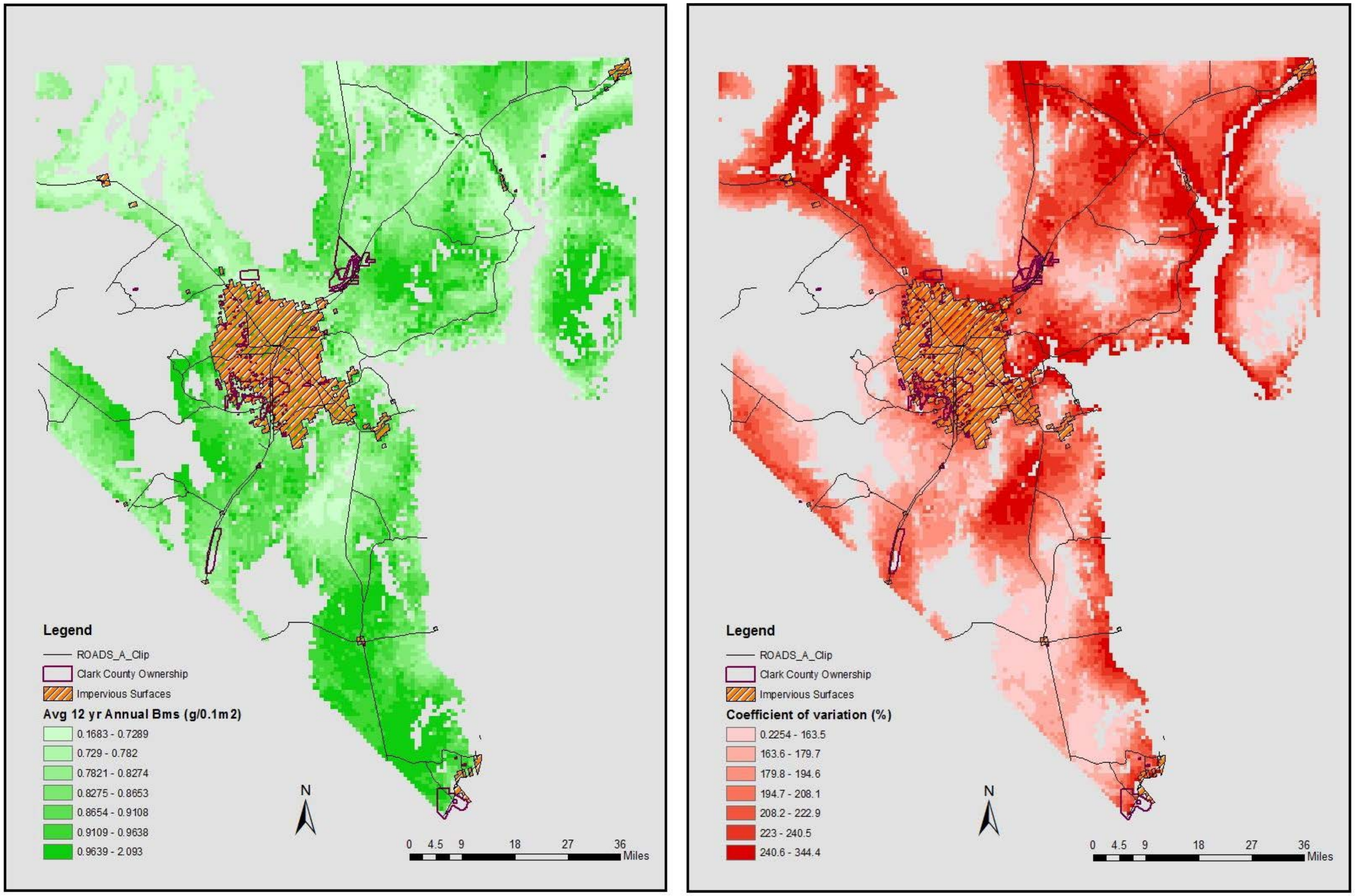

Figure 8. "Fine fuel" abundance depicted by the average annual plant biomass (left) and variability depicted by coefficient of variation (CV, right) for the period 2001 through 2012, based on the best model estimate (see equation in text). Areas with high annual plant biomass (deeper green) and lower CV (lighter red) imply areas of consistently high fine fuels for guiding prioritization of fuels management.

Goal 2, Rehabilitation Treatment Assessment

Part I. Monitoring seeding effectiveness for augmenting native species establishment on the Goodsprings and Bonnie Springs fires, Red Rock Canyon National Conservation Area

Sphaeralcea ambigua was by far the most abundant of seeded species across all plots at the Goodsprings Fire. However, there was no difference in the abundance of *S. ambigua* between seeded and control sites ($F_{1,8=}$0.01, P=0.99). *S. ambigua* was also the only seeded species with observed seedlings in most plots (Table 7). Due to the majority of datapoints being zeroes for all other species, ANOVAs are inappropriate to use for statistical interpretation. However, summary data indicate that there may be trends toward increased abundance of *Achnatherum speciosa*, *Atriplex canescens*, and *Ephedra nevadensis* in seeded plots. *A. canescens* seedlings were observed only in seeded plots in 2011. Overall density of seedlings of the target species except *S. ambigua* was low, generally below 50 seedlings per acre.

Table 7. Seedling density/acre (number of individuals per acre scaled from 100 m^2 plots; top) and frequency (percent of plots with at least one seedling found; bottom) across all sampled plots seeded and control (unseeded burned) plots within the Goodsprings Fire.

	density/acre			
	2010		2011	
Species	Control	Seeded	Control	Seeded
Achnatherum speciosa	4 ± 3	36 ± 15	2 ± 2	34 ± 14
Atriplex canescens	2 ± 2	6 ± 3	0 ± 0	247 ± 243
Coleogyne ramosissima	6 ± 4	6 ± 4	4 ± 4	30 ± 24
Ephedra nevadensis	14 ± 9	38 ± 21	8 ± 4	38 ± 15
Eriogonum fasciculatum	0 ± 0	8 ± 6	2 ± 2	63 ± 47
Krascheninnikovia lanata	2 ± 2	32 ± 32	0 ± 0	2 ± 2
Purshia tridentata	0 ± 0	4 ± 4	2 ± 2	2 ± 2
Sphaeralcea ambigua	1696 ± 712	1307 ± 423	684 ± 170	1070 ± 206
	Seedling frequency (% occurrence across all plots)			
	Control	Seeding	Control	Seeding
Achnatherum speciosa	10	35	5	35
Atriplex canescens	5	15	0	15
Coleogyne ramosissima	10	10	5	15
Ephedra nevadensis	15	25	20	35
Eriogonum fasciculatum	0	10	5	20
Krascheninnikovia lanata	5	5	0	5
Purshia tridentata	0	5	5	5
Sphaeralcea ambigua	85	95	80	95

For the Bonnie Springs Fire, there were no differences among abundances of seeded species in seeded plots relative to unseeded control plots (Table 8, $P > 0.05$ for all tests). *Sphaeralcea ambigua* was the most abundant seeded species observed in the plots, with widely ranging abundances ranging from an estimated 1,118 to 2,365 seedlings/acre, followed by *Baileya multiradiata*. Seedlings from other seeded species were rare within the site. There appears to be a trend toward greater density of *B. multiradiata* in seeded than unseeded plots, but this pattern is not statistically significant.

Table 8. Seedling density (number of individuals per acre, mean ± 1 SE.; top) and frequency (percent of plots with at least one seedling found; bottom) across all sampled plots seeded and control (unseeded burned) plots within the Bonnie Springs Fire.

	density/acre			
	2010		2011	
Species	Control	Seeded	Control	Seeded
Aristida purpurea	41 ± 28	16 ± 12	41 ± 30	36 ± 29
Baileya multiradiata	409 ± 182	1344 ± 807	437 ± 156	1393 ± 682
Hymenoclea salsola	24 ± 14	24 ± 20	77 ± 44	4 ± 4
Sphaeralcea ambigua	2365 ± 625	1721 ± 407	1632 ± 345	1118 ± 215
Salvia columbridae	24 ± 17	16 ± 13	332 ± 332	0
	Seedling frequency (% occurrence across all plots)			
	2010		2011	
	Control	Seeded	Control	Seeded
Aristida purpurea	20	30	40	40
Baileya multiradiata	90	90	100	100
Hymenoclea salsola	30	30	30	20
Sphaeralcea ambigua	100	100	100	100
Salvia columbariae	20	30	40	40

No seeded species were observed in the greenhouse grown seed bank samples for either fire site (Table 9). The seed bank was primarily composed of exotic annual species *Bromus rubens* and *Erodium cicutarium*. Seed bank communities did not differ between seeded and control plots (MRPP: $P > 0.05$). At the Bonnie Springs site, there appears to be a trend toward more *Bromus rubens* in control plots, but differences are only marginally significant (t-test: $P=0.07$).

Table 9. Mean (± 1 SE) individuals emerged (individuals per m^2 at 2 cm soil depth) from seed bank samples from seeded and control (unseeded but burned) areas of the Goodsprings and Bonnie Springs fires.

	Goodsprings Fire		**Bonnie Springs Fire**	
Species	Control	Seeded	Control	Seeded
Amaranth sp.	5 ± 5	0	0	0
Astragalus nuttallianus	0	5 ± 5	0	0
Bromus rubens	265 ± 104	220 ± 66	1440 ± 557	360 ± 91
Chaemasyce albomarginata	5 ± 5	0	10 ± 10	10 ± 10
Chaenactis stevioides	0	0	0	10 ± 10
Cryptantha nevadensis	0	5 ± 5	0	10 ± 10
Draba cuneata	10 ± 7	15 ± 8	0	0
Encelia virginensis	0	5 ± 5	0	0
Eriogonum deflexum	0	5 ± 5	0	0
Erodicum cicutarium	285 ± 70	430 ± 98	380 ± 155	210 ± 31
Lepidium lasiocarpum	5 ± 5	20 ± 14	50 ± 22	20 ± 20
Pectocarya recurvata	10 ± 10	0	0	0
Phacelia fremontii	0	0	30 ± 30	0
Plantago ovata	15 ± 11	10 ± 7	30 ± 21	10 ± 10
Schismus arabica	30 ± 21	15 ± 8	0	0
Sphaeralcea ambigua	0	5 ± 5	0	0
Unknown	10 ± 7	45 ± 18	30 ± 21	30 ± 30

There was no difference in plant community composition between seeded and control plots for either the Goodsprings or Bonnie Springs fires (Goodsprings, MRPP $P>0.05$, $A<0.2$ for both 2010 and 2011; Bonnie Springs, MRPP $P>0.05$). However, there were some differences in community composition across sites within the Goodsprings Fire (Fig. 9). In 2010, composition among sites varied slightly with sites 3 and 4 generally had a greater abundance of *Erodium cicutarium*, while sites 5 and 6 had a greater abundance of *Bromus rubens*, *Gutierrezia sarothrae*, and *Prunus fasciculata* (MRPP: $P<0.0001$, A=0.08). In 2011, patterns of differences among sites was similar, but the species driving the patterns varied with sites 3 and 4 still being associated with greater *E. cicutarium*, and also *Schismus*, site 2 had greater association with *Bromus rubens*, and sites 5 and 6 were more variable (MRPP: $P<0.0001$, A=0.10). There is no indication that seeding had an effect on overall plant community composition across sites. Additionally, there were no differences in richness (S) or diversity (*H'*) between seeded and control sites in either 2010 (S = 20.3 ± 1.0, *H'* = 1.88 ± 0.10; mean ± 1 SE) or 2011 (S = 17.2 ± 0.8, *H'* = 1.69 ± 0.08; Seeding: $P>0.05$).

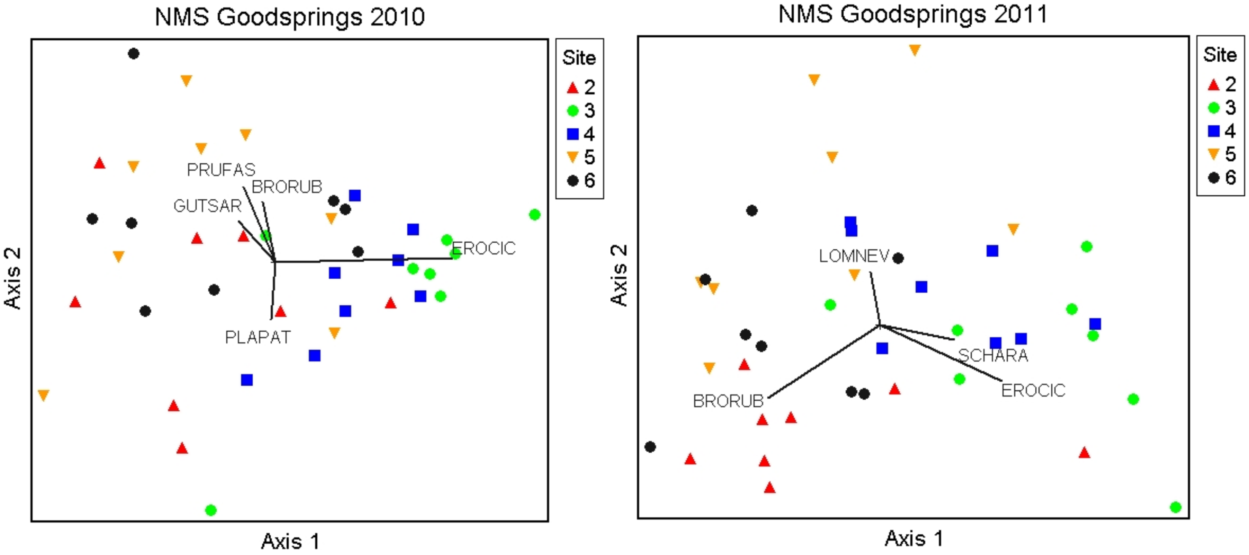

Figure 9. NMS (non-metric multidimensional scaling) ordinations with overlayed species joint plots for the Goodsprings seeding plots for 2010 (left) and 2011 (right). BRORUB = *Bromus rubens*, EROCIC = *Erodium cicutarium*, GUTSAR = *Gutierrezia sarothrae*, LOMNEV = *Lomatium nevadense*, PLAPAT = *Plantago patagonica*, PRUFAS = *Prunus fasciculata*, SCHARA = *Schismus arabicus.*

Part II. Evaluation of hand seeding and herbicide application for rehabilitation of Mojave Desert tortoise habitat

Seeding effectiveness

Despite high seeding rates (Table 3), seeding did not reduce the biomass of exotic annual grasses and forbs (SEED; $F_{1,16.9}=0.28$, $P=0.60$) or increase the biomass of native annual grasses and forbs (SEED; $F_{1,14.6}=0.04$, $P=0.85$). Biomass did vary across years for exotics (YEAR; $F_{4,31.8}=26.05$, $P<0.01$) and natives (YEAR; $F_{4,28.4}=33.47$, $P<0.01$), which was positively related to abundance of winter/spring precipitation.

Hand-seeding during the winters of 2005/2006 increased plant densities of seeded perennial species in treated areas compared with controls beginning in 2009 and persisting through 2012 (YEAR×TRT, $F_{6,41.1}=2.59$, $P=0.03$; Fig. 10). This pattern was largely attributed to *Baileya multiradiata*, which by 2012 had increased to 38.7 plants/100 m^2 in seeded areas but only 1.9 plants/100 m^2 in controls (YEAR×TRT, $F_{6,43.69}=6.75$, $P<0.01$). Densities were not statistically different between seeding treatments for *Sphaeralcea ambigua* but did increase through time to an average 9.3 plants/100 m^2 by 2012 (YEAR, $F_{6,40.59}$=9.26, *P*<0.01) as did *Ambrosia dumosa* (YEAR; $F_{6,40.19}$=23.82, *P*<0.01) with 2.1 plants/100 m^2 by 2012. All other perennial species were too rare for individual statistical analyses.

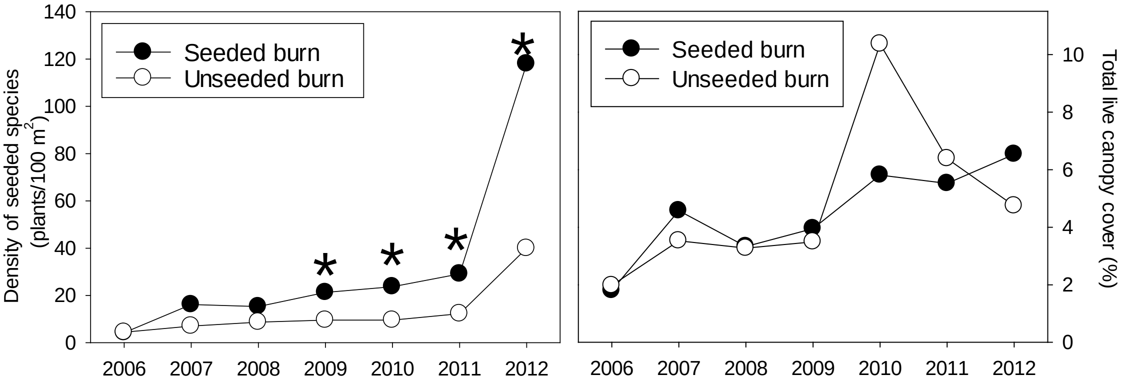

Figure 10. Least square means for plant establishment of seeded perennial species (left panel) and perennial canopy cover (right panel) in burned areas. Asterisks indicate years that had significantly greater densities in seeded versus unseeded burned areas.

During the seven years of study, total live canopy cover of perennials increased through time (YEAR; $F_{6,33.6}$=4.95, *P*<0.01), but did not differ between seeded and control treatments (Fig. 10). However, when seeded species were analyzed separately, seeding effects on cover became more pronounced over time for *Baileya multiradiata* (YEAR×TRT, $F_{6,33.8}$=4.67, *P*<0.01). Perennial species richness per plot also increased through time (YEAR; $F_{4,134.9}$=17.46, *P*<0.01) from 7.4 species in 2008 to 10.7, 12.0, 12.2, and 12.8 species in 2009, 2010, 2011, and 2012, respectively, but was not different between unseeded (10.7 species) and seeded burned areas (10.9 species). Similarly, species diversity was not different between control and seeded burned areas but increased in later years compared with 2008 (YEAR; $F_{4,135}$=3.69, *P*<0.01).

Perennial plant recruitment and mortality were related to presence of neighboring plants. Recruitment (i.e., the number of plants establishing per year) was positively correlated with the previous year's live perennial plant cover (Table 10, Fig. 11, left). Mortality (i.e., the number of plants decreasing per year) was positively related to the previous year's plant density (Table 10, Fig. 11, right). The seeding treatments had higher recruitment and mortality than the controls for a given cover or density, respectively (Fig. 11).

Table 10. Analysis of perennial plant recruitment and mortality in post-fire treatment plots in southern Nevada from 2006 to 2011. Only those models with some level of support (ΔAICc < 7) are presented. Following the most explanatory model (ΔAICc=0), there were no additional plausible models for mortality. Akaike weights (w_i) are included for comparing the relative importance of the variables.

Recruitment				Mortality			
*Model**	AICc	ΔAICc	w_i	*Model**	AICc	ΔAICc	w_i
SEED+COV	170.9	0.0	0.4846	SEED+DENS	92.8	0.0	0.9765
SEED+DENS	171.0	0.1	0.4610				
SEED+NMS1	175.4	4.5	0.0511				

* SEED, seeded or not seeded; COV, previous year's perennial canopy cover, square root-arcsine transformed; DENS, previous year's perennial plant density, $\log_{10}$-transformed; NMS1, first axis of non-metric multidimensional scaling, combination of win/spr precip, spring T_{max}, COV, and DENS.

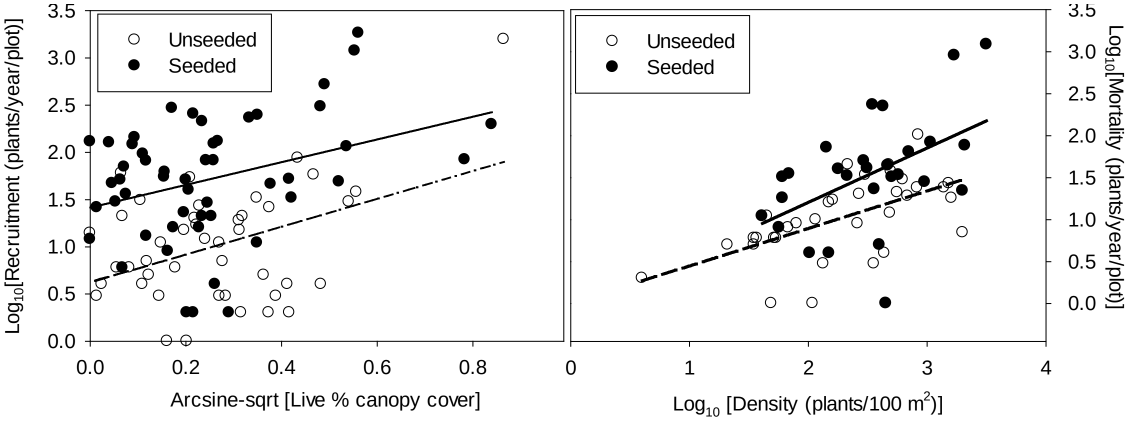

Figure 11. The best models from Table 10 depict early recruitment related to previous year's live canopy cover (left) and mortality related to previous year's perennial density (right). $Recruit_{unseeded}=0.6255+1.4749*arcsine\text{-}sqrt\text{-}cov$ ($r^2=0.18$); $Recruit_{seeded}=1.4160+1.2024*arcsine\text{-}sqrt\text{-}cov$ ($r^2=0.18$); $Mortal_{unseeded}=0.0031+0.4469*log10\text{-}density$ ($r^2=0.36$); $Mortal_{seeded}=-0.0885+0.6473*log10\text{-}density$ ($r^2=0.25$).

Soil seed bank: seeding effectiveness

Five years after broadcast seeding, soil seed banks had six times the seeded perennial forb density in the seeded areas compared to unseeded control (Table 11; SEED; $F_{1,17.65}=19.77$, $P<0.01$). These perennial forbs were namely *Sphaeralcea ambigua* and *Baileya multiradiata* (lsmeans for total seeded forbs; seeded = 66.4 seeds/m², unseeded = 13.08 seeds/m²). Seeded shrubs were also significantly different between treatments (SEED; $F_{1,12.94}=14.72$, $P<0.01$), although unexpectedly, seed densities in the controls were more than seven times greater compared with seeded areas. Seeding had no effect on any other plant functional group including native or exotic annual grasses or forbs, or non-seeded perennial forbs, grasses, or shrubs. Of the seeded annual species, only seedlings of *Vulpia octoflora* emerged in the soil seed bank, but no seeding effect was detected.

Table 11. Seeding effects on the densities of live seeds in the soil seed bank five years after application (expressed as number of seedlings that emerged from pots in the greenhouse, converted to seeds/m²). Least square means with 95% upper and lower limits are reported. Low densities of other shrubs (those not seeded) precluded statistical analysis. Different lower case letters denote a seeding effect within a functional group.

	Control (#/m²)		Seeded (#/m²)	
Functional group	Lsmean	CI	Lsmean	CI
Seeded perennial forbs	11 **a**	5--26	66 **b**	41--106
Other perennial forbs	23	13--41	18	10--34
Seeded shrub	22 **a**	11--45	3 **b**	1--10
Perennial grass	4	1--13	3	1--11
Exotic annual grass	10,093	5,913—17,225	11,511	6,764—19,589
Exotic annual forb	3,637	2,088—6,334	3,309	1,895—5,776
Native annual grass	2,376	1,439—3,921	2,618	1,603—4,276
Native annual forb	1,588	1,115—2,261	1,452	1,017—2,072

Herbicide/seeding effectiveness

The spot-application of RoundUp™ in early spring was ineffective at suppressing exotic annuals; however, the other herbicides were successful, and this suppression varied through time (HERB × TIME; $F_{8,197}=25.57$, $P<0.01$; Fig. 12). Oust™ suppressed exotics relative to controls for only one year. In contrast, the reduction of exotic annual biomass by Plateau™ and

Journey™ lasted for two years, although the later application of Journey™ caused the effects to lag one year behind Plateau™. Native species production was also unaffected by RoundUp™ but was influenced by the other three herbicides (HERB × TIME; $F_{8,161}$=16.15, $P<0.01$; Fig. 12). These three herbicides all reduced natives in 2009, but Oust™ increased natives in 2010, and Plateau™ increased natives in 2010 and 2011. The high rate of seeding for annual native species (Table 3) did not increase biomass of natives or decrease biomass of exotic species ($P>0.05$ for all SEED main effects and interactions).

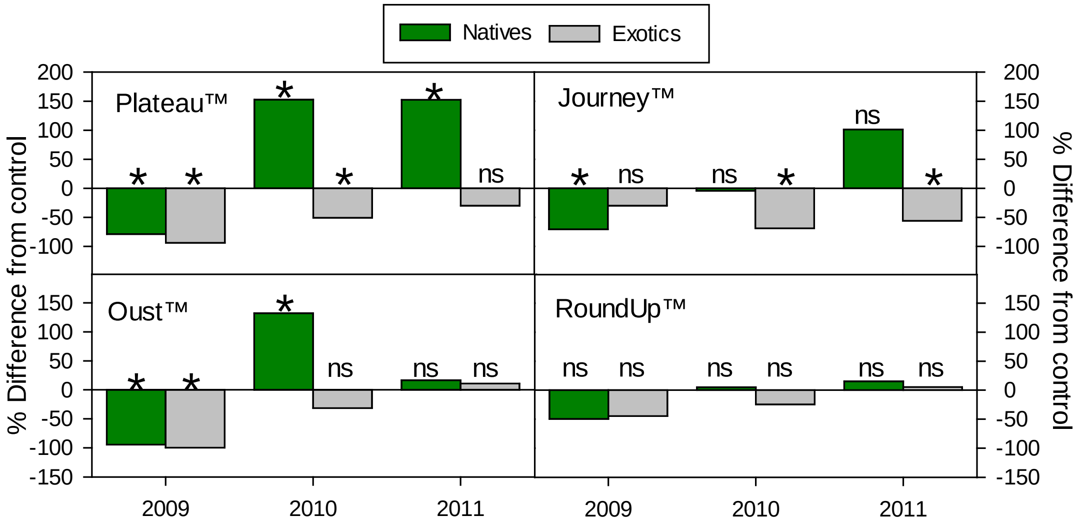

Figure 12. Herbicide application resulted in the reduction or augmentation of annual plant biomass relative to controls. Imazapic-containing herbicides Plateau™ and Journey™ were most effective at reducing exotic shoot biomass. The suppression of exotics increased native annuals at two (Oust™) to three years after application (Plateau™). Asterisks denote significant herbicide effect relative to control plots; "ns" = not significant. See text for details on methods and timing of seeding and herbicide applications.

In combination with herbicides, seeding enhanced densities of seeded perennials, and this pattern varied over time (SEED × HERB × YR; $F_{16,153.8}$=3.00, $P<0.01$; Fig. 13). By 2012, the seeding-herbicide treatment combinations had diverged such that seedling densities in the seeded Journey™ and Plateau™ treatments were greater than their unseeded counterparts, but there was no seeding effect for the Oust™ or control treatments. Of the total seeded species detected across all treatments, *Baileya multiradiata* comprised the greatest proportion of plants (81%) followed by *Sphaeralcea ambigua* (11%), *Larrea tridentata* (4%), *Ambrosia dumosa* (3%), *Encelia virginensis* (2%), and *Achnatherum hymenoides, Atriplex canescens,* and *Hymenoclea salsola* (each <1%).

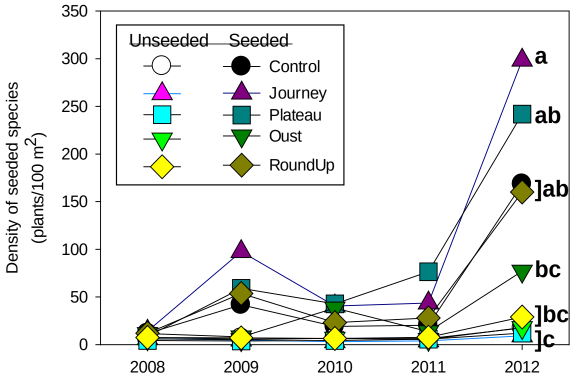

Figure 13. Least square means for plant establishment of seeded perennial species measured each May following the initial seeding (October 2007) and herbicide applications (Oust™ and Plateau™ in October 2008; RoundUp™ and Journey™ in February 2009). The SEED × HERB × YR interaction was significant (see text), and post-hoc comparisons of all treatment combinations were evaluated for 2012. Treatment-combination differences for this fifth year of monitoring are represented by different lowercase letters; although it is difficult to discern treatments, "]c" represents the unseeded Journey™ and Plateau™ treatments, and "]bc" represents unseeded Oust™ and control treatments.

Seed bank: Herbicide/seeding effectiveness

The impacts of herbicides on annual species persisted in the seed bank two years following herbicide application (Table 12). Seed densities of exotic annual grasses—*B. madritensis* spp. *rubens*, *B. tectorum*, and *S. barbatus*—remained lower across all herbicide treatments relative to control (HERB; $F_{3,90.44}=8.19$, $P<0.01$) with Oust™, Journey™, and Plateau™ reducing densities to 46%, 49%, and 57% of the control, respectively. Similarly, exotic annual forbs—*E. cicutarium*, *B. tournefortii*, *Salsola* sp., *Malcomia africana*, *Sisymbrium altissimum*—declined: Plateau™, 18%; Journey™, 29%; and Oust™, 42% of control densities (HERB; $F_{3,89.43}=16.87$, $P<0.01$). Native annual forbs were less sensitive to herbicides and were only reduced by Journey™ to 55% of the control densities (HERB; $F_{3,91.29}=6.09$, $P<0.01$). Native annual grasses—*V. octoflora* and a few *B. barbatus*—were reduced to 33% (Oust™) and 55% (Journey™) of the control densities, yet Plateau™ increased native grasses to almost three times the control densities (HERB; $F_{3,90.2}=22.51$, $P<0.01$). Independent of herbicide treatment, the addition of *V. octoflora* in the seed mix lead to a doubling of annual grass densities in the seed bank (SEED; $F_{1,90.2}=7.14$, $P<0.01$; Table 12). The seeded summer annual grass *B. aristoides* was not detected in the seed bank.

Densities of perennial seeds were lower than annuals and impacted less by herbicides (Table 12). An interactive herbicide and seeding effect was only evident for the seeded forbs (HERB × SEED; $F_{3,90.06}=4.37$, $P=0.01$), with no herbicide effect for unseeded plots, but a reduction of seed densities by 63% and 93% for Journey™ and Oust™, respectively, in the seeded plots.

Table 12. Herbicide and seeding effects on the densities of live seeds in the soil seed bank (expressed as number of seedlings that emerged from pots in the greenhouse, converted to seeds/m^2). Least square means (95% lower—upper limits) for seeds/m^2 are reported. Densities of seeded perennial grass species were too low for statistical analysis. Seeded and non-seeded shrub species were combined because they were too rare to analyze separately. Different lower case letters denote herbicide or seeding effects within a functional group. Herbicide comparisons were made at each level of seeding for seeded forbs (i.e., herbicide levels compared within each seeding level only), which had the only significant SEED×HERB interaction. See text for further explanation of significant main effects and interactions.

	Control™	Journey™	Oust™	Plateau™	Unseeded	Seeded
Annuals:	Seeds/m^2				Seeds/m^2	
Exo grass	5,345**a**	2,595**b**	2,464**b**	3,050**b**	3,195	3,196
	(3,390—8,457)	(1,577—4,272)	(1,487—4,084)	(1,876—4,958)	(2039—5008)	(2,036—5,018)
Exo forb	1,541**a**	452**bd**	640**bc**	274**d**	721	484
	(684—3,471)	(187—1,088)	(274—1,494)	(108—698)	(319—1,633)	(210—1,118)
Nat grass	1,976**a**	850**b**	667**b**	5,408**c**	1,117**a**	2,205**b**
	(977—3,994)	(367—1,969)	(283—1,572)	(2,853—10,251)	(548—2,274)	(1,142—4,258)
Nat forb	855**ac**	474**b**	1,169**a**	698**bc**	861	668
	(586—1,247)	(303—742)	(821—1,666)	(468—1,040)	(617—1,200)	(47—944)
Perennials:						
Forb	4 (1—11)	6 (2—15)	3 (1—10)	2 (1—8)	4 (2—9)	4 (1—9)
Shrub	9 (5—20)	5 (2—14)	2 (1—12)	4 (2—12)	4 (2—9)	5 (2—11)
Seeded forb:						
Unseeded	9 (2—36)**a**	5 (1—31)**a**	23 (9—60)**a**	4 (0—30)**a**		
Seeded	81 (40—162)**a**	30 (13—73)**b**	6 (1—33)**b**	81 (40—162)**a**		

Part III. Contrasting survival of two outplanted Mojave Desert perennials

Precipitation

Tramp and Fork, the two sites located furthest south, received generally less precipitation than the other outplant sites, but precipitation patterns were similar among all sites (Fig. 5). Immediately following the 2007 outplanting, all sites received above-average precipitation in December and January. Subsequent months remained within or below the normal range until December 2008, when above-average precipitation returned. Precipitation patterns were similar in 2009, except that the heaviest winter rains were delayed one month until January 2010. All sites received above-average fall rain in October 2010, followed by above-average December precipitation.

Jointfir

Overall survival for jointfir was 61% after 43 months, and site and herbicide were the most influential factors (Table 13; w_is = 0.9997 and 0.7655, respectively, summed over all candidate models). Jointfir survival was lowest at the Loop site (Fig. 14A) and lower in plots treated with Journey™ than those treated with RoundUp™ or no herbicide (Fig. 14B). Supplemental watering, soil amendment, and herbivore protection were not well-supported as explanations for survival by the set of most plausible models.

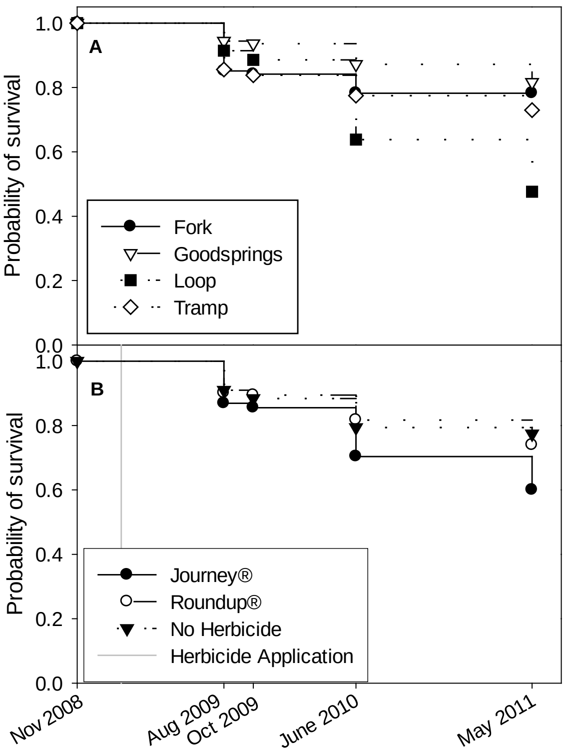

Figure 14. Survival functions for the best explanatory variables (highest weights in model with lowest AICc value) for Nevada jointfir seedlings outplanted in 2007: (A) site and (B) herbicide treatment.

Blackbrush

Overall survival of blackbrush was 4% after 43 months, and protection, site, and source influenced survival (Table 13; w_is = 1.0000, 1.0000, and 0.8974, respectively). Cones elevated air temperature within by approximately 4.4°C (Fig. 15A, inset). Consequently, seedlings with cones initially had higher survival through winter months, but by 24 months, survival was higher for seedlings protected by mesh cages (Fig. 15A). The influence of site on survival was mixed: Fork and Tramp had lower survival over the first year, while Loop and Goodsprings had decreased survival over the second year (Fig. 15B). Source also influenced survival for blackbrush, with plants from one greenhouse experiencing higher survival for more than 3.5 years after planting (Fig. 15C). Supplemental watering, soil amendment, and seedling age were not well-supported as explanations for survival by the set of most plausible models.

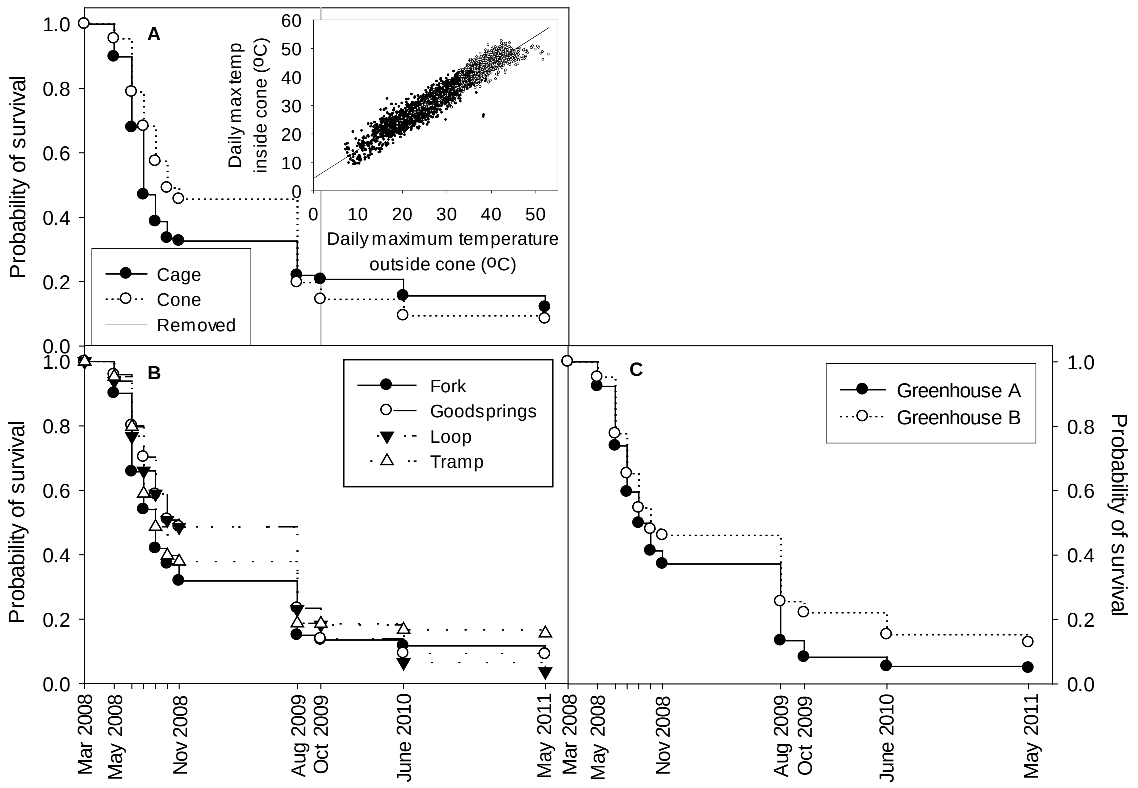

Figure 15. Survival functions for the best explanatory variables (highest weights in model with lowest AICc value) for blackbrush seedlings outplanted in 2007: (A) type of seedling protection, (B) site, and (C) source of plant. Inset within Figure 15A is a graph of daily maximum temperature inside the protective plastic cones (T_{cone}) versus daily maximum temperature outside the cones ($T_{cone} = T_{out} + 4.4$ °C; $r^2=0.934$).

Table 13. Analysis of seedling survival for Nevada jointfir and blackbrush outplanted into post-fire experimental plots in southern Nevada in 2007. Models are ranked by ΔAICc. Only those models with some level of support (ΔAICc <7) are presented here. Akaike weights (w_i) are included for comparing the relative importance of the environmental and treatment attributes.

Nevada jointfir, after herbicide application

*Model**	*k*	*AICc*	*ΔAICc*	w_i
Site, Herbicide	9	498.2	0.0	0.5543
Site	6	500.5	2.3	0.1727
Site, Herbicide, SA	13	501.3	3.2	0.1138
Site, Herbicide, Water	11	502.1	4.0	0.0767
Site, SA	10	503.9	5.7	0.0322
Site, Water	8	504.5	6.3	0.0234

Blackbrush, after cage placement

*Model**	*k*	*AICc*	*ΔAICc*	w_i
Site, Protect, Source	11	3793.1	0.0	0.7726
Site, Protect, Source, Water	13	3797.1	4.0	0.1048
Site, Protect, Age	11	3797.6	4.5	0.0828

* Plot and treatment attributes: Site, planting location of Fork, Goodsprings, Loop, or Tramp; Water, with or without water reservoir; SA, soil amendment comparison among none, mulch, Zeba, and Rain Bird IS treatments; Herbicide, comparison among none, RoundUp™, and Journey™ treatments; Protect, comparison between plastic cone or a wire cage; Source (blackbrush only), greenhouse where plants were grown before outplanting; Age (blackbrush only), comparison between 4 mo- or 12 mo-old plants.

Goal 3, Evaluation of Native Species

Seed densities for seeded species in surface soils

Five years after seeds were broadcast in the burned areas, *Baileya multiradiata* had ten-fold greater seed densities in the surface soils of the seeded treatment compared with controls, although overall densities were still lower than the seeding rate of 268 seeds/m^2 (SEED; $F_{1,30}$=7.84, *P*<0.01, Table 14). This perennial forb was the only species that had a significant increase in the surface soils of seeded areas.

Table 14. Seeding effects on the densities of live seeds in the soil surface five years after seeding (expressed as number of seedlings that emerged in the greenhouse, and converted to area = m^2). Least square means with 95% upper and lower limits are reported (N=16 sites). Different lower case letters denote a seeding effect within species. The annuals *Bouteloua aristidoides*, *Malacothrix glabrata*, and *Eschscholtzia minutiflora* and the perennial *Atriplex canescens* were not detected in the seed bank.

	Control (#/m^2)		Seeded (#/m^2)	
Species	Lsmean	CI (lower—upper)	Lsmean	CI (lower—upper)
Vulpia octoflora	2,696	1,634—4,441	2,972	1,803—4,895
Baileya multiradiata	6 **a**	2—20	62 **b**	18—213
Sphaeralcea ambigua	9	4—22	12	6—27
Encelia virginensis	14	7—25	0	0—0
Ambrosia dumosa	12	3—51	3	1—14
Camissonia claviformis	3	0—26	1	0—9
Larrea tridentata	2	0—17	1	0—9
Hymenoclea salsola	1	0—5	0	0—0
Achnatherum hymenoides	0	0—0	1	0—5

Likewise, two years following seeding and herbicide application, *B. multiradiata* had four-fold greater densities averaged over herbicide treatment (SEED; $F_{1,104}$=6.06, *P*=0.02), with 20 seeds/m^2 (CI=2-11 seeds/m^2) in seeded areas and 5 seeds/m^2 (CI=9-46 seeds/m^2) in unseeded areas (Table 15). Again these densities were lower than the seeding rate of 1,559 seeds/m^2. A significant SEED×HERB interaction for this species ($F_{3,104}$=6.08, *P*<0.01) indicates that herbicide had no effect on *B. multiradiata* seed densities in unseeded plots, while in the seeded areas there was a reduction of density for the Oust™ treatment compared with other herbicide treatments (Table 15). Similarly, seed densities of the annual grass *Vulpia octoflora* doubled in seeded areas; and while densities of this native decreased compared with the control in the Journey™ and Oust™ treatments, they were augmented in the Plateau™ treatment (Table 15). The perennial forb *Sphaeralcea ambigua* had marginally greater seed densities in the seeded areas compared with controls (SEED; $F_{1,104}$=3.01, *P*=0.09).

Community composition of the seed banks

Seeding did not change the compositions of the seed bank five years after seeding (MRPP, A=0.013, *p* =0.10). Similarly, seeding had no effect two years after seed and herbicide application (SEED, PerMANOVA, $F_{1,104}$=2.4, *P*=0.21) but compositions were different among the herbicide treatments (HERB, PerMANOVA, $F_{3,104}$=2.4, *P*<0.01). This herbicide effect reflected different communities for the Journey™ plots compared with the control (t=1.76, *P*<0.01, Fig. 16). While differences in seed bank communities along Axis 1 were likely due to differences we observed in soil texture among sites, differences along Axis 2 reflected the impact of Journey™ in reducing the native *Cryptantha* species and non-native *Bromus* species.

Table 15. Herbicide and seeding effects on the densities of live seeds in the seed bank (expressed as the number of seedlings that emerged from pots in the greenhouse and converted to seeds/m^2). Least square means (95% lower—upper limits) for seeds/m^2 are reported. Different lower case letters denote herbicide or seeding effects for *Vulpia octoflora*. Herbicide comparisons were made at each level of seeding for *Baileya multiradiata* (i.e, herbicide levels compared within each seed treatment), which had the only significant SEED×HERB interaction (see text). *Ambrosia dumosa*, *Achnatherum hymenoides*, *Hymenoclea salsola*, *Encelia virginensis*, *Larrea tridentata* and the annuals *Camissonia claviformis*, *Malocothrix glabrata*, and *Eschscholtzia minutiflora* were virtually undetectable. *Vuloct*=*V. octoflora*, *Sphamb*=*S. ambigua*, and *Baimul*=*B. multiradiata*.

	Control™	Journey™	Oust™	Plateau™	Unseeded	Seeded
Species	Seeds/m^2				Seeds/m^2	
Vuloct	2,800**a**	2,595**b**	2,464**b**	3,050 **c**	1,583 **a**	3,125 **b**
	(1,637—4,791)	(704—2,060)	(553—1,619)	(4,481—13,114)	(1,083—2,314)	(2,137—4,568)
Sphamb	3	4	6	6	3	7
	(1—8)	(2—10)	(3—16)	(3—16)	(2—6)	(4—14)
Baimul	Control™	Journey™	Oust™	Plateau™		
Unseeded	8 (2—41) **a**	5 (1—25) **a**	23 (5—114) **a**	2 (0—8) **a**		
Seeded	100 (20—505) **a**	24 (6—147) **a**	2 (0—8) **b**	74 (18—456) **a**		

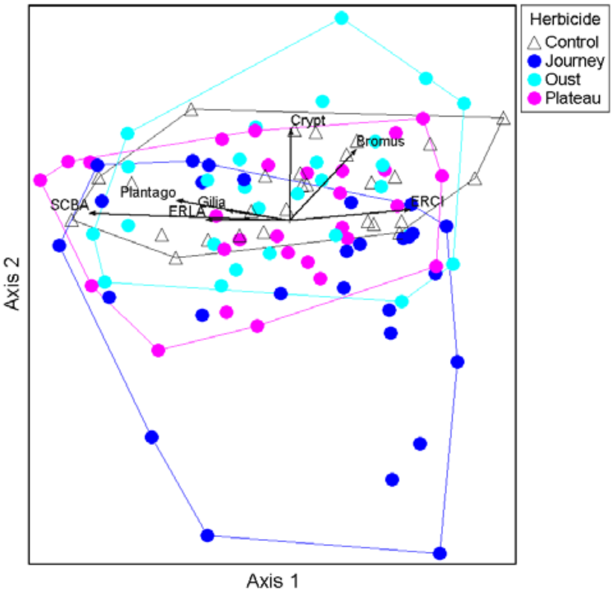

Figure 16. NMS ordination of seed bank communities among the herbicide treatments two years after application. Species include: *Schismus barbatus* (SCBA), *Plantago* species, *Eriophyllum lanosum* (ERLA), *Gilia* species, *Cryptantha* species, *Bromus* species, and *Erodium cicutarium* (ERCI).

Seedling recruitment and mortality for seeded species

Patterns in recruitment and mortality suggest that environment and biotic interactions shape the recruitment of seedlings from the seed bank and influence seedling mortality in post-burn habitats. The best explanatory models for high *B. multiradiata* recruitment were related to seeding treatment and warmer winter/spring minimum temperatures; high mortality for this species corresponded with seeding and greater plant densities (Table 16, Fig. 17). Similar patterns occurred for *S. ambigua* with canopy cover typically facilitating recruitment and greater plant densities corresponding to increasing mortality. In contrast, recruitment for the perennial grass *A. hymenoides* was negatively related to annual plant biomass, and mortality corresponded to a lower NMS 1 axis, which indicated low winter/spring precipitation and warm spring daytime temperatures (Table 16). *H. salsola*, *A. canescens*, and *E. virginensis* had similar explanatory models for recruitment and mortality; however, the slopes for these relationships were not significant ($P > 0.05$) in part due to low sample size for these responses.

Table 16. Best models (ΔAIC < 2) for recruitment and mortality of perennial seeded species with intercept-only model for comparison. *Ambrosia dumosa* and *Larrea tridentata* did not have any models that were better than an intercept-only model and are not presented. When the relationship between the variable and recruitment or mortality was significant, the positive or negative relationship is denoted in parentheses.

Recruitment					Mortality			
Species	Model	AICc	ΔAICc	wi	Model	AICc	ΔAICc	wi
Baimul	Seed, NMS3 (-)	427.4	0.0	0.3484	Seed, DENS (+)	203.3	0.0	1.0000
	Seed, T_{min} (+)	428.9	1.5	0.1646	Intercept	291.7	88.4	0.0000
	Seed, COV (+)	429.3	1.9	0.1347				
	Intercept	482.5	55.1	0.0000				
Sphamb	Seed, COV (+)	148.9	0.0	0.2260	DENS (+)	163.9	0.0	0.8570
	Seed	149.0	0.1	0.2150	Intercept	184.4	20.5	0.0000
	Seed, NMS2 (-)	150.2	1.3	0.1180				
	Seed, NMS3 (-)	150.2	1.3	0.1180				
	Intercept	153.5	4.6	0.0227				
Achhym	Seed, BMS (-)	24.2	0.0	0.4323	BMS (-)	2.1	0.0	0.2125
	Intercept	32.4	8.2	0.0072	NMS1 (-)	2.4	0.3	0.1829
					NMS2	2.7	0.6	0.1574
					Seed, BMS (-)	3.5	1.4	0.1055
					Seed, NMS1	3.6	1.5	0.1004
					Seed, NMS2	4.0	1.9	0.0822
					Intercept	8.1	6.0	0.0106
Hymsal	COV	2.5	0.0	0.3572	BMS	4.9	0.0	0.2224
	Intercept	5.1	2.6	0.0973	NMS3	6.1	1.2	0.1220
					Seed, BMS	6.5	1.6	0.0999
					Intercept	7.6	2.7	0.0576
Atrcan	Seed, BMS	36.6	0.0	0.2650	Seed	28.1	0.0	0.1525
	Seed	38.4	1.8	0.1077	Seed, COV	28.6	0.5	0.1188
	Seed, NMS3	38.4	1.8	0.1077	Intercept	28.6	0.5	0.1188
	Seed, COV	38.5	1.9	0.1025	DENS	29.1	1.0	0.0925
	Intercept	39.5	2.9	0.0622	COV	29.4	1.3	0.0796
					NMS3	29.4	1.3	0.0796
					Seed, NMS3	29.7	1.6	0.0685
					Seed, DENS	29.8	1.7	0.0652
Encvir	NMS3	28.3	0.0	0.1932	Seed, COV	19.8	0.0	0.2369
	Seed, NMS3	29.0	0.7	0.1362	COV	20.5	0.7	0.1670
	COV	29.4	1.1	0.1115	Seed	21.5	1.7	0.1013
	Seed, COV	30.0	1.7	0.0826	Intercept	21.7	1.9	0.0916
	Intercept	30.9	2.6	0.0527				

A high NMS1 axis value translates into lower spring maximum temperature (AJTmax, $r = -0.80$, $r^2=0.64$), high annual biomass (BMS, $r=+0.65$, $r^2=0.43$), and high winter/spring rainfall (OMmm, $r=+0.63$, r2= 0.39). A high NMS2 axis translates into low live canopy cover (COV, $r=-0.81$, $r^2=0.66$) and low perennial density (DENS, $r=-0.77$, $r^2=0.60$). Finally, a high NMS3 axis value translates to low winter/spring minimum temperature (OMTmin, $r=-0.91$, $r^2=0.83$).

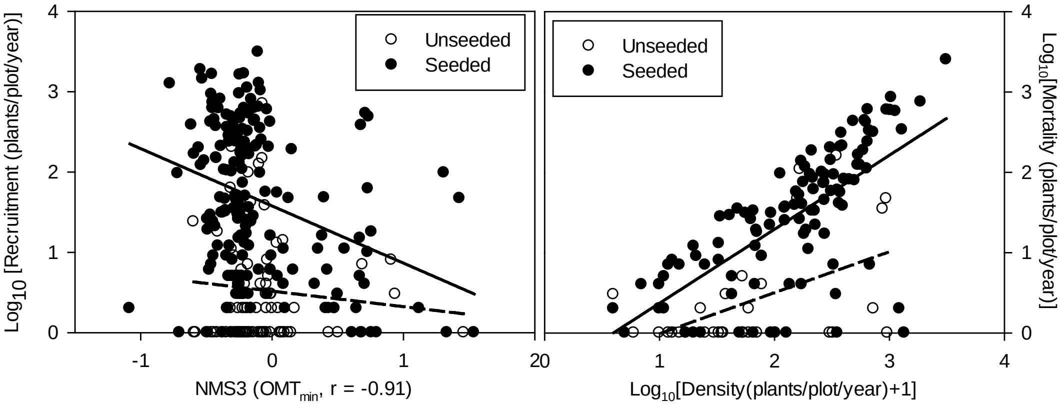

Figure 17. An example of the best explanatory models for annual recruitment (left) and mortality (right) illustrated for *B. multiradiata*. Solid lines represent seeded regression, dashed represents unseeded regression. Oct-Mar minimum temperature (OMT_{min}) was negatively related to NMS3 axis; therefore, greater recruitment is related to higher winter/spring minimum temperatures (left).

Discussion

Goal 1, Biomass Prediction

Areas of high exotic annual production on disturbed desert lands are of concern to managers because of their tendency to fuel wildfires. Once burned, these areas typically deter the re-establishment and growth of native annuals, perennial grasses and forbs, and shrubs. Although the majority of Nevada is public land (85%), little information is available for resource managers challenged with the landscape-level invasion of exotic annual grasses against a backdrop of changing disturbance regimes for native plants and wildlife habitats. We compared our field harvests of annual plant shoots with remote sensing data and interpolations of weather data to select a suitable estimate of fine fuel production. Our maps depict estimates for an interval of time (2001-2012) that represents a broad range of variability for this ecosystem. For example, 2000 through 2003 was a period when common shrub and grass species were selectively removed by drought in many regions of the Mojave Desert (Miriti et al. 2007, Webb et al. 2003). In contrast, other years produced abundant annual plant production including 2005, during which lightning strikes ignited fires that were fueled by abundant exotic annual plant production resulted in an unprecedented acreage of Mojave Desert habitat being burned (Brooks and Matchett 2006).

Although the years sampled were contrasting years of annual plant production (2010 and 2011), these years represent higher averages compared with the past seven years at multiple sites across the northeast Mojave Desert (L. DeFalco, unpublished data). We acknowledge that additional years representing a wider range of production will potentially reduce the average and increase the variability in the annual plant production, thereby de-emphasizing some areas identified by our maps for fuels management. Although we used only 2010 and 2011 data to select our best model, some of the areas depicted as high average and low variability production sites in our final maps do coincide with fires that recently burned in desert tortoise habitat (i.e., Southern Nevada Fire Complex). Additional sites we presented here are part of a larger network of sites that include areas of Grand Canyon-Parashant National Monument in northwestern Arizona. We will refine our results with on-going research by incorporating production from these sites along with more years of annual plant production (2006 through 2011). Future work will also include steps to validate our model with independent data sets. Additional refinements also include adding other candidate models, such as those that use seasonal differences in remote sensing reflectance data (comparison between summer versus spring imagery, Casady et al. 2013) instead of a single reflectance value for estimating production.

While our maps can be a starting place for discussion about prioritizing areas for managing fine fuels, we add the caveat that discussions about priority areas should include on-site evaluation before implementing control measures. Our approach did not differentiate between native and exotic annual plant production, or between previously burned and unburned habitats. While high production of mixed native and exotic species can fuel desert fires, these areas may also signal favorable habitat for wildlife that rely on annual plant production for food. Several of the areas identified as high annual plant production with low variation have historically supported populations of desert tortoises (e.g., Xmas Tree Pass and Piute Valley study plots; T. Esque, USGS, personal communication) and some of them are located in designated critical habitat for the desert tortoise (USFWS 2011). On-site assessment would distinguish between native-dominated annual plant diversity and exotic annual plant infestation. Alternatively, sites may have indicators of burning (typically short-lived, early colonizing species, Brown and Minnich 1984) or repeat burning (low perennial plant cover, exotic annual dominance, Brooks 2012). Recent USGS research on tortoise movements and feeding behavior in the Dry Rock and Dry Middle fires demonstrates that burned shrubland is still suitable habitat as it provides ample annual plants and herbaceous perennial forbs for food (Drake et al., unpublished data). These contrasting scenarios would guide differing intensities of management (e.g., a focus on reducing ignition sources such as along roadways in unburned areas versus directed suppression of consistently high exotic production and native plant establishment in burned and reburned areas).

Goal 2, Rehabilitation Treatment Assessment

Part I. Monitoring seeding effectiveness for augmenting native species establishment on the Goodsprings and Bonnie Springs fires, Red Rock Canyon National Conservation Area

Overall there is little evidence that the seeding efforts on the Goodsprings Fire were successful at re-establishing native perennial plants within the disturbed landscape at a rate greater than natural recovery. Success of seedings in desert ecosystems is difficult to predict due to large natural interannual fluctuations in precipitation and winter temperatures. Often successful seedings are those using many species, in effect bet hedging that a couple of species out of the mix will be successful that year (Abella, *et. al* 2009). For many species, cold stratification is often necessary to break dormancy patterns in seed.

Additionally, the rate of seed application was very low for some species, as low as two seeds per acre. We would not expect to see an increase in the population of those species when the application rate is so low. Herbivory rate is great in the desert, and we see that many seeds – especially those with large desirable seeds are lost to seed herbivory by rodents and ants. Therefore, to be successful, seeding rates need to overcome herbivory pressure

Unsurprisingly, with a lack of conspicuous differences in seeded species we also noted very few differences in plant community composition between seeded and control sites. We predicted that there may have been some indirect effects of seeding with greater granivory rates due to the addition of seed, but those results were not observed. At the seeding rate and lack of establishment of target species in seeded plots, there was apparently no impact on the established and naturally regenerating communities.

Unlike the Goodsprings seeding effort, there appears to be no shortage of seed material distributed across the Bonnie Springs seeding site. However, even with an apparent abundance of seed material, there still were few and only marginal benefits to the plant community in seeded plots. Only one species, *Baileya multiradiata*, showed signs of increased seedling abundance in plots that received supplemental seed input, and that trend is marginal. As qualitatively more plants add seeds to the seed bank, this trend may be amplified through time and result in a quantitative difference among treatments. Patterns like this suggest that continued monitoring is important for evaluating seeding treatment success in desert systems whose harsh environments result in relatively slow successional processes and population developments. Three main questions should be addressed to explain the success (or lack of success) in Mojave Desert seedings. First, which environmental and weather characteristics are necessary for successful germination (and establishment)? Second, what is the seed fate of distributed seed material? And third, what is the mortality rate of seedlings that do emerge?

Desert species have complex dormancy mechanisms. Highly variable precipitation and winter temperatures make predicting years that may produce abundant cohorts of seedlings difficult. One way to combat such variability is to apply a diverse seed mix, thereby increasing the chances of having species that are susceptible to germination across a range of conditions. Often seed availability is limited, especially when trying to use seed that is locally or regionally sourced, and when ordering seed after catastrophic disturbances (like wildfire), and when time is limiting. However, these are situations that land managers should be aware of, and prepare when possible by stockpiling seed or keeping in close contact with growers about what is available even in years when the supply is not needed.

We did not observe a presence of seeded species stored in the soil seed bank. Therefore, that begs many questions about the seed dynamics once it is distributed. Burned areas are generally denuded of native vegetation, therefore there may be a greater need for a food source by the local granivore populations (primarily rodents and ants) than exists in undisturbed communities. In Mojave desert ecosystems, rodent granivores may harvest as much as 80% of a plant’s seed production. They play a significant role in seed fate in intact systems, therefore, in disturbed systems there may be a greater need for food source, and they may predate a large proportion of what is distributed in experimental seedings. Adding seeds to the system may function as providing a food source for granivores, and many of the seed distributed may be consumed before it has a chance to establish.

Additionally, post-germination survival of seedlings in the Mojave Desert is tenuous under the best of conditions. If sufficient precipitation and temperature regimes occur to promote germination from seeds, then the seedlings still need to establish root and shoot systems sufficient to facilitate survival through the physiologically stressful summers. Additionally, young plant shoots are a prime food source for herbivores, so even if a seed is able to germinate, the seedling still must survive many environmental pressures to survive long enough to be counted in experimental monitoring.

We need additional research and information on seed fate and herbivory to explain exactly why seedings so often fail to establish greater abundances of reproductively mature seeded species. If the ultimate goal is establishing plants that can then serve as seed sources, it may be wise to conduct cost-benefit analyses of seeding success rates vs. outplanting success (creating islands) at the landscape scale to evaluate which management activities are most successful at establishing habitat and performing necessary ecological functions (e.g. soil stabilization, nutrient cycling, water infiltration rates).

Part II. Evaluation of hand seeding and herbicide application for rehabilitation of Mojave Desert tortoise habitat

Post-fire seeding with high densities of native low-desert species was effective at increasing establishment of perennials relative to unseeded burn areas. This seeding effect was detected at four to seven years after seeding, a time period beyond the duration of the three-year Emergency Stabilization and Rehabilitation (ESR) Program typically implemented by BLM immediately following wildfires. This seeding effect compares favorably with results from the Bonnie Springs Fire, another hand-seeded site that showed trends, although not yet significant, for the same early-colonizing species, *Baileya multiradiata*, that we detected across multiple fires. Multiple years of monitoring are imperative when evaluating treatment effectiveness because it takes time to capture the rainfall pulse and drought interpulse dynamics that drive regeneration of plant communities in arid environments. We found that total canopy cover was not a sensitive metric for detecting short-term differences in seeding treatments (as with the species *B. multiradiata*) because canopy cover is influenced by rainfall and is typically slow to develop in disturbed desert shrublands (Webb et al. 2003). Monitoring metrics that are sensitive to plant establishment—such as seedling density, frequency of occurrence, and seed bank abundance—are more appropriate for gauging short-term restoration success in desert shrublands (i.e., 1-3 years).

Low success in the establishment of perennial grass and later-colonizing shrubs from the seed mixture was likely due to lower seeding rates. *Atriplex canescens*, *Achnatherum hymenoides*, *Hymenoclea salsola*, *Larrea tridentata*, and *Encelia virginensis* seeding rates ranged from < 1 to 27 live seeds/m^2, which were likely inadequate to overcome rapid colonization and resource uptake by competitive exotic annuals that dominated the post-burn landscape (DeFalco et al. 2003). The recruitment of early-colonizing species such as *B. multiradiata* and *S. ambigua* that were each seeded at much higher rates (>250 seeds/m^2) was positively related to live canopy cover of the previous year. This correlation between recruitment and canopy cover implies a positive feedback either by improving conditions beneath the developing canopies for emerging seedlings (Titus 2002, Miriti 2006, Reynolds et al. 2012) or by replenishing seed bank for future recruitment. Individual *B. multiradiata* and *S. ambigua* plants often emerged from the soil, matured and reproduced within the same year. Mortality, in contrast, was positively related to the previous year's plant densities suggesting that competition in these burned areas was also intense. Despite a six-fold increase of seeded forbs in the soil seed bank relative to untreated plots, densities of seeded shrub species were found to be significantly lower in seeded areas, alluding to either failed establishment after germinating in seeded areas where competition was high, or greater seed and/or seedling predation by ants, rodents, or birds that may have been attracted to seeded areas. We noted active rodent burrows (e.g., kangaroo rats, *Dipodomys* spp.) and harvester ants carrying seeds to their nests in burned areas during our monitoring. Seed losses due to granivory after seeding at high rates can be substantial (600 seeds/m^2, >80% loss, DeFalco et al. 2012). Currently, we do not know if a seeding rate threshold exists where the establishment of plants cached by rodents, in particular, can offset the losses of seeds through consumption. In related research, we are evaluating seed encapsulation methods that protect seeds from granivores in combination with lower seeding rates at burned sites in northwestern Arizona.

It is important to note that high seeding rate did not always ensure establishment. While the summer annual species *Bouteloua aristidoides* was applied at a proportionally high seeding rate, it was rare to absent from a 2007 census following high summer rainfall (monitoring data not shown) and from the soil seed bank. The greater residence time in the soil surface awaiting the narrow opportunity for optimal conditions makes these seeds vulnerable to seed harvesting ants and rodents (DeFalco et al. 2012). Winter annuals such as *Vulpia octoflora* are likely to have broader climatic opportunities for establishment than summer annuals (e.g., winter storms are more predictable and cover greater area than localized summer storms), especially if seeded at sufficient rates and/or facilitated by other methods such as herbicides. Overall, species with broader germination and emergence requirements will be better candidates for seed mixtures used in future seedings, but establishment traits are poorly understood for many desert species and require further investigation.

Herbicides were successful at reducing exotic annual grasses and forbs and also increased native annual species. Reduction of exotic annuals using herbicides may be desirable in heavily infested areas because red brome has limited longevity in the soil seed bank and lacks seed dormancy (DeFalco et al. 2003, Jurand et al. 2013). Our experimental study plots were small (10 m × 10 m), so herbicide effects were diminished as new seeds from outside the plots rapidly recolonized. Even so, the suppressive effect of herbicides, particularly from the imazapic-based Journey™ and Plateau™, demonstrates that native annuals can be augmented by relaxing the competitive pressure from rapidly growing exotics that capture resources faster than their native counterparts (DeFalco et al. 2003) and readily re-establish in the soil seed bank following wildfire (Jurand and Abella 2013). We have also documented this pattern of native plant augmentation using Plateau™ in severely degraded tortoise habitat at northwestern Arizona at sites that have repeatedly burned: a second application of herbicide continues to suppress exotic annuals with negligible impacts on the native annuals (L. DeFalco, unpublished data). We expect that a program of herbicide application over larger treatment areas (e.g., applications of several acres per year staggered through time) will diminish exotic annuals for a longer period of time, thereby providing sufficient opportunity for proliferation of native annual species. Additionally, these opportunities will augment the perennials that provide habitat structure and thermal cover for recovering desert tortoise habitat. Larger experimental areas of herbicide application in tortoise habitat have not yet been attempted but are nevertheless important for validating such expectations and potentially improving tortoise habitat for the recovery of this species.

Part III. Contrasting survival of two outplanted Mojave Desert perennials

Seedling survival differed markedly between the two shrub species outplanted across burned areas in the northeast Mojave Desert, with 15-fold higher survival for Nevada jointfir than blackbrush after 3.5 years. For jointfir, markedly decreased survival at the Loop site coincided with removal of the protective cones and cages. Plants at this site experienced two to seven times more herbivore damage than plants at the other sites after protection was removed. Removal occurred in a year when fall/winter rainfall arrived late, and herbivores consumed the exposed jointfir seedlings when winter annual production was predictably low (Beatley 1974). Although we did not monitor small mammals during our experiment, precipitation-mediated pulses in animal populations are often coincident with increased frequency of herbivory in North American deserts (Beatley 1969; Manrique et al. 2007; Gutierrez et al. 2007). In contrast to jointfir, we noted very little herbivore damage to blackbrush after protection was removed, which may be a consequence of tannins that deter herbivory (Provenza and Malechek 1983).

Plastic cones elevated maximum daily air temperatures around the seedlings by more than 4°C, enhancing blackbrush survival during establishment in late fall and winter, but ultimately reducing survival during warmer spring conditions. In contrast, survival of jointfir was no different for plants protected with cages or cones, suggesting that it is more tolerant of increased temperatures during establishment. This broader environmental tolerance is consistent with the widespread elevational distribution of jointfir in the Mojave Desert. The distribution of blackbrush, on the other hand, is considered to be limited by the narrow temperature and precipitation conditions necessary for recruitment (Meyers and Pendleton 2005).

In contrast to survival during times of low rainfall, lower survival during periods of higher precipitation may be attributed to a greater abundance of non-native annual grasses (Beatley 1974; Hunter 1991), whose rapid resource uptake creates a competitive environment for desert perennials (DeFalco et al. 2003, 2007) and places native perennial species with slower uptake rates at a disadvantage (Gebauer and Ehleringer 2000). Herbicides containing imazapic (*e.g.*, Journey™) can reduce non-native annual plant production in the Mojave Desert by 50–80% (L. DeFalco, unpublished data) and diminish post-emergent growth of downy brome (*Bromus tectorum*) in the Great Basin Desert (Hirsch et al. 2012). However, jointfir was susceptible to Journey™ application following outplanting, as reported for other shrub and perennial grass species (Shinn and Thill 2004, Elseroad and Rudd 2011, Sbatella et al. 2011). Lower jointfir survival may reflect herbicide absorption from shallow soils where preferential uptake of nutrients and water occurs (Gebauer and Ehleringer 2000).

In addition to conditions experienced after outplanting, survival of slow-growing blackbrush (4% after 3.5 years) was strongly influenced by greenhouse conditions. One greenhouse clipped the crowns of all plants 1–2 months after germination to promote root development (A. Dunn 2012, Nevada Division of Forestry, pers. comm.), and these blackbrush had greater survival in the field than those from the other greenhouse. Clipping potentially stimulated allocation of resources to root development, resulting in greater soil moisture uptake per unit leaf area as soil moisture declined during spring. Alternatively, plants that died after clipping would have been discarded, pre-selecting the most resilient plants for outplanting. While this study did not directly compare practices for propagating seedlings for outplanting, the impact of greenhouse conditions on post-outplant survival warrants further study.

Counter to short-term watering benefits on establishment of Mojave Desert species (Abella et al. 2012), supplemental watering and soil amendment treatments did not influence long-term (43 month) survival of either blackbrush or jointfir outplanted in 2007. Similarly, an initial increase in survival of outplanted seedlings with supplemental watering on the Colorado Plateau™ disappeared over the course of 4.5 years (Minnick and Alward 2012), implying environment overrides any treatment benefits to the long-term survival of outplanted seedlings. This lack of effectiveness may be explained by the relatively small volume of soil impacted by the applied treatment compared to the vast surrounding volume of dry, untreated soil drawing away soil moisture. We found that the localized increase in soil moisture that resulted from watering, for example, was smaller and lasted a shorter time than the soil moisture increase from a 2 mm rainfall event (data not presented).

Based on our findings, the most consistently important factor in successful outplanting of greenhouse-raised seedlings is careful site selection. Fast-growing species such as jointfir are desirable, but special consideration should be given to the long-term protection of seedlings from herbivores and the timing of herbicide application, if desired. Slow-growing blackbrush proved much less susceptible to herbivores and herbicide, but overall long-term survival was 15-fold lower. For blackbrush, the type and timing of herbivore protection needs to be considered carefully – plastic cones should be removed prior to the onset of late spring temperatures, while cages can remain in place indefinitely.

Goal 3, Evaluation of Native Species

Little information exists to guide managers and restoration specialists toward optimal seeding rates and suitable species for re-establishing perennial plants in burned Mojave Desert shrublands. Recent work at the Goodsprings Fire in the Mojave Desert (Clark County, NV) found that seeding with native perennial mixes ranging from 425 -1,270 pure live seeds/m^2 resulted in zero seedling establishment after 3 years (Abella et al. 2012). Individual species seeding rates ranging from 210-338 pure live seeds/m^2 (*Larrea tridentata*, *Ambrosia dumosa*, *Hymenoclea salsola*) and 420-425 seeds/m^2 (*Sphaeralcea ambigua*, *B. multiradiata*) (Abella et al. 2012). An extended dry period (as indicated by negative Palmer Drought Severity Index) occurred for more than two years after seeding in the Goodsprings study, and although plots were caged to reduce small mammal granivory, losses of seeds to harvester ants may have occurred as seeds awaited favorable conditions on the soil surface. Another recent seeding study at the Bonnie Springs Fire (Clark Co., NV) resulted in minimal increases of *B. multiradiata* after 4 years when seeded at 136 seeds/m^2, but no seeding effects for *S. ambigua* seeded at 123 seeds/m^2 *Aristida purpurea* seeded at 44 seeds/m^2, or *H. salsola* seeded at 22 seeds/m^2 (Goal 2, Part I). These studies demonstrate that re-vegetation is more complicated than expected because increased seeding does not always yield increased establishment. Favorable rainfall and temperature may stimulate seed germination and emergence, but seedlings will not become established unless favorable conditions persist. Although not well-studied for most Mojave Desert species, perennial shrub species are generally less likely than perennial forbs or annuals to form a persistent seed bank because they tend to produce seeds lacking dormancy (Jurado and Flores 2005). Thus, seeds of shrub species in particular likely have a brief opportunity to encounter suitable conditions. If emergence conditions are delayed, seeds that remain on the soil surface for extended time are vulnerable to seed-harvesting animals and decreasing viability.

Our studies that evaluated seeding and seeding in combination with herbicides demonstrate that for the early-colonizing perennial herbs, *B. multiradiata* increased in the seed bank two and five years after seeding and *S. ambigua* marginally increased. This enrichment in the soil seed bank associated with seeding parallels the increase we observed in plant density and cover. Because perennial herbs can emerge from the seed bank and reproduce in the same year, their success in burned areas is in part attributed to their ability to replenish the seed bank from one year to the next. This ability is especially important when seed longevity in the soil is short-lived and perennial seeds are consumed by granivores. Greater yearly recruitment *B. multiradiata* was related to warmer night time temperatures during the winter/spring period when seedlings establish in response to precipitation. Yearly mortality corresponded to high plant densities particularly in seeded areas. This shift from facilitation to competition was also evident for *S. ambigua* and is consistent with changing seasonal associations between adult and juvenile plants in desert shrublands (Miriti 2006, Reynolds et al. 2012). Interestingly, high recruitment for *A. hymenoides* corresponded to lower annual plant biomass and suggests exotic annual grasses exerted a competitive effect on establishment of this perennial grass.

Shrub species such as *Larrea tridentata* and *Ambrosia dumosa* dominated adjacent unburned areas, yet we did not detect these species in the seed bank of the burned areas where they were seeded ($\leq$ 40 seeds/m^2 seeding rate for each species). These species germinate in response to infrequent summer storms July through September when temperatures are above 15 °C, thus opportunities for recruitment during the 2- and 5-yr monitoring periods were rare (Barbour 1968, Kay et al. 1988). Consequently, their prolonged time in the surface soils awaiting favorable conditions make them vulnerable to predation by ants and rodents (DeFalco et al. 2012). We also speculate that large inputs of seeds into the burned areas (Table 3) could attract seed consumers to the seeding areas in general, and they could deplete the seeds of preferred species from the seed bank (Orrock et al. 2008). Alternative methods such as outplanting greenhouse-raised seedlings may result in greater establishment success for species that are vulnerable to granivores (Abella et al. 2012), but may be unsustainable over large disturbances such as desert wildfires.

We expect that as time passes, the plant communities in the recovering burned areas will shift. Early-colonizing species such as *B. multiradiata* will likely continue to dominate as long as they can replenish the seed bank and maintain yearly recruitment. This dominance may preclude establishment by later colonizing species if densities remain high enough to outcompete establishing seedlings. However, during drought periods, early-colonizing species typically experience high mortality and are expected to decline while drought-resistant species that are currently re-establishing in lower numbers such as *L. tridentata* will increase in dominance (Miriti et al. 2007). However, recovery trajectories can be rapidly reset if wildfire fueled by invasive annual grasses returns.

Conclusions

Identification of areas with consistently high annual plant production in Clark County, Nevada can increase the efficiency of land management efforts without spending many person-hours in the field to conduct production estimates that are based on actual harvests or ocular surveys of production. Sites of interest can be narrowed to a targeted list of high production areas for field teams to assess and prioritize. Our fine fuel maps provide a tool for managers to begin the discussion on where and how to proactively manage areas for fuels reduction in tortoise habitat (e.g., suppression of exotic annual plants using herbicide and re-seeding of native species). Identification of high production areas may also assist land managers in deciding where to place fire breaks and their orientation on broad landscapes. Careful effectiveness monitoring should be designed and implemented when such landscape-level management actions are conducted.

Seeding with native species demonstrated variable results at a variety of fire locations in our study. Neither the Goodsprings seeding nor the Bonnie Springs seeding produced greater abundances of seeded species for most of the species used in the seeding treatments. When plant and seed material is sparse, the likelihood of success of seeding as a restoration tool is decreased. The Dry Rock, Dry Middle, Garnet, Meadow Valley, Halfway, and Tramp fires, in contrast, demonstrated greater plant densities on seeded burned areas four to seven years after hand seeding. These variable responses across sites were likely influenced by differences in seeding method (aerial versus hand seeding), rates of seeding, granivore activity, species composition in seed mixes, and prevailing weather conditions following implementation. Overall seeding results demonstrate that seeding efforts encouraged native early successional species (*Baileya multiradiata* at most fires, and tenuously *Sphaeralcea ambigua*). Currently, because there is no difference in the canopy cover and species composition of these short-term studies, we are unable to predict long-term trajectories for recovery. As seen in other research, even when successful, artificial seeding in desert regions many take several years to establish. Future research objectives should track the fates of seeds used for restoration and evaluate the indirect effects of attracting granivores to seeded areas which could have a deleterious impact on naturally occurring seedling regeneration.

While seeding alone showed some success, the use of herbicides, particularly those with the active ingredient imazapic, can enhance the establishment of seeded native species. Dramatic reductions of exotic annual grasses and forbs in the seed bank that persisted for multiple years provide a tool for suppressing strong competitors of native plant establishment. We saw enhanced establishment of early-colonizing perennial species as well as a native annual grass, which carried over to the seed bank. Future research objectives should address the use of repeat application of herbicides on managing exotics in the seed bank as well as evaluating different rates of application and at larger spatial scales.

For species that are not conducive to broadcast seeding (e.g., large-seeded species that are preferred by heteromyid rodents), outplanting can be an effective alternative for establishing islands of plant material within broad disturbed landscapes. Fast-growing species such as Nevada jointfir can be useful for providing structure to wildlife in a relatively short period of time (1-5 years) and potentially encourage seed harvesters such as rodents to move into severe disturbances and import their seed caching behaviors. In addition, when seeding of late-colonizing species is unsuccessful, outplanting options may be more desirable, although planting densities should be augmented to account for potentially low survival. Used in combination with seeding of early-colonizing species, outplanting of late-colonizing species such as blackbrush or creosotebush may circumvent the vulnerability of these species during emergence and establishment stages. Ultimately, the combined approaches of seeding and outplanting together may increase the diversity of plant functional groups on the landscape, thereby augmenting the resistance and resilience of these unique shrubland communities to future disturbances.

Recommendations

- Imazapic-containing herbicides such as Plateau™ or Journey™ can be a useful tool for managing exotic annual grasses and forbs in burned areas, thereby providing a window of opportunity for establishing native perennials and annuals.
- A cost-benefit analysis should be conducted evaluating the effectiveness of seedings versus seedling outplantings as methods of increasing the density of reproductively mature native perennial plant species for habitat rehabilitation and ecosystem functions.
- *In situ* studies of granivore populations and seed preference would greatly inform questions about seed fate after seedings are conducted.
- Seeding rate is crucial to re-vegetation success, but we lack sufficient information on seed fate and seed species composition to know the threshold of how much seed overwhelms the native populations of granivores.

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