Fire Rehabilitation Effectiveness: A Chronosequence Approach for the Great Basin

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Abstract

Federal land management agencies have invested heavily in seeding vegetation for emergency stabilization and rehabilitation (ES&R) of non-forested lands. ES&R projects are implemented to reduce post-fire dominance of non-native annual grasses, minimize probability of recurrent fire, quickly recover lost habitat for sensitive species, and ultimately result in plant communities with desirable characteristics including resistance to invasive species and resilience or ability to recover following disturbance. Land managers lack scientific evidence to verify whether seeding non-forested lands achieves their desired long-term ES&R objectives. The overall objective of our investigation is to determine if ES&R projects increase perennial plant cover, improve community composition, decrease invasive annual plant cover and result in a more desirable fuel structure relative to no treatment following fires while potentially providing habitat for Greater Sage-Grouse, a species of management concern. In addition, we provide the locations and baseline vegetation data for further studies relating to ES&R project impacts.

We examined effects of seeding treatments (drill and broadcast) vs. no seeding on biotic and abiotic (bare ground and litter) variables for the dominant climate regimes and ecological types within the Great Basin. We attempted to determine seeding effectiveness to provide desired plant species cover while restricting non-native annual grass cover relative to post-treatment precipitation, post-treatment grazing level and time-since-seeding. Seedings were randomly sampled from all known post-fire seedings that occurred in the four-state area of Idaho, Nevada, Oregon and Utah. Sampling locations were stratified by major land resource area, precipitation, and loam-dominated soils to ensure an adequate spread of locations to provide inference of our findings to similar lands throughout the Great Basin.

Nearly 100 sites were located that contained an ES&R project. Of these sites, 61 were seeded by using a drill, 27 were broadcast aerially, and 12 had a combination of both. We randomly sampled three burned and seeded, burned and unseeded, and unburned and unseeded locations in the vicinity of the fire, each within the same ecological site. We measured foliar cover of all plant functional groups (perennial or annual, shrub, grass, forb, native or introduced), biological soil crusts, and abiotic (bare soil and litter) variables using the line-point intercept protocol. Fuel loads and horizontal fuel continuity were measured. We applied linear mixed models to response variables (cover and density of plant groups) relative to the dependent variables (seeding treatments and precipitation/temperature relationships.

Post-fire seedings with native perennial grasses or shrubs in mixes did not increase density or cover of these groups significantly relative to unseeded, burned areas. Seeded non-native perennial grasses and the shrub *Bassia prostrata* were effective in providing more cover in aerial and drill seedings. Seeded non-native perennial grass cover increased with increased annual precipitation regardless of seeding type. Seeding native shrubs, particularly *Artemisia tridentata*, did not significantly increase shrub cover in burned areas. Cover of undesirable non-native annual grasses was lower in drill seedings relative to unseeded areas but only at higher elevations. Seeding effectiveness after wildfire is unpredictable in drier, low elevation environments, and our findings indicate management objectives are more likely met when focusing efforts on higher elevation or higher precipitation locations where establishment of

perennial grasses is more likely. On sites where potential for invasion and dominance of non-native annuals is high, such as lower and drier sites, intensive methods of restoration that include invasive plant control before seeding may be required. Where establishment of native perennial plants is the goal, managers might consider using native-only seed mixtures, because we found that the non-native perennials typically used in Great Basin restoration efforts are selected for their competitive nature and may reduce establishment of less competitive native species. Although we attempted to include information on livestock grazing history after seedings, we were unable to extract sufficient data from files to address this topic that may play an additional role in understanding native plant abundance post-fire seeding.

Evaluation of drill and aerial seeding effects on fuel characteristics focused on two metrics that are standard inputs for fire behavior models, fuel load and fuel continuity. Fuel loads were evaluated separately for total fuel load biomass, and the individual components that sum to total biomass, namely herbaceous, shrub, shrub:herbaceous ratio, litter, 10-hour, and 100-hour fuel biomasses. Fuel continuity was evaluated using the following cover categories, total, annual grass, annual forb, perennial forb perennial grass, shrub, litter, vegetative interspace, and perennial interspace. Drill seeding did not affect fuel loads, except to reduce 10-hour fuels, probably due to mechanical destruction of dead and down fuels by the drill seeding equipment. Drill seeding did affect fuel continuity, specifically decreasing total plant cover by increasing perennial grass cover which suppressed annual grass and litter production resulting in a net decrease in continuity, but only at the elevations above approximately 1500m. Aerial seeding had no effect on any fuel load or fuel continuity category.

For the Greater Sage-Grouse habitat study, we developed multi-scale empirical models of sage-grouse occupancy in 211 randomly located plots within a 40 million ha portion of the species' range. We then used these models to predict sage-grouse habitat quality at 101 ES&R seeding projects. We compared conditions at restoration sites to published habitat guidelines. Sage-grouse occupancy was positively related to plot- and landscape-level dwarf sagebrush (*Artemisia arbuscula, A. nova, A. tripartita*) and big sagebrush steppe, and negatively associated with non-native grass and human development. The predicted probability of sage-grouse occupancy at treated plots was low on average (0.07–0.09) and was not significantly different from burned areas that had not been treated. Restoration was more often successful at higher elevation sites with low annual temperatures, high spring precipitation, and high plant diversity. No plots seeded after fire (n=313) met all overstory guidelines for breeding habitats, but approximately 50% met understory guidelines, particularly for perennial grasses. This trend was similar for summer habitat. Ninety-eight percent of treated plots did not meet winter habitat guidelines. Restoration actions in burned areas did not increase the probability of meeting most guideline criteria. The probability of meeting guidelines was influenced by a latitudinal gradient, local climate, and topography. Post-fire seeding treatments in Great Basin sagebrush shrublands generally have not created high quality habitat for sage-grouse. Understory conditions are more likely to be adequate than those of overstory, but in unfavorable climates, establishing forbs and reducing cheatgrass dominance is unlikely. Reestablishing sagebrush cover will require more than 20 years using the restoration methods of the past two decades. Given current fire frequencies and restoration capabilities, protection of landscapes containing a mix of dwarf sagebrush and big sagebrush steppe, minimal human development, and low non-native plant cover may provide the best opportunity for conservation of sage-grouse habitats.

Our database of ES&R locations has used the Land Treatment Digital Library to archive data and location information regarding our study (see Pilliod and Welty 2013). This has contributed to two additional studies. One examined the potential spread of *Bassia prostrata* (aka *Kochia prostrata*; forage kochia) from ES&R project locations (Gray and Muir 2013). The second used remote sensing to determine the phenology of vegetation green-up on post-fire seeded sites (Sankey et al. 2013).

Introduction

Federal land management agencies have invested heavily in seeding vegetation for emergency stabilization and rehabilitation (ES&R) of non-forested lands. ES&R projects are implemented to reduce post-fire dominance of non-native annual grasses, minimize probability of recurrent fire, and ultimately result in plant communities with desirable characteristics including resistance to invasive species and resilience or ability to recover following disturbance. Although some studies and monitoring have provided information on the short-term (typically ≤3 year) effects of post-fire seedings, land managers lack scientific evidence to verify whether seeding non-forested lands achieves their desired long-term ES&R objectives. The overall objective of this chronosequence investigation was to determine if ES&R projects that were implemented 8-21 years prior have increased perennial plant cover, improved community composition, decreased invasive annual plant cover and resulted in a more desirable fuel structure relative to no treatment following fires. The null hypotheses that were evaluated were as follows:

H1: Cover of perennial life-forms (grass and shrub) will not differ in seeded and unseeded plots regardless of average annual precipitation.
H2: Cover of exposed mineral soil will not differ in seeded and unseeded plots regardless of average annual precipitation.
H3: Invasive annual grass cover will not differ in seeded and unseeded plots regardless of average annual precipitation.
H4: Fuel loads will not differ in seeded and unseeded plots regardless of average annual precipitation.
H5: Continuity of fuels will not differ in seeded and unseeded plots regardless of average annual precipitation.

These hypotheses were tested in two separate analyses and the results summarized and reported in two separate journal manuscripts. The first three hypotheses (H1, H2, H3) were addressed in a manuscript that focused on vegetation responses, and the last two hypotheses (H4, H5) were addressed in a manuscript that focused on fuels responses. A third manuscript takes findings from the ES&R projects and relates them to habitat guidelines for Greater Sage-Grouse, a common species of concern throughout the study area. The key findings from these three manuscripts constitute the majority of the information in this final project report. Management implications, future studies needed, and deliverables are then presented jointly at the end of the report.

Background and Purpose

An average of two million hectares currently burn each year from wildfires within the Great Basin USA (US National Interagency Fire Center, 2001 – 2012 eastern and western Great Basin; http://www.nifc.gov/fireInfo/fireInfo_stats_lightng.html accessed 22 July 2013). Much of the area burned consists of federal lands managed under jurisdiction of the Bureau of Land Management (BLM). BLM currently implements an emergency stabilization and rehabilitation

(ES&R) program to mitigate potential negative effects from wildfires, particularly in Great Basin shrub steppe (USDI BLM 2007). This program's treatments often include aerial or drill seeding with native and non-native perennial grasses, forbs and shrubs. Treatment objectives are to decrease potential soil erosion, increase desirable perennial plant cover (typically deep-rooted perennial grasses), improve wildlife habitat, and reduce abundance of invasive plants, particularly non-native annuals (USDI BLM 2007).

Current ES&R policy mandates post-seeding effectiveness monitoring be conducted during the first three years after seeding (USDI BLM 2007). Although monitoring programs can detect initial establishment of seeded species, the time frame is typically too short (three years) to determine effects on relative species dominance or long-term community trajectories. Post-fire seedings are expensive requiring $28.58 million USD per year for BLM to complete ES&R treatments in Oregon, Idaho, Nevada and Utah (http://www.blm.gov/public_land_statistics/index.htm; accessed 17 September 2013). Aerial and drill seeding treatments may also pose potential ecological impacts (e.g., shifting the competitive balance among desired species or facilitating non-native annual grasses) to vast areas of fire-affected plant communities and wildlife on public lands, but few studies have evaluated ecological effects of post-fire seeding or how well treatments meet agency objectives (Pyke, Wirth & Beyers 2013). Fuelbed characteristics are a primary factor affecting fire behavior, and the relative abundance of perennial versus annual plant species (primarily non-native annual grasses) largely dictate fire behavior in shrubland communities (Scott & Burgan 2005). In addition, perennial herbaceous species, bare ground, and litter cover are primary determinants of resistance to non-native annual species (Chambers et al. 2007; Reisner et al. 2013) and soil erosion in desert shrublands (Sankey et al. 2009). To our knowledge, no one has examined long-term effectiveness of these treatments at a regional scale.

In addition, one of the staples of post-fire fuels management and ecosystem recovery in western North America is the Emergency Stabilization and Rehabilitation (ES&R) program (www.doi.gov/pmb/owf/es_bar.cfm). This program addresses the responsibility of federal agencies to take prompt action to determine the need for, and to prescribe and implement, emergency treatments to minimize threats to life or property or to stabilize and prevent unacceptable degradation to natural and cultural resources resulting from the effects of a fire on the lands they manage. ES&R treatments in shrub steppe ecosystems typically focus on seedings to decrease potential soil erosion, increase desirable perennial plant cover and species composition, and reduce abundance of invasive plants, particularly non-native annuals (USDI BLM 2007). Although "treating fuels within the burned area to accomplish fuel management objectives" is an explicitly prohibited use of ES&R funding (USDI BLM 2007, p. 79), this restriction is only due to the administrative structure of wildland fire management in the United States which includes a Hazardous Fuels Reduction Program (www.doi.gov/pmb/owf/hfr.cfm) that is separate from the ES&R program. ES&R seeding treatments are designed to suppress the growth of invasive non-native annuals and promote the dominance of native perennials on shrub steppe landscapes thus enhancing ecosystem recovery that includes the recovery of the ecosystem's fire regimes. Fire regimes in this case are recovered by producing fuelbeds that lead to reduced frequency and extent of fires from the altered state created by invasive plants.

Fuels management can be implemented by various methods, including actively targeting undesirable fuel types to reduce their abundance (e.g. using herbicides, mechanical thinning, livestock grazing, prescribed fire) or promoting the growth and abundance of desirable fuel types (e.g. by seeding, outplanting, protection for other disturbance factors). In some cases, increased abundance of desirable plant species can also facilitate the reduction of undesirable species through competitive suppression. The most common ES&R treatment applied in shrub steppe ecosystems is post-fire seeding. A few synthesis studies report that post-fire seeding rarely accomplishes soil conservation or invasive species management objectives in chaparral and forests (Robichaud et al. 2000, Beyers 2004) or arid shrublands or grasslands (Pyke et al. 2013), but these studies only focus on short-term effects. For example, the review that included seeding effect on invasive plants in arid shrublands evaluated 19 studies with an average of 4 years post-fire (range 1-10, mode 3) (Pyke et al. 2013, Table 2). In relatively low productivity ecosystems such as shrub steppe it may take longer for seeded species to establish and exhibit their intended effects. In addition, these syntheses do not include any assessment of seeding effects on fuel characteristics.

Although the ES&R program was not specifically designed to restore habitat for sensitive species such as Greater Sage-Grouse, successful projects should "restore or establish a healthy, stable ecosystem in which native species are well represented" (USDI BLM 2007). Further, these projects represent an important sage-grouse conservation opportunity for three reasons: (1) ES&R projects constitute by far the largest number of hectares treated and dollars spent on restoration in the Great Basin (e.g., $60 million in 2007), (2) most individual ES&R projects (73%) cite a need to improve wildlife or sage-grouse habitat as specific project objectives or concerns (these projects account for 3.9 million ha, or 81% of all hectares treated since 1990 according to LTDL data), and (3) studies have found that native plant restoration in degraded areas is significantly more successful when preceded by non-native plant removal via fire or other means (Davies 2010, McAdoo et al. 2013, Miller et al. 2013).

To address these information needs, we conducted a series of studies using measurements from post-fire ES&R seedings that were previously implemented within Great Basin shrub steppe of the semi-arid western USA. Our goals were to: (1) determine long-term (>5 years) effects of post-fire seedings on cover and/or density of native and non-native seeded life-forms, cover of non-native life-forms that were unseeded including annual bromes and forbs, and abiotic cover of bare ground and litter; (2) evaluate fuelbed responses that would be consistent with fuel treatment objectives of minimizing fuel conditions that promote higher fire frequency and extent, maximizing conditions that promote lower fire frequency and extent, and reversing the establishment of an invasive plant / fire regime cycle; and (3) determine plot- and landscape-scale habitat associations of sage-grouse and to use this information to quantify the effects of post-fire restoration treatments on habitat quality throughout the Great Basin.

Study Description and Location

From 2008 to 2010, 19 BLM offices in Idaho, Nevada, Oregon and Utah were visited to collect post-fire seeding data from historical post-fire ES&R plans and implementation records

(Knutson et al. 2009). Data sufficient to determine the location and basic characteristics (e.g., planned or actual species sown) of seeding treatments were generally available back to 1990. All data collected were within seven Major Land Resource Areas (MLRA, USDA NRCS 2006): Snake River Plains, Owyhee Plateau, Central Nevada Basin and Range, Humboldt, Fallon-Lovelock, Malheur High Plateau and Great Salt Lake (Fig. 1, Table 1).

Table 1. Number of aerial and drill seeding sites sampled within the study area. Numbers in parentheses are standard errors.

Major Land Resource Area	Aerial	Drill	Mean Elevation (m)	30-yr Mean Annual Precipitation (cm)
Central Nevada Basin & Range	3	4	1861 (78)	29.6 (3.7)
Fallon-Lovelock	1	4	1518 (131)	23.5 (5.7)
Great Salt Lake	9	10	1593 (126)	32.0 (4.4)
Humboldt	3	9	1437 (92)	25.1 (3.0)
Malheur High Plateau	2	7	1393 (93)	25.7 (2.6)
Owyhee Plateau	7	13	1457 (183)	29.1 (4.0)
Snake River Plains	2	14	1183 (267)	27.9 (2.8)
Total number of sites	27	61	1449 (16)	28.4 (0.3)

Before sites were selected, all seedings within each MLRA were stratified by age since seeding (old 1990-1997 or young 1998-2004) and mean annual precipitation (2.5 cm intervals) within each age strata to ensure an adequate distribution across ages and precipitation zones. Aerial and drill seeding sites were randomly selected from each age/precipitation stratum and selected projects were screened by fire history and soil texture. We restricted sites to projects that occurred in locations with a single wildfire since 1970 to minimize potential confounding of repeated burn/seeding. We also restricted sites to loamy surface textures based on dominant soil map unit components (http://soildatamart.nrcs.usda.gov accessed 1 September 2009) since these soils are most likely to successfully establish seedlings and to represent a best-case scenario of ES&R seeding success. This selection process resulted in 100 sites where post-fire seedings were implemented from 1990 to 2003 (Fig. 1; see Table S1, Supporting Information). Of these, 27 were aerial and 61 were drill, seedings (Table 1). Twelve were combinations of aerial-over-drill (AOD) seedings (Table S1 Supporting Information) and were not evaluated for the purposes of this paper.

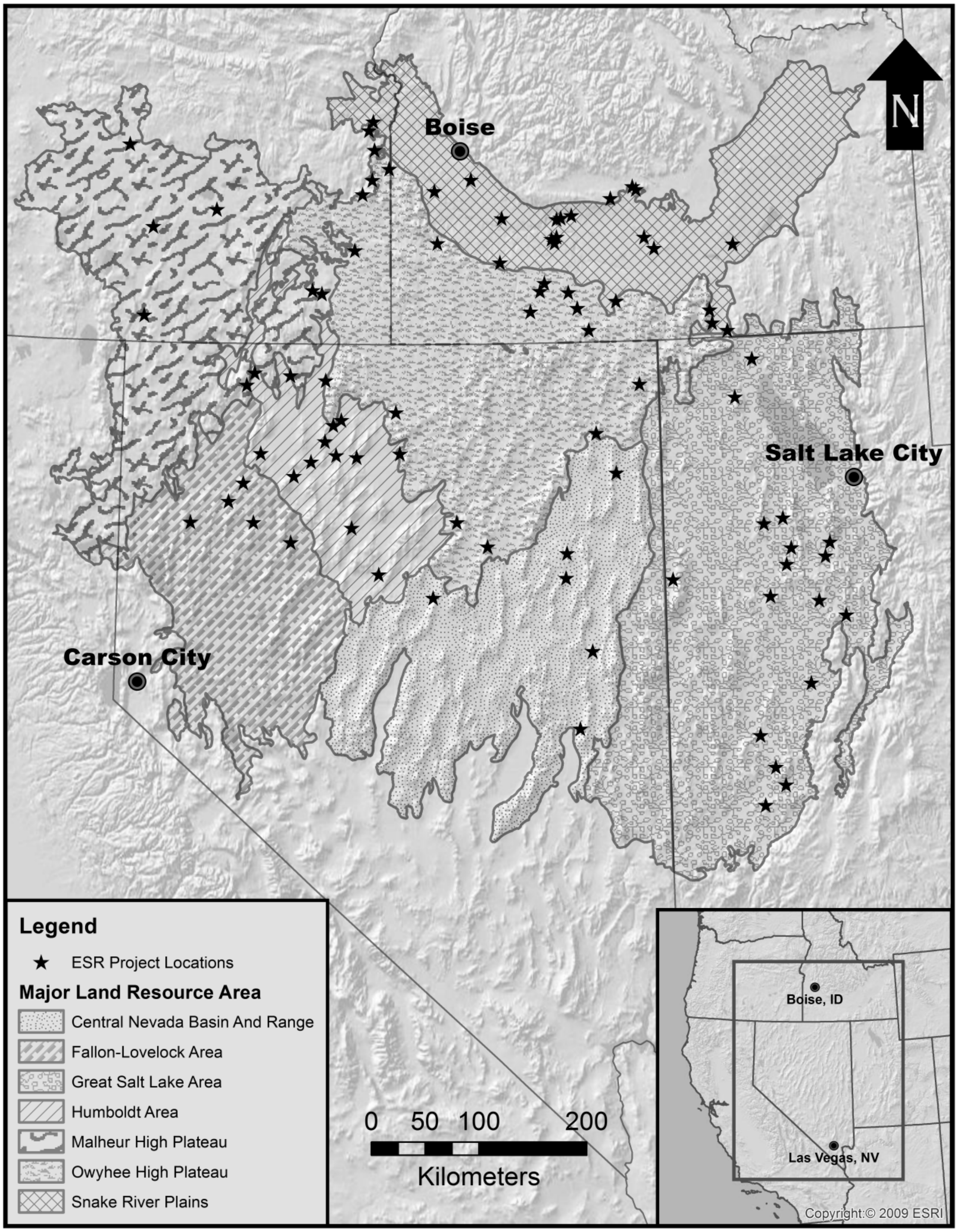

Fig. 1. Locations of ES&R post-fire seedings in Oregon, Idaho, Nevada, and Utah, USA.

Within each seeding site, burned-seeded (BS), burned-unseeded (BX), and unburned (UX) areas were mapped to delineate the three treatment levels at similar landscapes matched for soil map unit component, slope, aspect, and ecological site as defined by USDA NRCS (http://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/landuse/rangepasture/?cid=stelprdb1068392 accessed 17 September 2013). This matching was done to ensure that each treatment within a site had the potential to produce similar plant composition and biomass. Within each treatment area random point locations for subsampling plots were generated and sequentially visited by random draw to verify the treatment and within-site matching criteria until three random locations within each of the three treatments (BS, BX, UX) were identified where possible. Exceptions included three seeding sites that only had sufficient matched area to place two subsampling plots in BX treatments, and 28 seeding sites where only two of three matched treatment areas could be established (22 with no BX treatment and 6 with no UX treatment; see Table S1, Supporting Information).

All data were collected from April through August of 2011. Each subsampling plot comprised three 50-m transects in an equally-spaced spoke design. Percent cover of biotic and abiotic components were collected using line-point intercept at 1-m intervals along each transect (Herrick et al. 2005). Canopy gaps (distances between plant canopies) between perennial plants and between perennial or annual plants were taken along each transect as a measure of fuel continuity (Herrick et al. 2005). Shrub density by species was measured using 2- or 6-m by 50-m belt sampling transects where the wider belt was used at burned (BS, BX) plots and the narrower belt was used at unburned (UX) plots.

Fuel loads as litter and herbaceous biomass were sampled separately by destructive harvest in a 1.5-m^2 area at sampling plots (two 0.25-m^2 quadrats along each transect) . Shrub biomass was estimated using allometric equations developed for common species present at our sites. We created these allometric equations for 23 focal species by destructively harvesting a reference branch for each focal species found at an ES&R site. At each plot, allometric units based on the reference branch were then assigned to the focal species nearest to 6 incremental sampling points along each sampling transect. Reference branches were then dried and weighed, and the mass of each species was then determined from from the number of allometric units present at a plot. Shrub biomass (kg/ha) was modeled by multiplying shrub mass by shrub density. 10- and 100-hour dead and down woody fuels were sampled as intercepts along each transect using the planar intercept technique and equations for conversion of intercepts to biomass described in Brown (1974). These conversion calculations assumed a slope correction factor of 1.00 since our sites were relatively flat (Brown 1974, Table1), composite values of squared average diameters for nonslash fuels (Brown 1974, Table 2), and approximate specific gravity and angle correction factors for each size class as reported in Brown (1974, page 17). Total fuel load was the sum of litter, herbaceous, shrub, 10-hour, and 100-hour biomasses.

In addition to the study location and measurements listed above, the sage-grouse habitat study used the entire Great Basin portion of the sagebrush biome. In this study, empirical data on plot-level sage-grouse occupancy and habitat conditions were collected in 2006 at 211 plots that were randomly located on public land throughout the study area (Hanser and Knick 2011). At each of these 180 x 180 m plots, we measured the percent cover and height of plant species and abiotic habitat components (e.g., plant litter, rock, soil) using line-point intercept (LPI) on two parallel 50-m lines separated by 20 m. We recorded species or abiotic group intercepts at 0.5 m increments along transect lines (200 sampling points per plot). We used pellet surveys to identify plots that were used by sage-grouse (Boyce 1981, Hanser et al. 2011). Observers walked three parallel 120-m transect lines, which were connected by two, 36-m transects, and searched within 2 m of each transect line for a total search area of 864 m^2 per plot. If one or more sage-grouse pellets were found during this search, the plot was considered occupied by sage-grouse (Hanser and Knick 2011).

Landscape composition surrounding each sage-grouse occupancy and ES&R plot was quantified using Landfire Existing Vegetation Type data (LANDFIRE 2009, 2011) within a 5-km radius (78.5 km^2). This distance was selected based on a recommendation for the management of non-migratory sage-grouse populations (Connelly et al. 2000). Within each buffer, we calculated the proportion of 30-m pixels in each of 29 landcover types, which were reclassified from Landfire data.

Key Findings (All Findings are Preliminary Pending Final Peer-review)

H1: Cover of perennial lifeforms (grass and shrub) will not differ in seeded and unseeded plots regardless of average annual precipitation.

- Drill seedings had significantly more perennial plant cover than matching unseeded sites, but this relationship was dependent on average annual precipitation. Above about 28 cm (11 inches) drill seedings had higher perennial plant cover. Below this value, there was no significant difference between drill seeding and unseeded perennial plant cover; cover converged as precipitation declined.
    - If we tease out the contribution of shrubs and perennial grasses to this result, perennial grasses contribute most of this increase.
        - Perennial grasses contribute the greatest amount to the increase in cover with drill seedings. Above 33 cm (13 inches) average annual precipitation, the cover of perennial grasses increases exponentially from 10 to 20% as precipitation increases.
            - The increase is largely due to non-native perennial grasses, not to natives. If native perennial grasses were sown with non-native perennial grasses, then the native grass cover in drill seedings did not differ from their cover in unseeded areas. However, if sown without non-native perennial grasses, cover in drill seedings was twice that of unseeded areas (18 vs. 9%).
        - Native shrubs, including sagebrush (*Artemisia tridentata* ssp. *vaseyana* or *wyomingensis*) did not differ between drill seeded or unseeded areas when sagebrush was in the seed mixture.
        - When forage kochia (*Bassia prostrata*) was included in seed mixtures, it added between 0.7 to 1.5% cover over unseeded sites. The value increased with elevation, but was always higher than the unseeded areas.

- Aerial seedings had no significant effect in contributing additional perennial plant cover when nativity of the species was ignored, but forage kochia and non-native perennial grasses showed significant increases above the non-seeded levels. Native perennial grasses and sagebrush were not different from unseeded areas.
  - Non-native perennial grass cover on aerial seedings increased with increasing precipitation. Above 28 cm (11 inches) average annual precipitation, non-native perennial grasses provided significantly higher cover relative to unseeded areas and this increase continued exponentially with increasing precipitation to nearly 10% when precipitation averaged 35 cm (14 inches).
  - Forage kochia on aerial seeded areas contributed between 0.4 and 1.0% more cover on than unseeded areas and increased with increasing elevation.

Relationship to Recent Studies

This is the first study to assess the long-term effectiveness of ES&R treatments across a region. Meta-analyses and reviews have found mixed successes among these revegetation efforts (Pyke et al. 2013), but long-term precipitation and temperature have been indicated as major contributing factors to ecosystem resilience from disturbances and resistance from plant invasions in the Great Basin (Chambers et al. 2013, Miller et al. 2013). Our results support these findings by demonstrating that revegetation success follows climate related drivers of precipitation and temperature (elevation).

Aerial seedings were generally unsuccessful in contributing more perennial plant cover. Even though non-native perennial grasses and shrubs were establishing and contributing cover, they appeared to replace the cover that would have been contributed by pre-existing native perennial plants through natural recovery.

In drill seedings, however, successful increases in perennial plant cover depended on higher precipitation. The variation in establishment success seen in previous studies (Pyke et al. 2013) may relate to the likelihood of the site having adequate moisture for establishment to occur. More arid sites may require additional revegetation attempts to achieve successful establishment.

H2: Cover of exposed mineral soil will not differ in seeded and unseeded plots regardless of average annual precipitation.

- On aerial seedings bare mineral soil cover did not differ between seeded and unseeded areas (5%).
- On drill seedings, the amount of bare mineral soil was about 1.5 to 8% higher on seeded than unseeded areas with this amount increasing exponentially with elevation.

Relationship to Recent Studies

Although this seems counterintuitive, this result relates to reduction in cover of annual grasses when seedings with perennial plants are successful. This value might vary annually as annual grass cover varies with annual precipitation, but this increase is significantly *lower than the unburn-unseeded areas on the same soils, indicating that, as perennials become established, managers should anticipate some increase in bare mineral soil. Maintaining soil aggregate stability, a quantitative indicator of soil stability (Herrick et al.2001, Pyke et al. 2002), is important in protecting soil erosion. In addition, litter contribution from perennial grasses also protects soils from raindrop impacts and water erosion (Hester et al. 1997, Thurow et al. 1988a,b).*

H3: Invasive annual grass cover will not differ in seeded and unseeded plots regardless of average annual precipitation.

- Drill seedings had significantly lower cover of non-native annual *Bromus* sp. than unseeded areas, but this relationship depended on the elevation of the site.
  - Below about 1400 m (4600 ft), there was no difference in annual *Bromus* cover between drill-seeded and unseeded, burned areas. Above this level, drill-seeded areas had lower annual *Bromus* cover than unseeded areas and this cover declined more rapidly in drill-seeded than unseeded areas as elevations increased reaching cover of annual *Bromus* in drill-seeded areas that was nearly 1/2 the cover in unseeded areas.
- Aerial seedings had no significant effect on non-native annual *Bromus* cover compared with unseeded areas.
  - Cover of non-native annual *Bromus* declines significantly from about 80% at 20 cm (8 inches) to about 20% at 40 (16 inches).

Relationship to Recent Studies

Non-native annual Bromus *species are a primary threat to the maintenance of perennial shrub grasslands in the intermountain west of the USA because of the role they play in changing fire regimes and competing with desirable vegetation establishment (D'Antonio and Vitousek 1992; Knapp 1996; Mazzola et al. 2011; James et al. 2011). Therefore, a major goal of ES&R projects on federal land is to control invasive species or species that may threaten sensitive plants or animals (BLM 2007).*

Drill seeding of perennial plants does appear to reduce annual Bromus *cover and this relationship is improved (greater* Bromus *reduction) as elevations increase. This interaction between seeding and elevation on* Bromus *cover supports existing data showing that cooler, moister environments are more resistant to annual* Bromus (Chambers et al. 2013a,b). *However,it also shows that seeding with competitive non-native grasses likely decreases recovery of perennial native herbaceous species as indicted by decreased cover. This interaction may also explain the variation in effectiveness seen among research studies (Pyke et al. 2013).*

Aerial seeding was largely ineffective in reducing annual Bromus *cover beyond that reduction that had been predicted by cooler moister sites having more resistance to annual* Bromus (Chambers 2013). *This finding does support the lack of effectiveness of aerial* *seedings in reducing annual* Bromus *from the meta-analysis of seeding studies in the literature (Pyke et al. 2013).*

Effects of seeding after wildfire on fuel conditions in Great Basin shrub steppe

H4: Fuel loads will not differ in seeded and unseeded plots regardless of average annual precipitation.

- Drill seeding had no effect on any of fuel load (biomass) variables measured, including total fuel biomass, ratio of shrub:herbaceous biomass, shrub biomass, herbaceous biomass, litter biomass, 10-hour fuel biomass, 100-hour fuel biomass, or total fuel load. However, drill seedings (BS) did significantly reduce 10-hour fuel biomass compared to burned unseeded (BX) areas.
    - Precipitation had no interactive effect on drill seeding effects on fuel loads.
- Aerial seeding had no effect on any of the fuel load (biomass) variables measured, ratio of shrub:herbaceous biomass, shrub biomass, herbaceous biomass, litter biomass, 10-hour fuel biomass, 100-hour fuel biomass, or total fuel biomass.
    - Precipitation had no interactive effect on aerial seeding effects on fuel loads.
- There were some noteworthy differences between drill and aerial seeding plots regarding shrub:herbaceous and shrub biomass responses along the elevation gradient.
    - For aerial seedings, shrub biomass and ratio of shrub:herbaceous biomass both increased in burned areas with increasing elevation, and above 1750m these response variables were statistically the same in burned and unburned areas.
    - For drill seedings, there was no such convergence of burned and unburned areas.
- An opposite pattern was observed for the herbaceous biomass response along the elevation gradient.
    - For aerial seedings, there was no interactive effect of elevation.
    - For drill seedings, unburned and burned areas were statistically the same up to 1250m, but above that elevation herbaceous biomass diverged as unburned areas declined more precipitously than burned areas.
- Total fuel biomass response varied along the heat load gradient.
    - For aerial seeding, total fuel biomass was most affected by fire at the lower heat load index sites, but not at the highest HLI sites.
    - For drill seeding, no such interaction occurred.

Relationship to Recent Studies

H5: Continuity of fuels will not differ in seeded and unseeded plots regardless of average annual precipitation.

- Drill seeding had no effect on some of the fuel continuity variables, including annual forb cover, perennial forb cover, shrub cover, and vegetative interspace. However, other response variables were affected by drill seeding.

    - Total plant cover was decreased by drill seeding, but only at elevations above approximately 1500m. This seems to be have been driven by similar response patterns of annual grass cover and litter cover.
    - Perennial grass cover in BS plots increased at a faster rate with elevation than in BX plots, and diverged from the BX values at the highest elevation.
    - Annual interspace in both BX and BS plots were higher than that in UX plots at the lowest precipitation levels, but interspace values in BS plots converged with UX, whereas values in BX plots diverged from UX and BS plots, with increasing precipitation. This pattern suggests that drill seeding in BS plots led to similar annual interspace conditions as unburned UX plots at the highest precipitation sites.
- Aerial seeding had no effect on any of the fuel continuity variables measured, including the total plant cover, annual grass cover, annual forb cover, perennial forb cover, perennial grass cover, shrub cover, vegetative interspace cover, annual interspace cover, and perennial cover.
    - Precipitation had no interactive effect on aerial seeding effects on fuel continuity.

Relationship to Recent Studies

Quantifying restoration effectiveness using multi-scale habitat models: implications for sage-grouse in the Great Basin, USA

- Sage-grouse occupancy was positively related to plot- and landscape-level dwarf sagebrush (*Artemisia arbuscula, A. nova, A. tripartita*) and big sagebrush steppe prevalence, and negatively associated with non-native grass and human development.
    - The predicted probability of sage-grouse occupancy at burned and seeded plots was low on average (0.07–0.09) and was not significantly different from burned areas that had not been seeded.

- Restoration sites with high quality habitat tended to occur at higher elevation locations with low annual temperatures, high spring precipitation, and high plant diversity.

- Post-fire seeding treatments in Great Basin sagebrush shrublands generally have not created high quality habitat for sage-grouse. Of the plots seeded after fire, none met all sagebrush guidelines for breeding habitats.
    - Approximately 50% met understory guidelines, particularly for perennial grasses. This trend was similar for summer habitat. Ninety-eight percent of seeded plots did not meet winter habitat guidelines.

- Restoration actions did not increase the probability of burned areas meeting most guideline criteria.

- The probability of meeting guidelines was influenced by a latitudinal gradient, local climate, and topography.

Relationship to Recent Studies

Given the findings of recent research, this association with dwarf sagebrush should not be particularly surprising and may be attributed to at least two factors. First, contemporary studies spanning four western states and multiple seasons of sage-grouse habitat use have found that sage-grouse use dwarf sagebrush habitats disproportionately to their availability or more frequently than big sagebrush sites (Erickson et al. 2009, Atamian et al. 2010, Bruce et al. 2011, Hagen et al. 2011, Frye et al. 2013). This is likely because the leaves of dwarf sagebrush species have significantly lower monoterpene concentrations than those of Wyoming sagebrush (Frye et al. 2013). Second, dwarf sagebrush species are often associated with higher elevation sites with rocky soils or with wind-swept ridges, while adjacent stands of higher elevation big sagebrush often have relatively high forb and native grass cover. These community types may provide quality habitat with low susceptibility to invasion by non-native plant species because of climate and soil constraints on establishment (Chambers et al. 2013). Alternatively, because of fewer human impacts, these habitats may simply be the "best of what's left" for sage-grouse.

Habitat quality of seeded and unseeded ES&R locations is approximately equally limited (i.e., native species regeneration or treatment success) and landscape conditions, whereas habitat quality in mixed-treatment and unburned areas is more limited by landscape composition. Treatment technique limitations are currently being addressed through using seeds with local genotypes, low- or no-till rangeland drills, novel approaches for seed application (e.g., using imprinters for seeds that should not be buried, coating seeds prior to sowing) (Monsen et al. *2004, Shaw* et al. *2005, Madsen* et al. *2012a, Madsen* et al. *2012b), and planting seedlings (McAdoo* et al. *2013, Dettweiler-Robinson* et al. *2013). Landscape limitations on sage-grouse occupancy were not unexpected since ES&R sites are, by definition, in need of restoration action and they are often imbedded in disturbance-prone locations. However, as restoration actions proceed in the Great Basin, it is important to consider that a high quality plot embedded in a low quality landscape is still unlikely to be occupied by sage-grouse. Consequently, if sage-grouse habitat restoration is a primary goal, land managers may want to evaluate the probability of restoration success at a given site and the quality of the surrounding landscape (see previous section), potentially focusing restoration dollars on relatively intact landscapes (Meinke* et al. *2009), or implementing a triage-type strategy (Pyke 2011).*
Within 20 years of treatment, none of the treated plots met breeding season overstory (sagebrush) guidelines, few met brood-rearing overstory guidelines, and only 2% potentially meet winter overstory guidelines. Artemisia *spp. can be slow to reestablish dominance*

following disturbance, especially when seed sources are distant (Wambolt et al. *2001, Hemstrom* et al. *2002, Lesica* et al. *2007, Beck* et al. *2009). However, despite up to 20 years since burning and* Artemisia *spp. being sown at 62% of our sites,* Artemisia *spp. struggled to reestablish at all, let alone to reestablish dominance.*

In contrast to sagebrush restoration results, ES&R-treated areas met breeding and brood-rearing season perennial grass cover guidelines fairly often. Native perennial grass cover and height are important for hiding nests and young during these seasons (Connelly et al. *2011c). Despite the relative success of native perennial grass recovery, it is important to consider that establishment of these grasses does not necessarily preclude an abundance of invasive plants. For example, of the 554 plots that met Connelly* et al. *(2000) criterion for perennial grass and forb cover, cheatgrass cover averaged 36% and non-native annual forb cover averaged 14% (i.e., 50% total non-native annual plant cover). Our habitat association models indicated that these plots are unlikely to be occupied by sage-grouse.*

Few treated areas met Stiver et al. *(2010) forb canopy cover guideline, and this understory component limited the proportion of plots meeting this set of breeding season understory guidelines in all plot types (including unburned plots). Establishing native forbs is difficult because they are naturally sparse in many parts of the Great Basin, they often do not compete well with non-native plants, are difficult to procure, and can require specialized seeding application (Pyke 2011). Treated plots met all of Connelly* et al. *(2000) breeding season understory guidelines more frequently than did burned-untreated plots (indicating a positive treatment effect). A relatively high proportion of treated areas met brood-rearing understory habitat guidelines, but these plots often had high cheatgrass and non-native forb abundance. Surprisingly few unburned areas surrounding ES&R sites met understory habitat guidelines, especially for the breeding season. A lack of native perennial grass cover and height was the main cause of this trend.*

Management Implications

- At lower elevations or precipitation zones where potential for invasion by non-native annuals is severe, intensive methods of restoration (e.g., pre-treatment invasive plant control, irrigation) with the potential for multiple interventions may be required to effectively establish seeded species. If multiple interventions are not feasible then it is reasonable to consider not seeding if soil stabilization is not an issue.
- ES&R project managers might consider giving higher priority to seeding locations at higher elevations with higher precipitation where the likelihood of meeting ES&R goals of reducing annual non-native grasses, attaining higher perennial plant cover and meeting sage-grouse quality habitat needs are improved. However, it is important to recognize that seeding with introduced species may prevent recovery of native species on cooler and moister sites, and if adequate cover of perennial native grasses exists post-fire then seeding is probably not necessary.
- Mixing native with non-native perennial plants together in the same project or at least in the same seed row may be counterproductive. Further work is needed on this topic to

determine seeding rates, species mixes, and methods that would provide for greater establishment of native perennial plants when seeded with non-native perennial plants.

Future Studies Needed

- Determine the cover percentages of native perennial herbaceous plants that provide for natural site recovery without ES&R seeding and how these differ over temperature/precipitation gradients.
- Determine if optimal seeding rates, mixtures, and methods exist for establishing both native and non-native perennial plants in areas where inadequate perennial herbaceous species exist for site recovery.
    - These approaches should be allow establishment and survival of both the native and non-native plants
    - Studies should determine if mixtures achieve the same % vegetation cover as their monospecific counterparts.
    - Studies should determine if different mixtures have the same impact on non-native invasive species.
- Evaluate if the cover percentage of perennial plants on ES&R projects is the same as on sites with varying numbers of fires?
    - We restricted our study to locations that had only one fire in the recorded past. Do multiple fires change this relationship?
- Examine how the degree of tree encroachment changes ES&R success relationships?
    - We intentionally restricted our work to areas where tree encroachment was not a factor. Sagebrush ecosystems with tree encroachment tend to result in higher severity burns with burn severity increasing as infilling progresses and woody biomass increases.
    - Since one of the management implications from this study is that higher elevation and higher precipitation environments tended to be more successful at meeting ES&R goals, would these relationships change in areas of tree encroachment and would the cover of trees influence this relationship?

Deliverables Crosswalk Table

Proposed	Delivered	Status
Project Webpage	http://fresc.usgs.gov/sites/FRE/FireRehabilitationEffectiveness.aspx	Update as needed
Project Factsheet	Refer people to the webpage	Webpage provided same information
Final Report	Uploaded to JFSP Website	Update as needed pending peer review comments
Datasets		
Field site information	Land Treatment Digital Library for all ES&R file information; http://ltdl.wr.usgs.gov/	Posted and available
Metadata & sampled data	Microsoft Access database file will be posted on SageMap website upon publication acceptance	MS Access database available from D.A. Pyke upon request; Will make public upon acceptance of the three papers in review
Referreed Publications		
1. Vegetation Composition Response after ES&R Treatments	Pyke et al. Longterm effects of seeding after wildfire on vegetation composition in Great Basin shrub steppe	Complete: Submitted Journal of Applied Ecology
2. Fuel Response after ES&R seedings	Brooks et al. Effects of seeding after wildfire on fuel conditions in Great Basin shrub steppe	Draft
3. Sage-grouse habitat quality of ES&R seedings	Arkle et al. Quantifying restoration effectiveness using multi-scale habitat models: implications for sage-grouse in the Great Basin, USA	Complete: Submitted EcoSphere

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Literature Cited

Atamian, M. T., J. S. Sedinger, J. S. Heaton, and E. J. Blomberg. 2010. Landscape-level assessment of brood rearing habitat for Greater Sage-grouse in Nevada. The Journal of Wildlife Management 74:1533-1543.

Beck, J. L., J. W. Connelly, and K. P. Reese. 2009. Recovery of Greater Sage-grouse habitat features in Wyoming big sagebrush following prescribed fire. Restoration Ecology 17:393-403.

Beyers, J. L. 2004. Postfire seeding for erosion control: effectiveness and impacts on native plant communities. Conservation Biology 18**:**947–956.

Boyce, M. S. 1981. Robust canonical correlation of sage grouse habitat. Pages 152-159 *in* D. Capen, editor. The use of multivariate statistics in studies of wildlife habitat. USDA Forest Service General Technical Report RM-87. USDA Forest Service, Rocky Mountain Research Station, Ft. Collins, CO.

Bruce, J. R., W. D. Robinson, S. L. Petersen, and R. F. Miller. 2011. Greater Sage-grouse movements and habitat use during winter in central Oregon. Western North American Naturalist 71:418-424.

Brown, J.K. 1974. Handbook for inventorying downed woody material. US Department of Agriculture, Forest Service, Intermountain Research Station, General Technical Report GTR-INT-16, Fort Collins, CO. 24 p.

Brooks, M.L., C.M. D'Antonio, D.M. Richardson, J.B. Grace, J.E. Keeley, J.M. DiTomaso, R.J. Hobbs, M. Pellant and D. Pyke. 2004. Effects of invasive alien plants on fire regimes. BioScience 54:677-688.

Chambers, J.C., Roundy, B.A., Blank, R.R., Meyer, S.E. and Whittaker, A. 2007. What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? Ecological Monographs 77:117-145.

Chambers, J.C., B.A. Bradley, C.S. Brown, C. D'Antonio, M.J. Germino, J.B. Grace, S.P. Hardegree, R.F. Miller, and D.A. Pyke. 2013a. Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. invasion in cold desert shrublands of western North America. Ecosystems DOI: 10.1007/s10021-013-9725-5

Chambers, J.C., R. F. Miller, D. I. Board, J. B. Grace, D. A. Pyke, B. A. Roundy, E. W. Schupp, and R. J. Tausch. 2013b. Resilience and resistance of sagebrush ecosystems: implications for state and transition models and management treatments. Rangeland Ecology and Management. *in press*.

Connelly, J. W., E. T. Rinkes, and C. E. Braun. 2011. Characteristics of Greater Sage- grouse habitats: a landscape species at micro and macro scales. Pages 69-84 *in* S. T. Knick and C. J.

W., editors. Greater Sage-Grouse: ecology of a landscape species and its habitats. University of California Press, Berkeley, California.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. Wildlife Society Bulletin 28:967-985.

D'Antonio C. M., and P. M. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Review of Ecology and Systematics 23:63-87.

Davies, K. W. 2010. Revegetation of medusahead-invaded sagebrush steppe. Rangeland Ecology and Management 63:564-571.

Dettweiler-Robertson, E., J.D. Bakker, J.R. Evans, H. Newsome, G.M. Davies, T.A. Wirth, D.A. Pyke, R.T. Easterly, D. Salstrom, and P.W. Dunwiddie. 2013. Outplanting Wyoming big sagebrush following wildfire: stock performance and economics. Rangeland Ecology and Management 66:657-666

Erickson, H. J., C. L. Aldridge, and N. T. Hobbs. 2009. Progress Report: Stratton Ecological Research Site—An Experimental Approach to Assess Effects of Various Grazing Treatments on Vegetation and Wildlife Communities Across Managed Burns and Habitat Controls. US Geol. Survey Report 1016:1-15.

Frye, G. G., J. W. Connelly, D. D. Musil, and J. S. Forbey. 2013. Phytochemistry predicts habitat selection by an avian herbivore at multiple spatial scales. Ecology 94:308-314.

Gray, E.C., and P.S. Muir. 2013. Does Kochia prostrata spread from seeded sites? An evaluation from southwestern Idaho, USA. Rangeland Ecology and Management 66:191-203.

Hagen, C. A., M. J. Willis, E. M. Glenn, and R. G. Anthony. 2011. Habitat selection by greater sage-grouse during winter in southeastern Oregon. Western North American Naturalist 71:529-538.

Hanser, S. E., C. L. Aldridge, M. Leu, M. M. Rowland, S. E. Nielsen, and S. T. Knick. 2011. Greater sage-grouse: general use and roost site occurrence with pellet counts as a measure of relative abundance.*in* S. E. Hanser, M. Leu, S. T. Knick, and C. L. Aldridge, editors. Sagebrush Ecosystem Conservation and Management: Ecoregional Assessment Tools and Models for the Wyoming Basins. Allen Press, Lawrence, KS.

Hanser, S. E. and S. T. Knick. 2011. Greater sage-grouse as an umbrella species for shrubland passerine birds: a multiscale assessment. Pages 475-488 *in* S. T. Knick and J. W. Connelly, editors. Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. University of California Press, Berkely, CA.

Hemstrom, M. A., M. J. Wisdom, W. J. Hann, M. M. Rowland, B. C. Wales, and R. A. Gravenmier. 2002. Sagebrush-steppe vegetation dynamics and restoration potential in the interior Columbia Basin, USA. Conservation Biology 16:1243-1255.

Herrick, J.E., W.G. Whitford, A.G. de Soyza, J.W. Van Zee, K.M. Havstad, C.A. Seybold, and M. Walton. 2001. Soil aggregate stability kit for field-based soil quality and rangeland health evaluations. CATENA 44: 27-35.

Herrick, J.E., Van Zee, J.W., Havstad, K.M., Burkett, L.M. & Whitford, W.G. 2005. Monitoring manual for grassland, shrubland and savanna ecosystems, Volume 1: Quick Start. USDA – ARS Jornada Experimental Range: Las Cruces, New Mexico.

Hester, J.W., T.L. Thurow, and C.A. Taylor Jr., 1997. Hydrologic characteristics of vegetation types as affected by prescribed burning. Journal of Range Management 50:199-204.

James J.J., R.E. Drenovsky, T. A. Monaco, and M.J. Rinella. 2011. Managing soil nitrogen to restore annual grass infested plant communities: effective strategy or incomplete framework? Ecological Applications 21:490-502

Knapp, P. A. 1996. Cheatgass (*Bromus tectorum*) dominance in the Great Basin Desert. Global Environmental Change 6:37-52.

LANDFIRE. 2009. LANDFIRE 1.0.0 Existing Vegetation Type layer. U.S. Department of the Interior, Geological Survey.

LANDFIRE. 2011. LANDFIRE 1.1.0 Existing Vegetation Type layer. U.S. Department of the Interior, Geological Survey.

Lesica, P., S. V. Cooper, and G. Kudray. 2007. Recovery of big sagebrush following fire in southwest Montana. Rangeland Ecology and Management 60:261-269.

Madsen, M. D., K. W. Davies, C. J. Williams, and T. J. Svejcar. 2012a. Agglomerating seeds to enhance native seedling emergence and growth. Journal of Applied Ecology 49:431-438.

Madsen, M. D., S. J. Kostka, A. L. Inouye, and D. L. Zvirzdin. 2012b. Postfire restoration of soil hydrology and wildland vegetation using surfactant seed coating technology. Rangeland Ecology & Management 65:253-259.

Mazzola, M.B., Chambers, J.C., Blank, R.R., Pyke, D.A., Schupp, E., Allcock, K.G., Doescher, P.S., and Nowak, R.S. 2011. Effects of resource availability and propagule supply on native species recruitment in sagebrush ecosystems invaded by *Bromus tectorum*. Biological Invasions 13:513-526.

McAdoo, J. K., C. S. Boyd, and R. L. Sheley. 2013. Site, competition, and plant stock influence transplant success of Wyoming big sagebrush. Rangeland Ecology & Management 66:305-312.

Meinke, C. W., S. T. Knick, and D. A. Pyke. 2009. A spatial model to prioritize sagebrush landscapes in the intermountain west (USA) for restoration. Restoration Ecology 17:652-659.

Miller, R. F., J. C. Chambers, D. A. Pyke, F. B. Pierson, and C. J. Williams. 2013. Fire effects on vegetation and soils in the Great Basin region: response and site characteristics. USDA Forest Service, Rocky Mountain Research Station, Research Paper RMRS-GTR-308. 126 p.

Monsen, S. B., R. Stevens, and N. L. Shaw, compilers. 2004. Restoring western ranges and wildlands. Gen. Tech. Rep. RMRS-GTR-136-1, USDA, Forest Service, Fort Collins, CO.

Noss, R.F., E.T. LaRoe III, and J.M. Scott. 1995. Endangered ecosystems of the United States: A preliminary assessment of loss and degradation. US Department of the Interior, National Biological Service. Biological Report 28. Washington DC. 58 p.

Pilliod, D.S., and Welty, J.L., 2013, Land Treatment Digital Library: U.S. Geological Survey Data Series 806, http://dx.doi.org/10.3133/ds806.

Pyke, D. A. 2011. Restoring and rehabilitating sagebrush habitats. Pages 531-548 *in* Knick, S.T and Connelly, J.W. (editors). Graeter Sage-Grouse: ecology and conservaation of a landscape species and its habitats. Studies in Avian Biology (vol. 38), University of California Press, Berkeley, CA.

Pyke, D.A., T.A. Wirth, and J.L. Beyers. 2013. Does seeding after wildfires in rangelands reduce erosion or invasive species? Restoration Ecology. doi: 10.1111/rec.12021.

Pyke, D.A., J.E. Herrick, P. Shaver & M. Pellant. 2002. Rangeland health attributes and indicators for qualitative assessment. Journal of Range Management 55:584-597.

Reisner, M.D., Grace, J.B., Pyke, D.A. & Doescher, P.S. 2013. Conditions favouring *Bromus tectorum* dominance of endangered sagebrush steppe ecosystems. Journal of Applied Ecology, 50:1039-1049.

Robichaud, P. R., J. L. Beyers, and D. G. Neary. 2000. Evaluating the effectiveness of postfire rehabilitation treatments. RMRS-GTR-63. U.S. Department of Agriculture, Forest Service, Fort Collins, Colorado.

Sankey, J.B., M.J. Germino, and N.F. Glenn. 2009. Aeolian sediment transport following wildfire in sagebrush steppe. Journal of Arid Environments 73:912-919.

Sankey, J.B., C.S.A. Wallace, and S. Ravi. 2013. Phenology-based, remote sensing of post-burn disturbance windows in rangelands. Ecological Indicators 30:35-44.

Scott, J. H., and R.E. Burgan. 2005. Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. Gen. Tech. Rep. RMRS-GTR-153. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO. 72 p.

Shaw, N. L., A. M. DeBolt, and R. Rosentreter. 2005. Reseeding big sagebrush: techniques and issues. Pages 99-108 *in* NL Shaw, SB Monsen, and M. Pellant, comps. Sage-grouse Habitat Improvement Symposium: Proceedings. Fort Collins, CO: US Department of Agriculture, Forest Service, Rocky Mountain Research Station Proc. RMRS-P-38.

Stiver, S. J., E. T. Rinkes, and D. E. Naugle. 2010. Sage-grouse habitat assessment framework: multi-scale habitat assessment tool. U.S. Bureau of Land Management, Idaho State Office, Boise, Idaho.

Thurow, T.L., W.H. Blackburn, and C.A. Taylor, Jr. 1988a. Infiltration and interrill erosion responses to selected livestock grazing strategies, Edwards Plateau, Texas. Journal of Range Management 41:296-302.

Thurow, T.L., W.H. Blackburn, and C.A. Taylor, Jr. 1988b. Some vegetation responses to selected livestock grazing strategies, Edwards Plateau, Texas. Journal of Range Management 41:108-114.

United States Department of Agriculture, Natural Resources Conservation Service (USDA NRCS). 2006. Land resource regions and major land resource areas of the United States, the Caribbean, and the Pacific Basin. USDA Handbook 296.

United States Department of the Interior, Bureau of Land Management (USDI BLM). 2007. Burned area emergency stabilization and rehabilitation handbook. BLM Handbook H-1742-1.

Wambolt, C. L., K. S. Walhof, and M. R. Frisina. 2001. Recovery of big sagebrush communities after burning in south- western Montana. Journal of Environmental Management 61:243-252.

Supporting Information

Table S1. Locations and features of ESR projects, including number of subsamples within different treatments sampled, year of wildfire/seeding, site mean elevation, 30-year mean annual precipitation (1971-2000), and soil surface texture.

Fire Name*	Fire Code*	MLRA†	State	Latitude (N)	Longitude (W)	Seeding Type	No. of subsamples BS	BX	UX	Year	Elevation (m)	Precipitation‡ (cm)	Soil Texture (A Horizon)
Atkins Butte	N258	OP	Oregon	43.40	-117.16	Drill	3	3	3	2003	1189.0	32.2	Clay Loam
Basque	N252	MHP	Oregon	42.46	-117.89	Drill	3	-	3	2002	1308.0	22.9	Loam
Bell Mare	F584	SRP	Idaho	43.05	-115.12	AOD	3	2	3	2000	1046.0	26.5	Clay-Silt Loam
Big Crow	FA16	OP	Idaho	42.51	-115.28	Aerial	3	3	3	2002	1372.0	23.9	Silt Loam
Big Spring	X022	CNV	Nevada	40.91	-114.54	AOD	3	-	3	2000	1813.7	25.7	Silty Clay Loam
Big Spring	X022	CNV	Nevada	40.90	-114.54	Drill	3	-	3	2000	1824.3	26.2	Silty Clay Loam
Black Mesa	F062	SRP	Idaho	42.89	-115.15	AOD	3	3	3	1995	926.3	25.2	Silt Loam
Black Mesa	F325	SRP	Idaho	42.87	-115.19	Drill	3	3	3	1999	907.0	25.4	Loam - Silt Loam
Black Rock	R521	GSL	Utah	38.67	-113.08	Drill	3	3	-	1994	1518.7	24.9	Sandy Clay Loam
Bloody Run	J489	HA	Nevada	41.26	-117.71	Drill	3	-	3	1996	1426.0	25.1	Very Fine Sandy Loam
Blue Gulch	F070	OP	Idaho	42.42	-114.95	AOD	3	3	3	1995	1325.7	26.6	Silt Loam
Buck and Doe	F339	OP	Idaho	42.41	-115.33	Drill	3	3	3	1999	1418.7	24.0	Silt Loam
Buffalo	X393	HA	Nevada	40.46	-117.44	Aerial	3	-	3	1995	1631.0	30.8	Very Fine Sandy Loam
Buffalo	X393	HA	Nevada	40.48	-117.44	Drill	3	3	3	1995	1638.3	30.8	Very Fine Sandy Loam
Butte	K267	CNV	Nevada	40.04	-115.11	Drill	3	3	3	2001	1955.0	32.2	Sandy Clay Loam
Cain	K921	HA	Nevada	40.08	-117.15	Drill	3	3	-	1999	1513.7	22.7	Loam
Calf Creek	G303	SRP	Idaho	43.06	-114.96	Drill	3	3	3	1997	1199.0	27.9	Silt Loam
Castle Creek	F052	OP	Idaho	42.83	-116.51	Aerial	3	3	3	1990	1781.3	42.6	Coarse Sandy Loam
Cherry Creek	M720	OP	Oregon	43.62	-117.21	Drill	3	3	3	2003	1211.3	28.5	Silt Loam
Chimney	J527	OP	Nevada	41.43	-117.04	Aerial	3	3	3	1999	1723.3	32.9	Silty Clay Loam
Cinder Butte	N576	MHP	Oregon	43.63	-120.02	Drill	3	3	3	1995	1308.7	25.3	Sandy Loam
Clover	J185	OP	Nevada	41.01	-116.93	Aerial	3	3	3	1999	1513.7	26.8	Sandy Loam
Clover	J185	OP	Nevada	41.12	-116.83	Drill	3	3	3	1999	1617.7	29.9	Loam
Cold Spring	R021	GSL	Utah	40.00	-113.96	Drill	3	-	3	1991	1707.3	28.4	Silty Clay Loam
Cottonwood	F441	SRP	Idaho	42.13	-113.43	AOD	3	3	3	1999	1621.0	30.9	Silty Clay Loam

*Fire name and code are the assigned name/code at the time of wildfire by BLM personnel.
†MLRA = major land resource area the ESR project was sampled from: CNV = Central Nevada Basin and Range; FL = Fallon-Lovelock; GSL = Great Salt Lake; HA = Humboldt Area;

MHP = Malheur High Plateau; OP = Owyhee High Plateau; SRP = Snake River Plains.
‡Mean annual precipitation (cm) of the site from 1971-2000 derived from PRISM (2010).

Table S1 (Continued). Locations and features of ESR projects, including number of subsamples within different treatments sampled, year of wildfire/seeding, site mean elevation, 30-year mean annual precipitation (1971-2000), and soil surface texture.

Fire Name*	Fire Code*	MLRA†	State	Latitude (N)	Longitude (W)	Seeding Type	No. of subsamples BS	BX	UX	Year	Elevation (m)	Precipitation‡ (cm)	Soil Texture (A Horizon)
Cow Creek	X381	FL	Nevada	40.66	-118.78	Drill	3	3	3	2000	1498.7	20.2	Loam
Cow Hollow	M754	SRP	Oregon	43.78	-117.24	Drill	3	3	3	1996	941.3	29.3	Loam
Cow Hollow	N107	SRP	Idaho	43.86	-117.19	Drill	3	-	3	2001	892.0	28.2	Sandy Loam
Crump	M019	MHP	Oregon	42.20	-119.80	Drill	3	3	3	1999	1538.3	30.7	Loam
Davis Knoll	R157	GSL	Utah	40.25	-112.64	Drill	3	3	3	1996	1529.7	33.5	Loam
Davis Mountain	R122	GSL	Utah	40.07	-112.70	Drill	3	3	3	1994	1580.7	31.4	Loam
Denio	J520	MHP	Nevada	41.78	-118.57	Drill	3	3	3	1999	1409.7	25.1	Sandy Loam
Divison	J106	OP	Nevada	41.66	-114.21	Aerial	3	3	-	1996	1803.7	30.0	Sandy Loam
East Slick	Z269	SRP	Idaho	42.89	-115.27	AOD	3	3	3	1999	863.7	25.0	Silt Loam
East Slick	Z269	SRP	Idaho	42.86	-115.25	Drill	3	3	3	1999	893.0	25.0	Silt Loam
Eight Mile	Q161	GSL	Utah	40.44	-112.94	Aerial	3	3	3	2001	1595.3	36.2	Sandy Loam
Faust	Q077	GSL	Utah	40.27	-112.24	Drill	3	3	3	1998	1651.3	43.7	Loam
Flowell	R567	GSL	Utah	39.09	-112.52	Aerial	3	3	3	1996	1456.0	27.3	Sandy Loam
Frenchie Flat	J194	OP	Nevada	40.50	-116.27	Drill	3	3	3	1996	1615.0	27.1	Loam
Goat	G434	SRP	Idaho	43.02	-115.13	AOD	3	3	3	1998	942.7	25.5	Sandy Loam
Guff	F345	SRP	Idaho	43.28	-116.53	Aerial	3	3	-	2002	902.3	23.9	Silt Loam
Heusser	K114	CNV	Nevada	39.42	-114.84	Aerial	3	-	3	2001	1900.0	22.7	Sandy Clay Loam
High point	G198	SRP	Idaho	42.75	-114.04	Drill	3	3	3	2000	1328.3	26.8	Silty Clay Loam
Hogup	Q025	GSL	Utah	41.52	-113.20	Aerial	3	3	3	2000	1438.3	27.5	Silty Clay Loam
Island Ranch	R147	GSL	Utah	40.48	-112.73	Drill	3	3	-	1994	1411.7	35.6	Loam
Jack Mountain	M648	MHP	Oregon	43.11	-119.00	Drill	3	-	3	1999	1406.3	29.1	Clay Loam
Jackson	J521	HA	Nevada	41.08	-118.46	Drill	3	3	3	1999	1374.7	21.8	Loam
Junction	J458	OP	Nevada	41.32	-117.64	Drill	3	3	3	1999	1385.7	24.0	Silt Loam
Jungo Complex	X379	HA	Nevada	41.03	-117.89	Aerial	3	3	3	2000	1402.7	25.4	Loam
Jungo Complex	X379	HA	Nevada	41.01	-117.89	Drill	3	-	3	2000	1420.7	25.8	Sandy Loam

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†MLRA = major land resource area the ESR project was sampled from: CNV = Central Nevada Basin and Range; FL = Fallon-Lovelock; GSL = Great Salt Lake; HA = Humboldt Area; MHP = Malheur High Plateau; OP = Owyhee High Plateau; SRP = Snake River Plains.

‡Mean annual precipitation (cm) of the site from 1971-2000 derived from PRISM (2010).

Table S1 (Continued). Locations and features of ESR projects, including number of subsamples within different treatments sampled, year of wildfire/seeding, site mean elevation, 30-year mean annual precipitation (1971-2000), and soil surface texture.

Fire Name*	Fire Code*	MLRA†	State	Latitude (N)	Longitude (W)	Seeding Type	No. of subsamples BS	BX	UX	Year	Elevation (m)	Precipitation‡ (cm)	Soil Texture (A Horizon)
Juniper Complex	M200	MHP	Oregon	42.93	-119.61	Aerial	3	3	3	2002	1508.0	27.7	Clay Loam
Kane Creek	G175	SRP	Idaho	42.24	-113.43	Drill	3	3	3	2000	1549.7	32.4	Silty Clay Loam
Keg Mountain	Q989	GSL	Utah	39.85	-112.89	Drill	3	3	3	2001	1546.0	29.6	Sandy Loam
Keystone	X378	HA	Nevada	40.88	-118.10	Aerial	3	3	3	2000	1427.3	24.1	Loam
King	X465	MHP	Nevada	41.72	-118.14	Aerial	3	3	3	1995	1295.0	24.5	Silt Loam
Kumiva	J428	FL	Nevada	40.50	-119.22	Drill	3	3	3	2001	1505.3	23.7	Sandy Loam
Lambert	J423	HA	Nevada	41.05	-117.67	Drill	3	-	3	2001	1346.3	22.7	Sandy Loam
Leamington	Q613	GSL	Utah	39.61	-112.09	Aerial	3	3	3	1996	1679.0	35.4	Loam
Lone Butte	J530	HA	Nevada	41.03	-117.47	Drill	3	3	3	1999	1386.0	22.4	Sandy Loam
Mahogany Mountain	N255	OP	Oregon	43.24	-117.36	Drill	3	-	3	2002	1302.3	35.4	Clay Loam
Mallard Lake	F494	SRP	Idaho	42.87	-114.14	AOD	3	3	3	1999	1270.0	26.3	Loam
Mallard Lake	F494	SRP	Idaho	42.87	-114.14	Drill	3	3	3	1999	1267.3	26.2	Sandy Clay Loam
Marshall Well	R384	GSL	Utah	38.08	-113.06	Aerial	3	-	3	1994	1959.7	36.0	Sandy Clay Loam
Milford Bench	R372	GSL	Utah	38.39	-112.93	Drill	3	3	3	1994	1640.7	31.1	Sandy Loam
Minersville	R318	GSL	Utah	38.22	-112.85	Aerial	3	-	3	1998	1701.7	33.8	Sandy Loam
Minersville	R318	GSL	Utah	38.27	-112.80	Drill	3	-	3	1998	1742.7	33.2	Sandy Loam
North Can	F174	OP	Idaho	42.10	-114.79	Drill	3	-	3	2001	1654.7	29.5	Loam
Overshoe Well	N238	MHP	Oregon	42.42	-117.79	Drill	3	3	3	2002	1451.3	24.6	Silt Loam
Pass Creek	X478	MHP	Nevada	41.63	-118.59	Drill	3	3	3	1999	1289.0	22.8	Sandy Loam
Pigtail Butte	F074	OP	Idaho	42.29	-114.92	Drill	3	3	3	1990	1549.3	29.5	Silt Loam
Pinto Horse	N237	OP	Oregon	42.79	-117.46	Aerial	3	3	3	2002	1278.3	28.1	Silt Loam
Pinto Horse	N237	OP	Oregon	42.79	-117.45	Drill	3	3	3	2002	1310.7	28.6	Silt Loam
Poison Creek	F191	OP	Idaho	42.26	-115.51	AOD	3	3	3	1996	1501.7	27.7	Silt Loam
Poison Creek	F191	OP	Idaho	42.25	-115.49	Drill	3	3	3	1996	1519.0	28.1	Silt Loam
Poker Brown	J517	FL	Nevada	40.40	-118.80	Drill	3	3	3	1999	1324.0	19.9	Sandy Loam

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‡Mean annual precipitation (cm) of the site from 1971-2000 derived from PRISM (2010).

Table S1 (Continued). Locations and features of ESR projects, including number of subsamples within different treatments sampled, year of wildfire/seeding, site mean elevation, 30-year mean annual precipitation (1971-2000), and soil surface texture.

Fire Name*	Fire Code*	MLRA†	State	Latitude (N)	Longitude (W)	Seeding Type	No. of subsamples BS	BX	UX	Year	Elevation (m)	Precipitation‡ (cm)	Soil Texture (A Horizon)
Rabbit	X075	CNV	Nevada	40.27	-115.07	AOD	3	3	3	2000	1930.7	32.6	Sandy Loam
Rabbit	X075	CNV	Nevada	40.28	-115.07	Drill	3	3	3	2000	1920.0	31.8	Sandy Loam
Rattlesnake	F209	SRP	Idaho	43.05	-115.76	Drill	3	3	3	1996	933.0	23.7	Silt Loam
Rochester	J514	FL	Nevada	40.34	-118.14	Drill	3	3	-	1999	1717.7	35.9	Sandy Loam
Rosebud	J510	FL	Nevada	40.82	-118.64	Aerial	3	3	3	1999	1625.7	21.2	Silty Clay Loam
Round Mountain	G174	SRP	Idaho	42.06	-113.25	Drill	3	-	3	1996	1536.0	28.7	Silty Clay Loam
RRMP	F116	SRP	Idaho	43.36	-116.10	Drill	3	3	3	1991	962.0	30.7	Silt Loam
Saddler	J244	OP	Nevada	40.20	-116.15	Drill	3	2	3	1999	1669.7	28.0	Silt Loam
Sixmile	K177	CNV	Nevada	38.78	-114.99	Aerial	3	2	3	2001	1799.0	29.8	Sandy Loam
South Cricket	X039	CNV	Nevada	41.16	-114.84	Drill	3	3	3	2000	1865.3	33.5	Loam
Squaw Joe	F555	OP	Idaho	42.35	-114.48	AOD	3	-	3	1994	1431.0	27.6	Loam
Thorn Creek	G191	SRP	Idaho	43.14	-114.48	Drill	3	-	3	1990	1521.0	32.2	Loam
Timmerman	F463	SRP	Idaho	43.28	-114.29	AOD	3	-	3	2003	1517.7	32.2	Silty Clay Loam
Topliff	Q106	GSL	Utah	40.12	-112.33	Aerial	3	3	3	1998	1647.0	29.1	Loam
Topliff	Q106	GSL	Utah	40.15	-112.26	Drill	3	3	3	1998	1575.7	30.7	Sandy Loam
Trail Canyon	K909	CNV	Nevada	39.72	-116.69	Aerial	3	3	3	1999	1767.0	26.8	Sandy Loam
Trimbly	N245	OP	Idaho	43.47	-116.98	Aerial	3	3	3	2002	1202.7	30.8	Loam
Trimbly	N245	OP	Idaho	43.46	-116.97	Drill	3	3	3	2002	1198.3	30.7	Clay Loam
Wapi	F480	SRP	Idaho	42.77	-113.15	Drill	3	3	3	1999	1374.3	28.8	Silty Clay Loam
Wash O Neil	X428	HA	Nevada	41.37	-117.58	Drill	3	-	3	1995	1364.3	25.5	Loam
Wedge Butte	F390	SRP	Idaho	43.26	-114.28	Aerial	3	3	3	1999	1509.3	31.8	Silt Loam
Wedge Butte	F390	SRP	Idaho	43.25	-114.19	Drill	3	3	3	1999	1482.7	32.2	Silt Loam
West Rockwell	R492	GSL	Utah	39.77	-112.38	Aerial	3	3	3	1999	1695.0	33.1	Sandy Loam
Wildcat	R028	GSL	Utah	41.80	-113.02	Aerial	3	3	3	1999	1361.7	26.8	Silt Loam
Willow Creek	X357	HA	Nevada	41.71	-117.75	Drill	3	-	3	2000	1392.3	27.6	Sandy Loam

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‡Mean annual precipitation (cm) of the site from 1971-2000 derived from PRISM (2010).

Table S2. Summary of grass species seeded and the percentage of study sites seeded with each species or genus.

Seeded genus/species scientific name	Life-form type*	Nativity	% of total burn-seeded (BS) study sites seeded with each species: Aerial	Drill
Achnatherum hymenoides (Roemer & J.A. Schultes) Barkworth	DRPG	native	22.2	19.7
Elymus elymoides (Raf.) Swezey	DRPG	native	11.1	3.3
Elymus lanceolatus (Scribn. & J.G. Sm.) Gould	DRPG	native	7.4	1.6
Elymus lanceolatus ssp. Lanceolatus (Scribn. & J.G. Sm.) Gould	DRPG	native	11.1	13.1
Elymus trachycaulus (Link) Gould ex Shinners	DRPG	native	3.7	1.6
Elymus wawawaiensis J. Carlson & Barkworth	DRPG	native	7.4	11.5
Festuca idahoensis Elmer	DRPG	native	7.4	0.0
Hesperostipa comata (Trin. & Rupr.) Barkworth	DRPG	native	3.7	1.6
Leymus cinereus (Scribn. & Merr.) A. Löve	DRPG	native	11.1	14.7
Pascopyrum smithii (Rydb.) A. Love	DRPG	native	3.7	16.4
Pseudoroegneria spicata (Pursh) A. Löve	DRPG	native	37.0	29.5
Sporobolus airoides (Torr.) Torr.	DRPG	native	0.0	1.6
Sporobolus cryptandrus (Torr.) Gray -	DRPG	native	7.4	1.6
Agropyron cristatum (L.) Gaertn.	DRPG	non-native	14.8	11.5
Agropyron desertorum (Fisch. ex Link) J.A. Schult.	DRPG	non-native	29.6	55.7
Agropyron fragile (Roth) P. Candargy	DRPG	non-native	7.4	36.1
Bromus inermis Leyss. ssp. inermis	DRPG	non-native	3.7	0.0
Dactylis glomerata L.	DRPG	non-native	7.4	1.6
Festuca brevipila Tracey	DRPG	non-native	3.7	0.0
Lolium perenne L.	DRPG	non-native	3.7	0.0
Psathyrostachys juncea (Fisch.) Nevski	DRPG	non-native	14.8	18.0
Secale cereale L.	DRPG	non-native	0.0	1.6
Thinopyrum intermedium (Host) Barkworth & D.R. Dewey	DRPG	non-native	18.5	9.8
Thinopyrum ponticum (Podp.) Z.-W. Liu & R.-C. Wang	DRPG	non-native	7.4	14.8
Triticum aestivum L.	DRPG	non-native	3.7	4.9
Hordeum L.	DRPG	unknown	0.0	1.6
Poa L.	SRPG	native	0.0	6.6
Poa secunda J. Presl.	SRPG	native	11.1	9.8

*DRPG = deep-rooted perennial grass; SRPG = shallow-rooted perennial grass

Table S3. Summary of perennial forb and shrub species seeded and the percentage of study sites seeded with each species or genus.

Seeded genus/species scientific name	Life-form type*	Nativity	% of total burn-seeded (BS) study sites seeded with each species: Aerial	Drill
Achillea millefolium L.	PF	native	11.1	14.8
Balsamorhiza sagittata (Pursh) Nutt.	PF	native	0.0	1.6
Dalea purpurea Vent. var. purpurea	PF	native	0.0	1.6
Linum lewisii Pursh	PF	native	7.4	14.8
Penstemon palmeri Gray	PF	native	7.4	1.6
Sphaeralcea A. St.-Hil.	PF	native	3.7	0.0
Sphaeralcea coccinea (Nutt.) Rydb.	PF	native	3.7	3.3
Linum perenne L.	PF	non-native	11.1	13.1
Medicago sativa L.	PF	non-native	33.3	51.9
Medicago sativa L. ssp. sativa	PF	non-native	0.0	1.6
Melilotus officinalis (L.) Lam.	PF	non-native	7.4	8.2
Onobrychis viciifolia Scop.	PF	non-native	0.0	11.5
Sanguisorba minor Scop.	PF	non-native	14.8	11.5
Linum L.	PF	unknown	14.8	11.5
Artemisia L.	SHRUB	native	7.4	4.9
Artemisia tridentata Nutt.	SHRUB	native	7.4	1.6
Artemisia tridentata Nutt. ssp. tridentata	SHRUB	native	7.4	13.1
Artemisia tridentata Nutt. ssp. vaseyana (Rydb.) Beetle	SHRUB	native	0.0	3.3
Artemisia tridentata Nutt. ssp. wyomingensis Beetle & Young	SHRUB	native	48.1	42.6
Atriplex canescens (Pursh) Nutt.	SHRUB	native	48.1	31.1
Atriplex confertifolia (Torr. & Frém.) S. Wats.	SHRUB	native	3.7	1.6
Atriplex gardneri (Moq.) D. Dietr.	SHRUB	native	3.7	0.0
Chrysothamnus Nutt.	SHRUB	native	3.7	0.0
Krascheninnikovia lanata (Pursh) A.D.J. Meeuse & Smit	SHRUB	native	3.7	4.9
Purshia tridentata (Pursh) DC.	SHRUB	native	7.4	3.3
Bassia prostrata (L.) A.J. Scott	SHRUB	non-native	40.7	36.1

*PF = perennial forb