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Post-fire seeding on Wyoming big sagebrush ecological sites: Regression analyses of seeded nonnative and native species densities

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ABSTRACT

Since the mid-1980s, sagebrush rangelands in the Great Basin of the United States have experienced more frequent and larger wildfires. These fires affect livestock forage, the sagebrush/grasses/forbs mosaic that is important for many wildlife species (e.g., the greater sage grouse (*Centrocercus urophasianus*)), post-fire flammability and fire frequency. When a sagebrush, especially a Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis* (Beetle & A. Young)), dominated area largely devoid of herbaceous perennials burns, it often transitions to an annual dominated and highly flammable plant community that thereafter excludes sagebrush and native perennials. Considerable effort is devoted to revegetating rangeland following fire, but to date there has been very little analysis of the factors that lead to the success of this revegetation. This paper utilizes a revegetation monitoring dataset to examine the densities of three key types of vegetation, specifically nonnative seeded grasses, nonnative seeded forbs, and native Wyoming big sagebrush, at several points in time following seeding. We find that unlike forbs, increasing the seeding rates for grasses does not appear to increase their density (at least for the sites and seeding rates we examined). Also, seeding Wyoming big sagebrush increases its density with time since fire. Seeding of grasses and forbs is less successful at locations that were dominated primarily by annual grasses (cheatgrass (*Bromus tectorum* L.)), and devoid of shrubs, prior to wildfire. This supports the hypothesis of a "closing window of opportunity" for seeding at locations that burned sagebrush for the first time in recent history.

1. Introduction

Great Basin wildfires have recently occurred more frequently and at larger scales than in previous decades. Between the late 1980s and late 1990s, the number of fires doubled and mean fire size increased by 400% in the Great Basin lands of Idaho, Nevada, Oregon, and Utah (Pyke and McArthur, 2002; Pyke et al., 2003). In response, the U.S. Bureau of Land Management (BLM) has channeled more resources toward fire rehabilitation through its Emergency Fire Rehabilitation (EFR) Program. One component of EFR consists of post-fire seeding. The EFR program has eight principal objectives (Environmental Assessment for Normal Year Fire Rehabilitation Plan, June 2004, Winnemucca District):

1. To promptly stabilize and prevent further degradation to affected resources on lands within the fire perimeter.
2. To repair damages caused by fire suppression operations in accordance with approved land management plans, regulations, policies, and all relevant federal, state, and local laws.
3. Prevent losses to private structures and property on public lands.
4. To prescribe cost effective post-fire stabilization measures necessary to protect human life, property, critical cultural and natural resources.
5. To repair or improve lands damaged directly by the wild land fire and unlikely to recover naturally from severe fire damage by emulating historic or pre-fire ecosystem structure, function, diversity, and dynamics.
6. To restore and/or establish healthy, stable ecosystems in the burned area, even if these ecosystems cannot fully emulate historic or pre-fire condition.
7. To restore sagebrush habitats that fall within sage-grouse/sagebrush obligate species use areas as a high priority.
8. Deter the establishment and spread of noxious and invasive species.

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In northern Nevada and southern Idaho, Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis* (Beetle & A. Young)) communities are undergoing rapid conversion to monocultures of nonnative annual grasses (specifically, cheatgrass (*Bromus tectorum* L.)) and other weeds. Nonnative invasive species such as cheatgrass are well adapted for invading and occupying open ecological niches following fire in Wyoming big sagebrush communities (Mosley et al., 1999). In turn, the invasive species that fill post-fire niches tend to tolerate and fuel subsequent fire. This sets in motion a fire–invasion–fire cycle that has been increasing in both magnitude and frequency in xeric (e.g., Wyoming big sagebrush) communities (D'Antonio and Vitousek, 1992; Pyke et al., 2003). The period immediately following a fire is a window of opportunity for breaking the fire–invasion–fire cycle by seeding species less prone to burn (Evans and Young, 1978) and that allow sagebrush to reoccupy the site.

For two decades, it has been possible to devote EFR funding (for three years following fire) and watershed funding (for the subsequent years) toward the monitoring of EFR seeding efforts. A recent study, however, finds that such monitoring has rarely resulted in data that can be summarized for widespread use and review (Pyke and McArthur, 2002). Furthermore, most managers do not keep track of monitoring data in a routine and systematic fashion (U.S. General Accounting Office, 2003, p. 5). Given the vast financial resources that are being devoted to EFR, it seems appropriate to improve the collection and analysis of data that may shed light on the relative success of ongoing EFR efforts (U.S. General Accounting Office, 2003). Data collection and analysis are necessary to: (1) indicate whether EFR seeding treatments are accomplishing objectives, (2) justify (or not) seeding programs on mid and early ecological status areas to prevent creation of annual grass and nonnative perennial weed communities, and most importantly, (3) learn to accomplish objectives more effectively. In a broad policy context, it is useful to examine data from areas after EFR treatments have been applied to assess the potential success of such efforts.

We use a unique dataset from burned areas in the U.S. BLM's Winnemucca District in Nevada (M. Zielinski, unpublished data). The Winnemucca District is approximately 10 million acres, and over 2.3 million acres of that have burned in the last 20 years. We estimate that 25–30% of the Wyoming big sagebrush communities in the Winnemucca District have been impacted by fire.

The objectives of our analysis are to examine the influence of certain site-specific and site management characteristics on the subsequently measured prevalence of (1) seeded perennial grasses, (2) seeded perennial forbs, and (3) Wyoming big sagebrush, in areas burned by wildfire in the Winnemucca District, Nevada in the 1980s and 1990s. The data include vegetation measurements from multiple times on multiple locations following fires (the data are both time series and cross-sectional in nature, i.e., longitudinal data). The dataset includes locations that were seeded following fire as well as ones that were not, thereby providing a control group of locations. Because seed or funds often limited the total area seeded, control locations could be set up without neglecting acreage that would otherwise have been seeded.

This paper presents the first completed analysis of the full dataset (i.e., using all of the time series observations). A recent paper by Eiswerth and Shonkwiler (2006) used a system of equations to examine only the most recent monitoring data following the fires. Use of only the most recent monitoring measurements allowed those authors to use a multiple equation systems approach to incorporate potential linkages between unobserved factors underlying the growth of multiple plant species. However, this was done for econometric reasons at the expense of not using earlier post-fire monitoring measurements. The present manuscript uses simpler statistical techniques, thus enabling use of the full range of time series observations, and concentrates on basic hypotheses put forth by land managers regarding causal factors of EFR success. Also in contrast to Eiswerth and Shonkwiler (2006), we include analysis of post-fire densities of seeded forbs. Forbs are seeded as part of EFR for watershed stabilization.

2. Methods

2.1. Study locations

Our analysis involved geographic areas in the BLM Winnemucca Nevada District at which wildfires occurred between 1984 and 1997. We designated each distinct wildfire area as a "fire location" ($n = 60$ fire locations). Furthermore, at each fire location, we delineated one or more independent "study locations" ($n = 111$ study locations) that were monitored one or more times ($n = 287$ observations) following the fires, from 1986 to 1999. Multiple study locations established at any one fire were selected to represent different or treatment areas, areas with different seeding prescriptions, or areas classified as distinctly different soils (Web Soil Survey and Soil Data Mart on line at Soils.USDA.GOV) or ecological sites. An ecological site is a distinctive kind of land with specific physical characteristics that differs from other kinds of land in its ability to produce a distinctive kind and amount of vegetation (NRCS, 2003). An "observation" corresponds to a study location monitored at a moment in time; the data therefore are time series/cross-sectional (longitudinal) in nature. For each observation, the plant density of each perennial species present was measured. Study locations consisted of areas that were seeded as well as ones that were not (thereby providing "controls" – see below). The majority (about 66%) of the total observations corresponded to locations that were seeded following fire.

Monitoring plots were located in key areas that represent the dominant soil and ecological site. Prior to monitoring site selection, key areas were stratified based on dominant treatment by pasture, allotment, and access. One or two key areas were selected per treatment per fire. Key areas were established the first growing season after treatment, generally during the months of May and June. The monitoring team evaluated the treatment area to determine final site selection location, chosen on the basis of uniform landform and uniform treatment.

In our dataset, control plots were not technically statistical controls. Control plots were untreated areas on the same or similar soils and on the same ecological site. Unseeded site observations were sometimes directly adjacent to the treatments, but they also could exist in untreated pastures on the same burn or another burn. In large fire years, it was impossible to treat all suitable areas; insufficient seed and equipment and a limited seeding window prevented completion of treatments. BLM field office capability in the District under study was 60–80 thousand acres of drill seeding and 60–80 thousand acres of aerial seeding per year. Across the whole of the District, approximately 12% of the burned areas were seeded. Monitoring focused on the treatment areas, with treatment boundary areas designed for the ease of drill contractors by using roads, fences, and section lines. The intent of an unseeded site observation (location) was to have (at a minimum) data for the same ecological site affected by the same fire or fire complex.

The seeding method consisted of using standard rangeland drills. Seeds were mixed (grasses, forbs, and shrubs) prior to filling the seed boxes. Where sagebrush was seeded, the sagebrush seed was mixed directly with the other seeds and then placed into the seed boxes. No aerial seeding was included. Contamination of control sites by aerial application was not possible. Most of the seeding was conducted on a soil texture of very fine sandy loam high in volcanic ash. The northern two-thirds of the field office are covered by a loess cap. Drill arms were not chained to a specific depth. Generally soils were dry or frozen; seed tubes were removed when humidity was high or ground was snow covered.

The density data collection procedure used 10 circular plots on one transect placed perpendicular to drill rows or perpendicular to slope for nonseeded areas. All perennial plants were counted individually; annual plants (cheatgrass) were counted as individuals until density became too high to effectively count and were then estimated to the nearest 10 or 25. The totals of each plot were added together then divided by 10 for plot average. Plot size was 0.89 sq. meters or a 9.6 square foot production plot. This plot size is generally used in areas where vegetation density and production are relatively light (300–800 pounds of annual air-dry production per acre).

We narrowed our focus of study to ecological sites of the Wyoming big sagebrush type – specifically, Loamy 8–10″ precipitation and Droughty Loam 8–10″ precipitation ecological sites (NRCS, 2003). In addition to the importance of and widespread interest in this vegetation type, we focused on these sites because the majority of the 287 total observations in the dataset (about 60%) were of these ecological site types, with the balance scattered fairly evenly among four other heterogeneous vegetation types on other ecological sites. Thus there were 166 monitoring observations corresponding to 67 study locations of the Wyoming big sagebrush type in our final dataset.

2.2. Data

Study location characteristics contained in the dataset included soil type (i.e., soil series, surface texture) and the pre-fire vegetative condition (specifically, whether shrubs were present before the fire). Land manager response data included: (1) species seeded (if any) and (2) the seeding rate used for each species (recorded in units of pure live seed (PLS) per square foot). In aggregate, 23 species were seeded in the collection of study locations (though not all 23 were seeded at any one location), with seeding primarily consisting of nonnative perennial grasses (crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.) and Siberian wheatgrass (*Agropyron fragile* (Roth) P. Candargy)) and forbs (e.g., alfalfa (*Medicago sativa* (L.)), forage kochia (*Bassia prostrata* (L.) A.J. Scott), and sanfoin (Onobrychis viciaefolia (Scopoli))). In addition, Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis* (Beetle & A. Young)) was seeded at some locations. Introduced grass and forb seeds were used simply because a source of native seed was not available at the time of seeding and the cost was required to be below $5.00 per pound. At that time, BLM's objective was not to restore but rather to stabilize the watershed.

Plant density measurements were performed at each of the study locations in various years after seeding. Monitoring was conducted as soon as one year and as long as 13 years after the fire. The typical location was monitored at 3–4 moments in time following seeding. The units for the plant density measurements were numbers of plants per 0.89 square meters.

2.3. Explanatory variables and hypotheses

We created explanatory variables for testing hypotheses about the factors that influence plant densities following wildfire (Table 1). We anticipated that the densities of various species would be influenced by whether post-fire seeding operations were conducted (SEEDED = 1). We also expected the density of Wyoming big sagebrush to be higher, other things being equal, at locations where this species was seeded after a fire (ARTRWSEED = 1) and at locations where the seeding of *any* species took place (SEEDED = 1), since seeding may be expected to reduce annual species and fire frequency, allowing Wyoming big sagebrush time to establish before a repeat fire. Conversely, we had reason to hypothesize that it may be difficult to pick up the influence of the indicator variables SEEDED = 1 (since there has been limited time for repeat fires) and ARTRWSEED = 1 (since only 6% of locations were seeded with sagebrush (Table 1)).

For each ecological site or soil and set of seeded species there would be an optimum seeding rate influenced by seed quality, seedbed conditions, species growth habits, tolerance of stress at various life stages, and competition (see Whisenant, 1999 and Pyke and Archer, 1991). We tested hypotheses regarding the influence of nonnative perennial grass seeding rates (GRASSRT) and forb seeding rates (FORBRT) on their subsequent densities. Young et al. (1994) found an inverse relationship between seeding rates and subsequent density of seedlings of Indian ricegrass in Nevada, possibly due to factors such as intraspecific competition. Similarly, Francis and Pyke (1996) found that lower seeding rates of crested wheatgrass led to increased biomass per plant, tiller production, and competitive ability with higher seeding rates leading to higher production per unit area in a greenhouse study. Mueggler and Blaisdell (1955), working in an 11-inch per year precipitation zone found that self-thinning and variable plant size eliminated biomass production differences within three years after seeding at rates that varied from 2 to 24 pounds per acre (2.2–27 kg ha^{-1}) of crested wheatgrass.

We expected the densities of most species to change as time passed following a wildfire (YRSSINCE, Table 1). Densities of native species should increase as more time allows for natural recovery and recruitment. For seeded perennial grasses, however, we expected densities to be high in the first and second years following seeding (in our data, measurements of first year densities were taken to determine emergence), with decreases after that (as the plants mature) before stabilizing at 3–5 years after establishment as implied by Mueggler and Blaisdell (1955) and Pyke and Archer (1991).

We also tested the hypothesis that seeded species have higher densities following seeding on those locations where shrubs were present during the fire (PRESHRUB = 1) in comparison with locations dominated by annual grasses (Young and Allen, 1997, pp. 533–534). This may occur because fires on sagebrush stands burn hot enough (i.e., much hotter than fires on an annual grass monoculture) to kill cheatgrass seeds, thereby lowering subsequent competition (Young and Allen, 1997, pp. 533–534). This is relevant for land managers because failure to successfully establish a perennial stand may exclude sagebrush and perpetuate

Table 1
Variable names, mean values, and definitions for the independent variables used in the analysis

Variable name	Mean value	Variable definition
YRSSINCE	3.34	Number of years that elapsed between the time of the fire and the time of plant density monitoring
PRESHRUB	0.90 (90% of locations)	A binary indicator variable indicating whether shrubs were present at the location before the fire occurred (=1 if shrubs were present; otherwise = 0)
ARGID	0.51 (51% of locations)	Binary indicator variable indicating whether the soil type at the location is Argid (=1 if soil type is Argid; otherwise = 0)
ARTRWSEED	0.06 (6% of locations)	A binary indicator variable indicating whether the fire location was seeded with Wyoming big sagebrush [ARTRW] following the fire (=1 if ARTRW was seeded; otherwise = 0)
SEEDED	0.66 (66% of locations)	A binary indicator variable indicating whether the fire location was seeded (with any species) following the fire (=1 if seeding was undertaken; otherwise = 0)
GRASSRT	22.39	Seeding rate for nonnative perennial grasses (PLS per square foot)
FORBRT	4.53	Seeding rate for nonnative perennial forbs (PLS per square foot)

a cheatgrass fire cycle with frequent but cooler fires. That is, there may be an opportunity (for successful seeding) at a location that has had a fire that has burned sagebrush for the *first time* in recent history.

Finally, we also wished to test whether soil type had any influence on seeded species density. We hypothesized that on argid soil types, infiltration would be shallower than on orthids. Based on field observations, calcium carbonate and silica will deposit generally between 15 and 25 inch in argids on a Wyoming big sagebrush site, whereas the wetting front on orthids would be 25–35 inch on the same site with the same moisture. Soil moisture in the upper soil profile on an argid would be greater than an orthid. The subsoil restricts infiltration until the surface layer is saturated. This increased moisture availability would increase germination and establishment of new seedlings. We therefore tested the hypothesis that seeded species would perform better on this soil type (ARGID = 1).

2.4. The statistical models

Four ordinary least-squares regression models were estimated. Each of these regressions took the following general form:

$$\text{DEN}_t = \alpha + \beta(\text{YRSSINCE}_t) + \delta X + \gamma Z \qquad (1)$$

where: DEN_t = density measurement in year t for the vegetation type in question (depending on the regression; see below); YRSSINCE_t = number of years elapsed between the time of the fire and year t (the year the plant density monitoring observation was taken), X = vector of location-specific characteristics, Z = vector of location management characteristics; and where α and β are parameters to be estimated and δ and γ are vectors of parameters to be estimated.

The first regression concerned the density of nonnative seeded perennial grasses following post-fire seeding. For that regression the dependent variable DEN_t was the sum of density measurements in year t for the nonnative seeded perennial grass species crested wheatgrass and Siberian wheatgrass. The second regression modeled the evolution of nonnative seeded perennial forbs following post-fire seeding. For that regression the dependent variable DEN_t was the sum of density measurements in year t for the nonnative seeded perennial forb species alfalfa, forage kochia, and sanfoin. Finally, the third and fourth regressions examined the evolution of Wyoming big sagebrush following fire. In those regressions the dependent variable DEN_t was the density of Wyoming big sagebrush plants in year t.

Initially (in the unrestricted regression models), each of the independent variables available for analysis was included in a particular regression equation according to the hypotheses presented in the section above. However, variables with estimated coefficients less than one standard error away from the null hypothesis were omitted from most final restricted models in the process of testing down (Kennedy, 1992). Seeding rate was included for consistency in all the final regression models so that comparisons of coefficient values and levels of statistical significance could be made across the different vegetation-type models.

3. Results

All of the preliminary (unrestricted) regressions estimated in this study shared in common the result that the soil type indicator variable ARGID was never statistically significant. For this reason we did not include this variable in any of the study's final regressions.

3.1. Nonnative seeded perennial grasses

In the regression equation for nonnative seeded perennial grasses, the estimated coefficient for YRSSINCE was negative and statistically significant at the 99% level of confidence (Table 2). Consistent with prior expectations, this indicates that as the seeded perennial grass plants mature, the densities decrease (holding the other variables constant). In addition, the densities of seeded perennial grasses were higher at locations where shrubs were present prior to the fire (PRESHRUB). Again, this is consistent with our stated hypothesis and the estimated coefficient is statistically significant at the 99% level of confidence. This result supports the hypothesis that seeding tends to be more successful at locations where shrubs were present before the fire (rather than an annual monoculture without shrubs). The estimated coefficient for GRASSRT was not statistically significant. Therefore, we cannot reject the null hypothesis that seeding rate has no effect on the subsequent density of the nonnative seeded perennial grasses. The R-square value (0.19), which indicates overall explanatory power, is in the range that one would expect for cross-section regressions of this sort. Our interest is in the statistical significance of the individual included variables rather than the overall explanatory power, which one expects to be relatively low.

3.2. Nonnative seeded perennial forbs

The regression equation for the density of nonnative seeded perennial forbs was estimated over those locations at which the seeding of those forbs took place (Table 2). The number of observations was thus relatively small ($n = 67$). Similar to the regression for nonnative seeded perennial grasses, the estimated coefficient for YRSSINCE was negative. Unlike the perennial grasses regression, however, the number of years since the fire was not statistically significant. Also similar to the regression for nonnative seeded grasses was the finding that nonnative seeded forbs performed better on locations at which shrubs were present prior to the fire (as opposed to annual grass-dominated locations). This is indicated by the positive and statistically significant estimated coefficient (90% level of confidence) for PRESHRUB. Finally, the seeding rate of the nonnative forbs (FORBRT) had a statistically significant (95% level of confidence) positive impact on the subsequent density of these plants. Higher seeding rates of forbs do lead to higher subsequent densities of forbs, at least over the sample of locations and seeding rates that we have available for analysis.

Table 2
Regression results: influence of factors on the densities of nonnative seeded perennial grasses[a] and forbs[b]

Variable	Perennial grasses regression coefficients[a] (p-values shown in parentheses)	Perennial forbs regression coefficients[b] (p-values shown in parentheses)
CONSTANT	3.44 (0.2389)	−0.07 (0.7304)
YRSSINCE	−0.65*** (0.0103)	−0.03 (0.3389)
PRESHRUB	7.23*** (0.0001)	0.39* (0.0621)
GRASSRT	−0.04 (0.6928)	
FORBRT		0.05** (0.0475)
	$n = 94$; R-square value = 0.19	$n = 67$; R-square value = 0.14

*Indicates statistical significance at the 90% level of confidence for a 2-tailed test.
**Indicates statistical significance at the 95% level of confidence for a 2-tailed test.
***Indicates statistical significance at the 99% level of confidence for a 2-tailed test.
[a] Dependent variable = sum of the density measurements for the following nonnative seeded perennial grass species: crested wheatgrass (*Agropyron cristatum*) and Siberian wheatgrass (*Agropyron sibericum*). The regression is estimated over those locations at which the seeding of nonnative perennial grasses occurred.
[b] Dependent variable = sum of the density measurements for the following nonnative seeded perennial forb species: alfalfa (*Medicago sativa*), forage kochia (*Kochia prostrata*), and sanfoin (Onobrychis viciaefolia). The regression is estimated over those locations at which the seeding of nonnative perennial forbs took place.

3.3. Wyoming big sagebrush

Table 3 shows the results for the regression equation that examines the influence on Wyoming big sagebrush densities of seeding *any* species following fire. This regression was estimated over all observations, since we may reasonably expect to find Wyoming big sagebrush at both seeded and nonseeded locations. The estimated coefficient for PRESHRUB was significantly positive (90% level of confidence), as expected. Furthermore, the estimated coefficient for YRSSINCE was significantly positive (95% level of confidence), indicating that as time elapses following a fire, the density of Wyoming big sagebrush increased. This is true regardless of whether seeding operations took place at the location or not, since we have controlled for this through the inclusion of the SEEDED indicator variable. This result indicates that natural recruitment of Wyoming big sagebrush is taking place over time following fire.

The estimated coefficient for the SEEDED indicator variable was positive but not statistically significant at the 0.90 level ($p = 0.13$). This means that, for the set of locations and length of longitudinal data included in our sample, we cannot say with confidence that the seeding of simply any species (primarily, crested wheatgrass in our data) tends to increase the subsequent density of Wyoming big sagebrush. Years since fire and pre-fire condition are more important determinants of sagebrush establishment.

Table 4 presents a slightly different regression equation that examines the success of seeding Wyoming big sagebrush specifically. The indicator variable denoting whether Wyoming big sagebrush was seeded (ARTRWSEED) was positive as expected and statistically significant at the 95% level of confidence. In addition, the variable denoting years since fire (YRSSINCE) was positive and statistically significant at the 99% level of confidence, indicating that the density of Wyoming big sagebrush increased through time following fire, after controlling for the influence of whether Wyoming big sagebrush was seeded or not. Finally, unlike any other regression reported in this study, the estimated coefficient of the variable PRESHRUB was not statistically significant ($p = 0.2262$).

4. Conclusions

These analyses can guide expectations for revegetation and management of rangelands following wildfire. First, densities of both seeded grasses and seeded forbs will be higher at those fire locations where the pre-burn vegetation included shrubs (e.g., sagebrush) rather than a monoculture of annual grasses. Once type conversion to annual grasses occurs, much more effort is apparently required to establish perennial grasses and shrubs or there is much less success in rangeland revegetation. This is expected at locations where annuals provide intense seedling competition and fuel a frequent fire cycle that prohibits sagebrush from reaching reproductive age. Our findings support those of Eiswerth and Shonkwiler (2006), who found that higher seeding rates for nonnative perennial grasses led to lower subsequent densities of cheatgrass, other factors held constant. These are reasons why it has been recognized as an imperative to streamline the seeding process to make it as successful as possible in this rapidly closing window of opportunity immediately after the first shrub-removing fire.

Table 3
Regression results: influence of post-fire seeding (of any species) and other factors on Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) density[a]

Variable	Coefficient (p-values shown in parentheses)
CONSTANT	−0.015 (0.1571)
YRSSINCE	0.002** (0.0163)
PRESHRUB	0.016* (0.0925)
SEEDED	0.009 (0.1271)

*Indicates statistical significance at the 90% level of confidence for a 2-tailed test.
**Indicates statistical significance at the 95% level of confidence for a 2-tailed test.
$n = 151$; R-square value = 0.07.
[a] Dependent variable is density of Wyoming big sagebrush plants.

Table 4
Regression results: influence of post-fire seeding of Wyoming big sagebrush and other factors on Wyoming big sagebrush (*Artemisia tridentata wyomingensis*) density[a]

Variable	Coefficient (p-values shown in parentheses)
CONSTANT	−0.006 (0.4618)
YRSSINCE	0.003*** (0.0086)
PRESHRUB	0.011 (0.2262)
ARTRWSEED	0.033** (0.0343)

**Indicates statistical significance at the 95% level of confidence for a 2-tailed test.
***Indicates statistical significance at the 99% level of confidence for a 2-tailed test.
$n = 151$; R-square value = 0.08.
[a] Dependent variable is density of Wyoming big sagebrush plants.

Second, the passage of time following fire leads to lower densities for nonnative seeded perennial grasses but higher densities of Wyoming big sagebrush. This latter result indicates that natural recruitment of Wyoming big sagebrush can occur over time. Of course, the necessary ingredients are a seed source and perennial grass under story. A source of sagebrush seed is often a problem as 90% of the seed falls within 2 m of the mother plant and the seed viability is generally one year.

Third, seeding of Wyoming big sagebrush is effective for increasing the densities of this species after wildfire. This finding is relevant given the key role sagebrush plays for some wildlife species in the rangeland ecosystem (McAdoo et al., 2004, 1989), for example the greater sage grouse (*Centrocercus urophasianus*) (Aldridge and Brigham, 2003; Sveum et al., 1998). Large areas of sage grouse habitat and winter range for mule deer have been eliminated in the Winnemucca District, but seeding sagebrush can mitigate some of this habitat. Sagebrush community connectivity needs to be maintained for migration corridors. The natural spread of Wyoming big sagebrush over time tends to occur slowly (Monsen et al., 2004; Pedersen et al., 2003), but land managers may be able to hasten the process by seeding sagebrush following fire. Currently, the BLM encourages sagebrush seedlings to be planted on rehabilitation projects; seedling establishment is successful but this is expensive, time consuming and extremely limited in scale. Monitoring data on seeding sagebrush is limited or unknown and such seeding is assumed to be generally unsuccessful. This dataset indicates that sagebrush seeding can be effective when pre-burn conditions were sagebrush with perennial grass under story.

Fourth, higher seeding rates for nonnative perennial grasses do not lead to higher perennial grass densities, at least for the sample of sites and seeding rates (mean = 22.39 PLS per square foot) used in this analysis. This suggests that there may be economic gains associated with not over-seeding an area with perennial grasses following fire. There is no reason to think that seeding rates used in this study are not within an optimum range, as they apparently were high enough to compete with weeds and low enough to prevent excess self competition and allow successful establishment. It also reinforces discussion in the literature regarding the need for further research on the linkages among starting-point plant densities, density-dependent mortality, plant density and cover dynamics, and optimal rangeland seeding rates (Pyke and Archer, 1991; Francis and Pyke, 1996). However, higher seeding rates for nonnative perennial forbs led to higher subsequent densities of forbs, at least for the sample of sites and seeding rates included in our analysis. Future analysis of forb density monitoring data from other rangeland fire locations is warranted to allow for a check of this finding and to sort out the specific responses of each species or species combination (Pyke and Archer, 1991).

To specifically address the EFR objectives 3–8 (stated in Section 1) and the objectives of this study, the following comments apply:

(1) Lowering the potential risk for invasion of burned areas by nonnative invasive plants – although not tested specifically by this study, establishing perennial grasses, forbs and shrubs occupies ecological niches that otherwise would be available to invasive annuals. This is consistent with results found by Eiswerth and Shonkwiler (2006).
(2) Increasing native perennial plants – this occurred in some seedings where native perennial grasses, forbs, and/or shrubs were established. These and other seedings are expected to contribute toward this objective by reducing fire frequency due to the extreme flammability of the alternative cheatgrass (Ziska et al., 2005; Blank et al., 2006).
(3) Maintaining bunchgrass–shrub communities – annual weeds would otherwise quickly dominate interspaces, preventing establishment of perennial grasses and many other native plants. Fire sensitive natives that can occasionally establish, such as sagebrush, would be vulnerable to frequent fires fueled by flashy annual fuels. Because sagebrush has small seeds that do not persist in the soil or travel far, it increases gradually after fire, coming in from edges, and out unburned islands and mother plants until the next fire. Where crested wheatgrass or other perennial bunchgrasses are established and will be in place to reduce fire frequency, sagebrush and other shrubs will survive more successfully, especially where seeded successfully.
(4) Lowering of fire suppression costs – the flammability of cheatgrass means that the success of seeded alternatives will reduce flammability and future fire suppression costs on site. These seeded areas will serve as fuel breaks that reduce fire size and therefore average fire frequency.
(5) Maintaining plant community function by promoting establishment of grasses, shrubs, and forbs – although not native, crested wheatgrass serves similar functions for wildlife habitat, nutrient cycling, watershed protection, and maintenance of the natural fire regime for a bunchgrass sagebrush ecosystem.

Little research to date has been undertaken to examine the success of post-fire treatments on rangeland in the western United States. There is a pressing need for further analyses, which will not be possible without the systematic collection of location-specific data on fire site characteristics, management techniques used in post-fire seeding, and plant community trajectories over time. Also, though the dataset employed in this paper is to our knowledge the first ever of its kind, it still is not the result of a strict scientific experiment with treatment and completely random controls. Rather, it is the result of an unusually good-faith effort to record the characteristics of and changes at locations where human fire rehabilitation efforts have been funded and applied. It would be interesting to assess the causal factors behind seeding success at additional types of rangeland sites, and at collections of sites for which scientifically generated control sites are available. We expect that doing so would yield management implications that are more robust and that vary across ecological sites (NRCS, 2003).

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