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Abstract The ecological integrity of the Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle and A. Young) alliance is being severely interrupted by post-fire invasion of non-native annual grasses. To curtail this invasion, successful post-fire revegetation of perennial grasses is required. Environmental factors impacting post-fire restoration success vary across space within the Wyoming big sagebrush alliance; however, most restorative management practices are applied uniformly. Our objectives were to define probability of revegetation success over space using relevant soil-related environmental factors, use this information to model cost of successful revegetation and compare the importance of vegetation competition and soil factors to revegetation success. We studied a burned Wyoming big sagebrush landscape in southeast Oregon that was reseeded with perennial grasses. We collected soil and vegetation data at plots spaced at 30 m intervals along a 1.5 km transect in the first two years post-burn. Plots were classified as successful (>5 seedlings/m^2) or unsuccessful based on density of seeded species. Using logistic regression we found that abundance of competing vegetation correctly predicted revegetation success on 51 % of plots, and soil-related variables correctly predicted revegetation performance on 82.4 % of plots. Revegetation estimates varied from \$167.06 to \$43,033.94/ha across the 1.5 km transect based on probability of success, but were more homogenous at larger scales. Our experimental protocol provides managers with a technique to identify important environmental drivers of restoration success and this process will be of value for spatially allocating logistical and capital expenditures in a variable restoration environment.

Keywords Annual grass · *Artemisia tridentate* · Restoration · Wildfire · Perennial grass

Introduction

Non-native annual grasses first appeared in western US sagebrush (*Artemisia tridentata* Nutt.) plant communities in the latter 1800's (Mack 1981). Establishment of these species was fostered by increases in site availability associated with intensive livestock grazing practices of the early 20th Century (Knapp 1996; Krueger-Mangold and others 2006; Mack 1981). Non-native annual grasses such as cheatgrass (*Bromus tectorum* L.) and medusahead (*Taeniatherum caput-medusae* (L.) Nevski) are poor competitors with established native perennial grasses (Chambers and others 2007; Davies 2008; Humphrey and Schupp 2004), however, annual grasses desiccate early in the growing season (compared to native grasses), creating a fine fuel bed that can dramatically shorten fire return intervals (Brooks and others 2004; Davies and Svejcar 2008; Pellant and others 2004). This increase in fire frequency can kill native perennial grasses and shrubs and creates a self-reinforcing feedback loop that favors proliferation of exotic annual grasses. The range of annual grasses has now spread to some 24 million ha in the

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C. S. Boyd (✉) · K. W. Davies
USDA-Agricultural Research Service, Eastern Oregon Agricultural Research Center (EOARC), 67826-A Hwy 205, Burns, OR 97720, USA
e-mail: chad.boyd@oregonstate.edu

western United States (Miller and others 1999; Whisenant 1990). Consequences associated with the proliferation of annual species include interruption of ecological processes and loss of soil resources, native plant communities, livestock forage, and habitat for sagebrush-associated wildlife species (D'Antonio and Vitousek 1992; Davies and Svejcar 2008; Melgoza and others 1990; Miller and Eddleman 2000; Stringham and others 2003; Whisenant 1990).

This expansion in the range and biological consequences of annual grass proliferation has been of particular consequence in low elevation sagebrush plant communities typified by the Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle and A. Young) alliance. In response, management agencies have undertaken a massive effort to restore perennial grasses in these systems. This effort has largely focused on seeding perennial grass species (both native and non-native) in the post-fire environment. Knutson and others (2009) estimated that in 2007 the Bureau of Land Management (BLM) spent over 60 million dollars in emergency rehabilitation and stabilization funds to seed critical areas (mainly burned areas) in the Intermountain West. Biological success of these efforts is often low, particularly at low elevations and when using native species (Hull 1974; Lysne and Pellant 2004; Mosley and others 1999; Richards and others 1998).

Despite low success of seeded perennial grasses in Wyoming big sagebrush communities, notable progress has been made in understanding the ecology of planted species (Drenovsky and others 2008; Hardegree and Van Vactor 2004; James and Drenovsky 2007). However, the successful application of these advances to field restoration will be contingent upon our ability to recognize windows of management opportunity in a restoration environment that varies strongly in both space and time (Boyd and Svejcar 2009).

At present, decisions regarding the timing of restoration efforts are strongly tied by funding policy to the immediate post-fire environment. For example, BLM managers are expected to submit restoration plans for expenditure of Emergency Stabilization funds within seven calendar days of containment of the fire and those funds must be spent within one year of fire containment (U.S. Department of the Interior 2004). By convention, broad-scale decisions regarding where to seed within the burned area are often associated with logistical convenience or perceived probability of success, however, these decisions are hampered by a lack of site specific ecologically-based guidance. We suspect that this latter point is key to understanding the reasons underlying limited success in post-fire restoration of Wyoming big sagebrush plant communities.

Given the increasingly limited funds available for post-fire restoration, it is critical for managers to have the tools necessary for determining spatial probability of success within a heterogeneous restoration environment. Such tools would allow managers to more wisely partition their logistical and capital restoration expenditures across the landscape. The actual cost of successful post-fire restoration on a unit area basis (e.g., dollars/ha) can be thought of as the cost of initial restoration treatment as modified by the probability of success. Our objectives were to model the influence of soil factors on the probability of revegetation success along a spatial gradient in a burned Wyoming big sagebrush landscape and to use this information to highlight spatial variability in the cost of successful restoration. We also contrasted the importance of soil-related variables with the role of vegetation competition in influencing restoration success. Results of this study will provide a technique for increasing understanding of the environmental factors influencing restoration success as well as the financial implications of seeding in a spatially variable restoration environment.

Methods

Study Area

This study was conducted in the Wyoming big sagebrush alliance approximately 65 km east of Burns, OR (lat 43.48 °N, long 119.72 °W). Elevation at study sites was approximately 1,100 m and slopes were 2–16 %. Aspect varied between sites in an approximately 180° range and sites were located within two Bureau of Land Management (BLM) grazing allotments with differing management histories. Soils were a complex series and surface textures ranged from clayey, to silty or gravelly loam underlain by clay pan or bedrock at depths from 10 to 50 cm (NRCS 2007). Our study occurred within multiple Rangeland ecological sites: SR Adobeland 9-12PZ, SR Clayey 9-12PZ and SR Shallow 9-12PZ (NRCS 2007).

A 13,000 ha area, which included our study site, was burned by wildfire in August of 2007. The area surrounding and including our study site was burned completely, indicating sufficiency of fine fuel loading for high fire continuity. Direct measurements of fire behavior/intensity were not available, however, weather on the day the fire started was extreme; maximum air temperature of 38.9 °C, low of 7 % relative humidity and wind gusts to 54.4 km/h (Burns District BLM file data). Prior to the fire, this area was sagebrush/bunchgrass vegetation characterized by Wyoming big sagebrush, bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve), Great Basin wildrye (*Leymus cinereus* (Scribn. & Merr.) A. Löve), Sandberg bluegrass (*Poa secunda* J. Presl), and squirreltail (*Elymus elymoides* (Raf.) Swezey). Medusahead and cheatgrass were also present. Our study sites were within a 4,000 ha area of the burn that was seeded with a rangeland drill in October of 2007. The seed mix included 4.6 kg/ha of crested wheatgrass (*Agropyron cristatum* L.), 2.3 kg/ha of Siberian wheatgrass (*Agropyron sibiricum* (Wild.) Beauv.), 2.3 kg/ha of Secar bluebunch wheatgrass (*Elymus wawawaiensis* J. Carlson & Barkworth), 1.14 kg/ha of Great Basin wildrye and 1.14 kg/ha of squirreltail. Prior to the fire, the study area was grazed by cattle during the growing season. Grazing was curtailed following fire in 2007 with continued non-use in 2008–2009.

Annual precipitation is highly variable but averages approximately 350 mm with the majority falling as rain or snow during the October to March period (Drewsey, OR weather station; Oregon Climatic Service 2007). Precipitation impacting germination, emergence and survival of seedlings (October through June) was 96 % of the long term average (24.73 cm) in the year of planting (2007–2008) and 97 % of normal in the year following planting (2008–2009, Drewsey, OR weather station; Oregon Climatic Service 2007).

Plot Layout and Data Collection

In October 2007, we installed permanently marked 6 × 6 m plots along a linear 1.5 km transect oriented northwest to southeast within the confines of the burned area. Location of the transect was within an area with topography and soils typical of the burned unit. A total of 51 plots were spaced at 30 m intervals along the transect. We counted density of seeded species in three randomly placed 1 m^2 quadrats for each plot in July of 2008 and 2009. Seedlings were only counted if they were in a drill row, and in 2009 were identified as "non-native" (*Agropyron* spp.) or "native" (Secar bluebunch wheatgrass, Great Basin wildrye, and squirreltail); seedlings were too immature to estimate by species in 2008. We estimated density of non-seeded herbaceous vegetation, one year post-fire, in July of 2008 by counting the number plants, by species, present in 10 randomly located 40 × 50 cm quadrats within each plot. These data were partitioned into functional groups (large perennial grasses, Sandberg bluegrass, forbs and shrubs) for presentation purposes (Davies and others 2007a).

At each plot we measured a panel of soil-associated environmental variables. Soil surface compaction was measured as the force (J) necessary to penetrate the top 5 cm of the soil surface in June of 2008 using a cone penetrometer (Herrick 2005; Herrick and Jones 2002). Soil samples for water content and nutrient and texture analysis were collected in May 2008 using a 2.5 cm-diameter auger at 0–5 cm (surface) and 5–10 cm (sub-surface); six randomly located samples were taken for each depth class within each plot and were composited by depth class and plot for analysis. Gravimetric soil water content was determined by the difference between field and oven dry weight. Soil texture was assessed using the hydrometer method (Gee and Bauder 1986); values were expressed on a percentage basis for sand, silt and clay fractions and textural class was assigned. Total soil carbon and nitrogen were determined using a LECO CN-2000 (LECO Corp., St. Joseph, MI). We used Plant Root Simulator (PRS) probes (Western Ag Innovations, Inc., Saskatoon, Canada) to monitor nutrient (N, NO_3, NH_4, H_2PO_4, and K) supply rate. Four cation and four anion probes were installed to approximately 5 cm depth in each plot in April of 2008 and collected in June 2008 for analysis by Western Ag Innovations. These probes consisted of a nutrient exchange membrane within a plastic stake. We installed probes so that the membrane was in contact with the top 5 cm of the soil profile. At Western Ag the probes were extracted using 0.5 N HCL for 1 h and extract was then analyzed for NH_4 and NO_3 using a Technicon Autoanalyzer II (Technicon Instrument Corporation 1977). Remaining nutrients were analyzed using ICP emission spectroscopy (Perkin Elmer Optima 3000-DV ICP; Perkin Elmer, Inc. Shelton, CT). We report values in units of μg/10 cm^2 (exchange membrane surface area)/80 days burial.

Statistical Analysis

Based on published data evaluating the relationship between perennial grass density and annual grass abundance (Davies 2008), our own experience, and the experience of other restoration practitioners, we defined a successful seeding effort as establishment of >5 seeded perennial grass plants/m^2. All plots were assigned values of successful (1) or unsuccessful (0) based on this criteria using total (native + non-native) density of seeded species in 2009 (i.e., two years post-seeding). We then used stepwise logistic regression (PROC LOGISTIC, SAS Institute Inc 1999) to determine the relationship between restoration success and the environmental variables measured in this study and to generate a predicted probability of revegetation success for each plot, based on soil or vegetation parameters. Vegetation (i.e., abundance of non-seeded vegetation) and soil variables were modeled separately, and the analyses were conducted using forward selection with an alpha level of 0.15 for variable entry into the model. "Non-seeded vegetation" included species that were not in the planted seed mix, as well as species that were in the planted mix, but occurred outside of the drill row. We used the CTABLE option to generate statistics indicating the percent of values correctly classified, the percent of event positive (sensitivity) and event negative (specificity) occurrences correctly predicted, and the percent occurrence of false positive and false negative predictions. We used a critical probability level of 0.5 to classify plots as successful or unsuccessful based on modeled probability (i.e., if estimated probability is >0.5 then the plot is classified as successful, <0.5 = unsuccessful). For variables included in the soil and vegetation models, we calculated a relative impact factor by dividing the regression coefficient by the average value of the variable; resulting values were then summed across all variables in each model, and the relative impact factor of each variable calculated as the percent of this total. To help understand the role of soil texture in influencing seedling success, we compared abundance of sand, silt and clay particles in surface and sub-surface samples with their respective values for soil water content in May (2008) using Pearson correlation coefficients (SAS Institute Inc 1999).

Cost per successful ha of revegetation was determined for each plot using the equation:

$$X = Y/Z$$

where X = cost of successful revegetation in dollars, Y = cost of drill seeding in dollars/ha and Z = predicted probability of success based on the soil regression model. The resultant figure indicates the financial expenditure necessary to produce one successful ha under a given set of environmental conditions. We used a drill seeding cost estimate of \$164.82; this is an average of \$95.06/ha for non-native perennial grasses and \$234.57/ha for native perennial grasses (Epanchin-Niell and others 2009). These estimates include cost for seed, equipment and cultural surveys. Because estimated cost per successful ha increased with decreasing restoration probability, extremely low probability values for a few plots (e.g., <0.01) can dramatically influence perception of restoration costs. To estimate costs at more broad scales, we averaged predicted probabilities of restoration success across plots within 500 m transect segments and divided drill-seeding costs/ha by this average probability of success. Means are reported with their associated standard errors.

Results

Density of seeded species in plots at 2 years post-seeding (2009) ranged from 0 to 24 plants/m^2 (Fig. 1) and 21 of 51 plots were successfully restored (>5 plants/m^2). Seeded non-natives were more abundant than seeded natives, making up about 94 % of the seeded species population by 2009 (Table 1). Of the natives species seeded, squirreltail was most successful, comprising about 4 % of the seeded species population (Table 1). Density of non-seeded plants in 2008 (Fig. 2) was highest for forbs (38.3 ± 6.2 plants/m^2) and lowest for shrubs (0.02 ± 0.01 plants/m^2). Of the non-seeded vegetation in 2008, curveseed butterwort (*Ceratocephala testiculata* (Crantz) Roth) was the dominant forb,

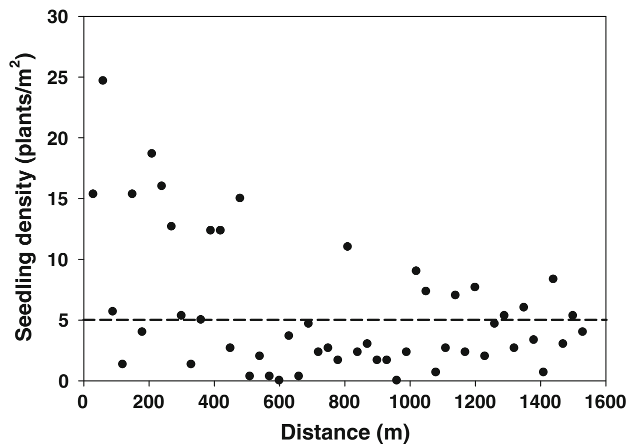

Fig. 1 Seedling density for seeded perennial grasses in 6 × 6 m plots spaced at 30 m intervals along a 1.5 km transect in Wyoming big sagebrush plant communities. Plots were drill seeded in October of 2007 following wildfire in August 2007. Density data were collected in July 2009

medusahead was the most abundant annual grass, and squirreltail the most prevalent perennial grass; Wyoming big sagebrush was the only shrub encountered in our plots (Table 2). By 2009, density of seeded perennial grasses averaged 5.7 ± 0.8 plants/m^2 (Fig. 2).

Value ranges for the 28 soil-related and 10 vegetation-related environmental variables measured in this study are listed in Table 3. Five variables (K supply rate, surface N (%), surface clay (%), sub-surface silt (%), and loam sub-surface texture) met the critical chi-square *P* value (<0.15) for inclusion in the regression equation for soil-related environmental variables. All variables in the model were associated positively with restoration success. Relative impact of variables included in the model varied strongly. Relative impact of soil surface N was 88.2 %, followed by 11.8 % for sub-surface loam texture; remaining variables in the model were <0.001 %. The overall model was significant based on the model chi-square statistic ($P = 0.005$). Because it was redundant with NO_3 and total nitrogen, NH_4 was not used in regression analysis. The regression equation correctly predicted restoration performance (i.e., successful or unsuccessful) for 82.4 % of plots. Model sensitivity was 71.4 % and specificity was 90 %. Percent false positive and negative predictions were 16.7 and 18.2 %, respectively. Cover of non-seeded large perennial grasses was negatively associated with restoration success and was the only variable to meet the significance criteria for inclusion in the model for vegetation-related environmental variables (Table 3). Canopy cover of these species averaged 1.84 % (±0.34) for successful plots and 3.07 % (±0.43) for unsuccessful plots. The vegetation model ($P = 0.041$) correctly predicted restoration performance for 51.0 % of plots. Model sensitivity was 28.6 %

Table 1 Seeding rate and 2 years post-planting (2009) population data for seeded species in low elevation Wyoming big sagebrush plant communities in southeastern Oregon

Species	Seeding rate (kg/ha)	% in mix (by weight)	% of population[b] (2009)
Crested wheatgrass[a]	6.84	60	93.89
Secar bluebunch wheatgrass	2.28	20	1.80
Great basin wildrye	1.14	10	0.35
Squirreltail	1.14	10	3.96

[a] Includes both *A. cristatum* and *A. sibiricum*

[b] Percentage based on seedling density of planted species

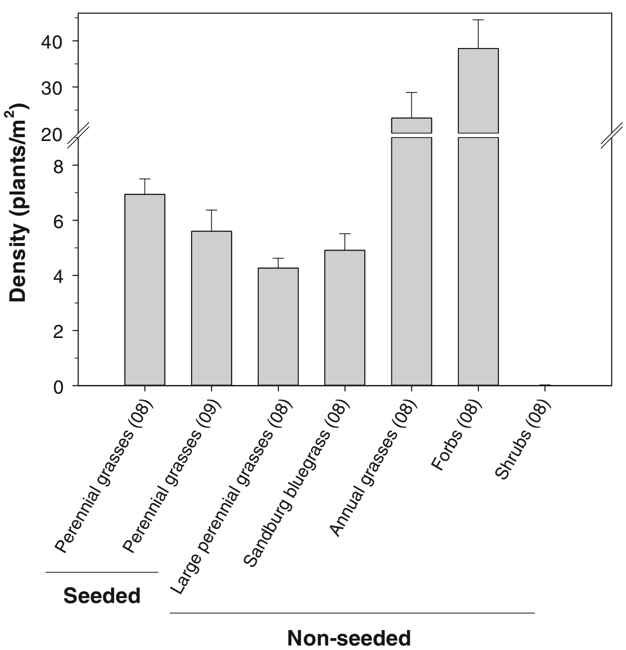

Fig. 2 One and 2 years post-fire density of seeded perennial grasses and 1 year post-fire density of non-seeded vegetation following an August 2007 wildfire. Data averaged across 6 × 6 m plots spaced at 30 m intervals along a 1.5 km transect in Wyoming big sagebrush plant communities in southeast Oregon

and specificity was 66.7 %. Percent false positive and negative predictions were 62.5 and 42.9 %, respectively.

Surface soil water content was associated negatively with surface sand content and positively with silt (weakly) and clay content (Fig. 3). Sub-surface soil moisture decreased with sand content ($r = -0.32$, $P = 0.02$) and was unrelated to silt ($r = 0.19$, $P = 0.33$) and clay ($r = 0.21$, clay $= 0.14$) content.

Predicted probabilities of revegetation based on the soil model ranged from 0.003 to 0.99 (Fig. 4a). The cost of successful restoration varied strongly between plots and ranged from $167.06 to $43,033.94/ha (Fig. 4b). At the 500 m scale, cost estimates ranged from $247.81 to $695.44/ha (Fig. 4c).

Discussion

In the annual grass-prone Wyoming big sagebrush alliance, post-disturbance re-establishment of perennial grasses is critical for maintaining ecological integrity and decreasing the probability of annual grass dominance (Davies 2008). However, restoration success can be influenced by a wide variety of environmental factors that vary spatially. The data and methodology presented here indicate that environmental variables can serve as accurate predictors of revegetation success across a landscape. While others (e.g., Eiswerth and others 2009) have used regression-based approaches to determine the relative influence of environmental factors on restoration success, to our knowledge, this study represents the first successful attempt to assign actual revegetation probabilities along a spatial gradient. Additionally, the quantitative nature of these predictions allowed us to estimate associated restoration costs.

Micro-scale efforts to associate environmental variables with restoration success (e.g., Abella and others 2009; Boyd and Davies 2010) may have limited value at scales used in application of management treatments. Conversely, associating broad spatial scale variables with restoration success (e.g., Pyke and Knick 2005) can be too coarse to uncover important predictors of restoration success at finer scales. Here we have taken an intermediate approach and aggregated plot-level results to scales that have utility within the context of restoration and identified environmental variables that correlate with restoration success.

Soil variables influencing restoration success in the current study were all related to texture and nutrient content/availability. Similarly, Davies and others (2007a) found that soil characteristics were the most common environmental variables associated with differences in vegetation characteristics across the Wyoming big sagebrush alliance. Others have reported decreased abundance of native perennial seedlings in association with high soil nutrient concentrations (see review by Vasquez and others 2008a). Generally, this inverse relationship takes place within the context of a high abundance of non-native annual grasses, which have been shown to respond positively to increases in soil nutrients (Vasquez and others 2008b) and can outcompete native perennial seedlings (Young and Mangold 2008). Annual grass density in our study plots averaged 23.3 (+/−5.6) plants/m^2. Annual grass densities of five to six times this amount have been reported at nearby sites (Sheley and others 2008) and Davies and Svejcar (2008) reported densities 30 times this level in an invaded Wyoming big sagebrush community. Thus, in our case, the lack of influence of annual grass abundance on revegetation success (Table 3) may be a consequence of relatively low initial density of annual grasses post-fire.

Table 2 Density of non-planted vegetation at 1 year post-fire (2008) for Wyoming big sagebrush communities in southeastern Oregon

Functional group	Common name	Scientific name	Density (no./m^2)	Standard error
Annual grasses	Medusahead	*Taeniatherum caput-medusae* (L.) Nevski	18.00	5.47
	Cheatgrass	*Bromus tectorum* L.	4.05	0.97
	Ventenata	*Ventenata dubia* (Leers) Coss.	1.20	0.33
Forbs	Curveseed butterwort	*Ceratocephala testiculata* (Crantz) Roth	18.53	5.03
	Rough eyelashweed	*Blepharipappus scaber* Hook.	3.64	0.55
	Desert madwort	*Alyssum desertorum* Stapf	3.36	0.95
Sandberg bluegrass	Sandberg bluegrass	*Poa secunda* J. Presl	4.91	0.59
Large perennial grasses	Squirreltail	*Elymus elymoides* (Raf.) Swezey	1.43	0.25
	Bluebunch wheatgrass	*Pseudoroegneria spicata* (Pursh) A. Löve	1.37	0.29
	Western wheatgrass	*Pascopyrum smithii* (Rydb.) A. Löve	0.86	0.22
Shrubs	Wyoming big sagebrush	*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle and A. Young	0.02	0.01

Values are listed for the 3 most abundant species in each functional group with the exception of "Shrubs", which had only one species present, and Sandburg bluegrass, which is a single species. These data include species that were not in the planted seed mix, as well as species that were in the planted mix, but occurred outside of the drill row

The abundance of non-seeded large perennial grasses was negatively associated with restoration success (Table 3). This finding is in agreement with previous empirical work indicating that post-fire competition from surviving perennial grasses may limit success of seeded species, and highlights the importance of developing methodology for accurately assessing fire-related mortality of perennial grasses when determining the need for post-fire seeding (Ratzlaff and Anderson 1995; James and Svejcar 2010). Our data suggest that the importance of vegetation competition (i.e., from non-seeded species) was less than that of soil-related environmental factors in predicting eventual revegetation success. The model for soil factors correctly predicted restoration performance for 82.4 % of plots as compared to 51 % for the vegetation model. Additionally the specificity and sensitivity values for the vegetation model (66.7 and 28.6 %, respectively) suggest that the predictive capacity of this model was heavily skewed toward correctly predicting restoration failure, as opposed to restoration success. In other words, relatively high abundance of non-seeded perennial grasses was generally associated with revegetation failure, but relatively low abundance did not necessarily portend revegetation success. In contrast, specificity and sensitivity values for the soil model (90 % vs. 71.4) indicate that it more equitably predicted revegetation successes and failure.

In our study, soils with smaller or mixed particle sizes had an increased probability of revegetation success. This relationship may relate to correlations between particle size and elevated soil water content in the first growing season following fire (2008). As surface sand content increased, soil water (May) decreased (Fig. 3). Conversely, water content increased with abundance of silt and clay particles (Fig. 3). Soils composed of smaller particle sizes have increased pore space available for water storage and elevated water holding capacity. Adequate soil water is critical for germination (Hardegree and Van Vactor 2004; Hardegree and others 2003) and establishment (Hironaka and Sindelar 1975) of perennial grasses. Additionally, smaller particle size in the subsoil may restrict infiltration and increase water content of the surface horizon (Eiswerth and others 2009).

Our rationale for using a binomial response variable (i.e., success or no success) for plot scores as opposed to a continuous variable such as density of seeded species was two-fold. First, we felt that, pragmatically speaking, revegetation attempts will ultimately be recorded as successes or failures. Thus, the binomial response matches the expectations of management professionals. Secondly, the high level of between-plot variability in seedling density (Fig. 1) may have obscured relationships with environmental variables. Using a binomial response variable decreases the degree of variability and makes it easier to define relationships between revegetation success and environmental conditions.

The high plot-to-plot variability in estimates of probability of revegetation success (Fig. 4a) is reflective of the inherent difficulty associated with determining where to seed desired perennial species on arid sagebrush rangeland. While current research efforts are making progress in disentangling the ecological mechanics that drive such variability (e.g., Chambers and others 2007; Hardegree and Van Vactor 2004; Krueger-Mangold and others 2006); Boyd and Svejcar (2009) argue that restoring Wyoming big

Table 3 Value ranges in 2008 for soil and vegetation-related environmental variables in seeded plots within low elevation Wyoming big sagebrush plant communities in southeastern Oregon

Variable class	Variable type	Variable name	*n*	Mean	Standard error	Minimum	Maximum	χ^2 probability	Coefficient	Relative impact factor
Soil crust	Continuous	Surface compaction (J)	51	31.09	2.376	5.88	80.37	>0.15	na	na
Soil nutrient	Continuous	N supply rate (μg/10 cm^2/80 days)	51	126.72	11.411	19.40	337.40	>0.15	na	na
Soil nutrient	Continuous	no3 supply rate (μg/10 cm^2/80 days)	51	126.44	11.421	17.60	337.40	>0.15	na	na
Soil nutrient	**Continuous**	**K supply rate (μg/10 cm^2/80 days)**	**51**	**48.87**	**3.290**	**14.00**	**115.60**	**0.011**	**0.052**	**<0.001**
Soil nutrient	Continuous	P supply rate (μg/10 cm2/80 days)	51	18.77	1.175	4.00	37.40	>0.15	na	na
Soil nutrient	**Continuous**	**Surface N (%)**	**51**	**0.11**	**0.004**	**0.06**	**0.18**	**0.011**	**54.27**	**88.2**
Soil nutrient	Continuous	Surface C (%)	51	1.16	0.050	0.54	2.06	>0.15	na	na
Soil nutrient	Continuous	Sub-surface N (%)	51	0.11	0.004	0.05	0.18	>0.15	na	na
Soil nutrient	Continuous	Sub-surface C (%)	51	1.09	0.059	0.49	2.73	>0.15	na	na
Soil nutrient	Continuous	Surface C:N	51	9.47	0.241	6.35	16.69	>0.15	na	na
Soil nutrient	Continuous	Sub-surface C:N	51	9.97	0.229	5.20	15.66	>0.15	na	na
Soil texture	Continuous	Surface sand (%)	51	24.33	1.110	8.19	45.54	>0.15	na	na
Soil texture	**Continuous**	**Surface clay (%)**	**51**	**31.15**	**1.024**	**15.59**	**49.25**	**0.002**	**0.38**	**<0.001**
Soil texture	Continuous	Surface silt (%)	51	44.51	0.896	26.25	58.00	>0.15	na	na
Soil texture	Continuous	Sub-surface sand (%)	51	23.27	1.211	9.73	50.00	>0.15	na	na
Soil texture	continuous	Sub-surface clay (%)	51	32.47	1.187	11.75	49.00	>0.15	na	na
Soil texture	**Continuous**	**Sub-surface silt (%)**	**51**	**44.26**	**0.997**	**25.62**	**58.60**	**0.007**	**0.303**	**<0.001**
Soil texture	Discrete	Surface texture = clay	51	0.06	0.033	0	1	>0.15	na	na
Soil texture	Discrete	Surface texture = clay loam	51	0.33	0.067	0	1	>0.15	na	na
Soil texture	Discrete	Surface texture = mixed loam	51	0.16	0.051	0	1	>0.15	na	na
Soil texture	Discrete	Surface texture = silty clay loam	51	0.29	0.064	0	1	>0.15	na	na
Soil texture	Discrete	Surface texture = silty loam	51	0.12	0.046	0	1	>0.15	na	na
Soil texture	Discrete	Sub-surface texture = clay	51	0.10	0.042	0	1	>0.15	na	na
Soil texture	Discrete	Sub-surface texture = clay loam	51	0.24	0.060	0	1	>0.15	na	na
Soil texture	**Discrete**	**Sub-surface texture = loam**	**51**	**0.10**	**0.042**	**0**	**1**	**0.003**	**6.62**	**11.8**
Soil texture	Discrete	Sub-surface texture = silty clay	51	0.08	0.038	0	1	>0.15	na	na
Soil texture	Discrete	Sub-surface texture = silty clay loam	51	0.31	0.066	0	1	>0.15	na	na
Soil texture	Discrete	Sur-surface texture = silty loam	51	0.18	0.054	0	1	>0.15	na	na
Vegetation	**Continuous**	**Large perennial grass cover**	**51**	**2.57**	**0.300**	**0**	**9.00**	**0.052**	**−0.324**	**100**
Vegetation	Continuous	Large perennial grass density	51	4.26	0.360	0	11.50	>0.15	na	na
Vegetation	Continuous	*Poa* cover	51	1.40	0.165	0	4.20	>0.15	na	na
Vegetation	continuous	*Poa* density	51	4.91	0.595	0	16.50	>0.15	na	na
Vegetation	Continuous	Annual grass cover	51	3.69	0.710	0	26.20	>0.15	na	na

Table 3 continued

Variable class	Variable type	Variable name	n	Mean	Standard error	Minimum	Maximum	χ^2 probability	Coefficient	Relative impact factor
Vegetation	Continuous	Annual grass density	51	23.25	5.557	0	184.50	>0.15	na	na
vegetation	Continuous	Forb cover	51	5.73	0.480	0.80	12.90	>0.15	na	na
Vegetation	Continuous	Forb density	51	38.32	6.228	1.50	231.50	>0.15	na	na
Vegetation	Continuous	Shrub cover	51	0	0	0	0.00	>0.15	na	na
Vegetation	Continuous	Shrub density	51	0.01	0.010	0	0.68	>0.15	na	na

The influence of soil and vegetation variables on revegetation success was modeled separately. Vegetation variables describe abundance for non-seeded species. Bold entries denote variables included in the final model

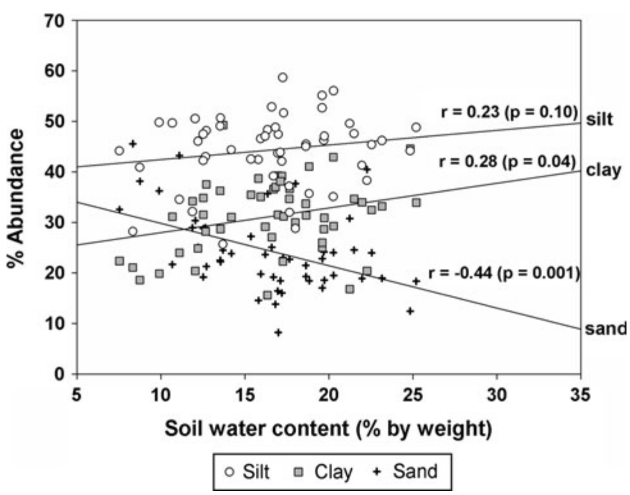

Fig. 3 Relationship between abundance of soil particle size class in the surface horizon and soil water content for low elevation Wyoming big sagebrush plant communities in southeast Oregon

sagebrush rangeland is a complex problem, and that the ecological drivers will vary in time and space. Thus, identifying spatial patterns of variability in restoration success, along with their environmental correlates, represents a viable approach for increasing odds of successful restoration in a complex environment.

Our study is limited by the fact that we employed a single study site and seeded in only one year. Accordingly, caution should be exercised when extrapolating our results to other areas. The specific variables and the regression coefficients reported here may or may not apply to other areas in the sagebrush steppe. For example, soil texture was an important predictor of restoration success in the present study, however, the range of soil textures we encountered may not be representative of that found at other locations. That said, the high variability in restoration success and environmental conditions reported here is consistent with other published findings in Wyoming big sagebrush ecosystems (Davies and others 2007b; Eiswerth and others 2009). The ultimate value of our work lies less in the quantitative importance of specific environmental variables to restoration success, and more in providing an ecologically-based methodology for estimating restoration probability. These probabilities have obvious value to restoration practitioners, but will also help researchers by providing an indirect means of quantifying incremental improvements in restoration techniques. Our work suggests that small improvements in the effectiveness of restoration techniques can have a dramatic influence on cost per successful ha of restoration. For example, on sites with low probability (e.g., 0.1) of restoration success, an increase of 0.1 in the probability of revegetation will cut the cost/unit area of successful restoration in half.

Implicit in our revegetation cost estimates is the assumption that these numbers, which are calculated based on plots arrayed unidirectionally over the landscape, will apply in two-dimensional space. In other words, the plots represent a suite of environmental conditions that can potentially apply to a larger area. One limitation of our data set is that the cost estimates, and the probabilities on which they are based, were developed over space and their applicability over multiple years is unknown. Further research that incorporates the interactive role of time in influencing restoration probabilities will greatly increase the utility of regression-based approaches for predicting restoration success.

In reality, the high end of our estimated cost continuum is unrealistic given that management agencies using our approach would likely set the lower restoration probability thresholds at levels higher than 0.0038 (the level required to produce our maximum estimated cost of \$43,033.94). Averaging restoration probabilities across a broader scale of analysis represents a more viable and practical method of assessing predicted cost of successful restoration. When values were averaged within 500 m increments, variation in predicted revegetation costs was almost three-fold. These data suggest strong spatial variability in cost of successful restoration at scales large enough to be

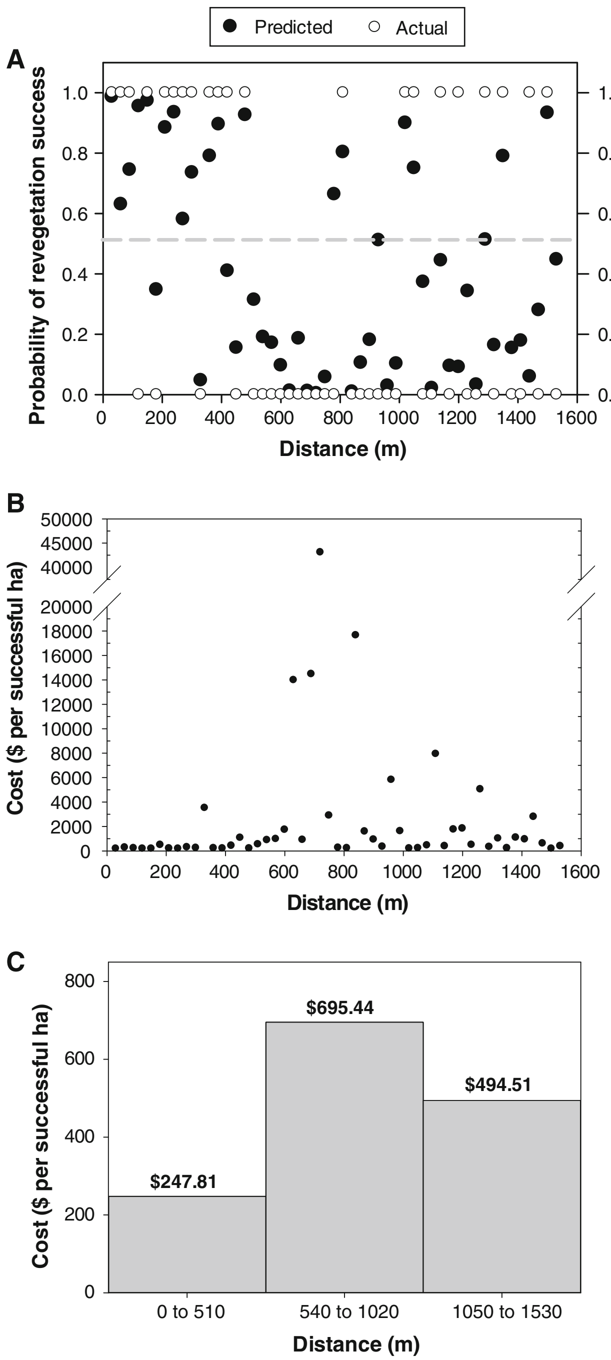

Fig. 4 Predicted probability of revegetation success based on soil factors (**a**) and estimated cost per successful (>5 plants/m^2) ha of revegetation at 30 m (**b**) and 500 m (**c**) intervals for Wyoming big sagebrush plant communities in southeast Oregon. In the *top graph*, predicted plot values ≥0.5 (*dashed line*) were considered successfully revegetated and values less <0.5 were considered unsuccessful

incorporated into management planning and implementation. Thus, when capital or logistical resources are limited our approach could be used to select the areas that have the highest probability of being successfully restored within a larger degraded landscape.

Conclusions

Restoration and rehabilitation of Wyoming big sagebrush rangeland is set within a spatially variable environment. This is in contrast to management practices, such as post-fire seeding, that are typically applied in a uniform manner across a management unit. The techniques described in this paper will help to identify important environmental drivers of restoration success as well as the financial ramifications of variability in these drivers over space. This process could be of particular value within the context of an adaptive management program to iteratively refine protocols for spatially allocating management effort and capital in a variable restoration environment. Additional research at alternate locations and years is needed to determine the range of conditions under which the use of the approach described here may be of value in predicting restoration success. Alternatively, important variables identified in the current study can be viewed as suggestive of underlying ecological processes. Further research to isolate such processes could increase the applicability of the current effort to alternate locations. Over time, increases in our ability to relate environmental variables to restoration potential may allow for the development of *a priori* models for guiding optimal spatial distribution of restoration effort.

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References

Abella SR, Engel EC, Lund CL, Spencer JE (2009) Early post-fire plant establishment on a Mojave Desert burn. Madroño 56:137–148

Boyd CS, Davies KW (2010) Shrub microsite influences post-fire perennial grass establishment. Rangeland Ecology and Management 63:248–252

Boyd CS, Svejcar TJ (2009) Managing complex problems in rangeland ecosystems. Rangeland Ecology and Management 62:491–499

Brooks ML, D'Antonio CM, Richardson DM, Grace JB, Keeley JE, DiTomaso JM, Hobbs RJ, Pellant M, Pyke D (2004) Effects of invasive alien plants on fire regimes. Bioscience 54:677–688

Chambers JC, Roundy BA, Blank RR, Meyer SE, Whittaker A (2007) What makes Great Basin sagebrush ecosystems invasible by *Bromus tectorum*? Ecological Monographs 77:114–145

D'Antonio CM, Vitousek PM (1992) Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Reviews in Ecology and Systematics 23:63–87

Davies KW (2008) Medusahead dispersal and establishment in sagebrush steppe plant communities. Rangeland Ecology and Management 61:110–115

Davies KW, Svejcar TJ (2008) Comparison of medusahead-invaded and noninvaded Wyoming big sagebrush steppe in southeastern Oregon. Rangeland Ecology and Management 61:623–629

Davies KW, Bates JD, Miller RF (2007a) Environmental and vegetation relationships of the *Artemisia tridentata* spp. *wyomingensis* alliance. Journal of Arid Environments 70:478–494

Davies KW, Bates JD, Miller RF (2007b) Short-term effects of burning Wyoming big sagebrush steppe in southeast Oregon. Rangeland Ecology and Management 60:515–522

Drenovsky RE, Martin CE, Falasco MR, James JJ (2008) Variation in resource acquisition and utilization traits between native and invasive perennial forbs. American Journal of Botany 95:681–687

Eiswerth ME, Krauter K, Swanson SR, Zielinski M (2009) Post-fire seeding on Wyoming big sagebrush ecological sites: regression analyses of seeded nonnative and native species densities. Journal of Environmental Management 90:1320–1325

Epanchin-Niell R, Englin J, Nalle D (2009) Investing in rangeland restoration in the arid west, USA: countering the effects of an invasive weed on the long-term fire cycle. Journal of Environmental Management 91:370–379

Gee GW, Bauder JW (1986) Particle-size analysis. In: Klute A (ed) Methods of soil analysis, Part 1: physical and mineralogical methods. American Society of Agronomy, Madison, pp 383–411

Hardegree SP, Van Vactor SS (2004) Microclimatic constraints and revegetation planning in a variable environment. Weed Technology 18:1213–1215

Hardegree SP, Flerchinger GN, Van Vactor SS (2003) Hydrothermal germination response and the development of probabilistic germination profiles. Ecological Modeling 167:305–322

Herrick JE (2005) Comments and letters to the editor. Soil Science Society of America Journal 69:925–927

Herrick JE, Jones TL (2002) A dynamic cone penetrometer for measuring soil penetration resistance. Soil Science Society of America Journal 66:1320–1324

Hironaka M, Sindelar BW (1975) Growth Characteristics of squirreltail seedlings in competition with medusahead. Journal of Range Management 28:283–285

Hull AC (1974) Species for seeding arid rangeland in southern Idaho. Journal of Range Management 27:216–218

Humphrey LE, Schupp EW (2004) Competition as a barrier to establishment of a native perennial grass (*Elymus elymoides*) in alien annual grass (*Bromus tectorum*) communities. Journal of Arid Environments 58:405–422

James JJ, Drenovsky RE (2007) A basis for relative growth rate differences between native and invasive forb seedlings. Rangeland Ecology and Management 60:395–400

James JJ, Svejcar T (2010) Limitations to postfire seedling establishment: the role of seeding technology, water availability, and invasive plant abundance. Rangeland Ecology and Management 63:491–495

Knapp PA (1996) Cheatgrass (*Bromus tectorum* L.) dominance in the Great Basin Desert. Global Environmental Change 6:37–52

Knutson KC, Pyke DA, Wirth TA, Pilliod DS, Brooks ML, Chambers JC (2009) A chronosequence feasibility assessment of emergency fire rehabilitation records within the Intermountain Western United States: Final Report to the Joint Fire Science Program—Project 08-S-08: U.S. Geological Survey Open-File Report 2009-1099

Krueger-Mangold JM, Sheley RL, Svejcar TJ (2006) Toward ecologically-based invasive plant management on rangeland. Weed Science 54:597–605

Lysne CR, Pellant M (2004) Establishment of aerially seeded big sagebrush following southern Idaho wildfires. Department of the Interior, Bureau of Land Management, Technical Bulletin 2004-01, Boise

Mack R (1981) Invasion of *Bromus tectorum* L. into Western North America: an ecological chronicle. Agro-Ecosystems 7:145–165

Melgoza G, Nowak RS, Tausch RJ (1990) Soil water exploitation after fire: competition between *Bromus tectorum* (cheatgrass) and two native species. Oecologia 83:7–13

Miller RF, Eddleman LL (2000) Spatial and temporal changes of sage grouse habitat in the sagebrush biome. Technical Bulletin 151. Oregon State University, Agricultural Experiment Station, Corvallis

Miller H, Clausnitzer CD, Borman MM (1999) Medusahead. In: Sheley RL, Petroff JK (eds) Biology and management of noxious rangeland weeds. Oregon State University Press, Corvallis, pp 271–281

Mosley JC, Bunting SC, Manoukian ME (1999) Cheatgrass. In: Sheley RL, Petroff JK (eds) Biology and management of noxious rangeland weeds. Oregon State University Press, Corvallis

NRCS (2007) Soil survey.http://websoilsurvey.nrcs.usda.gov/

Oregon Climatic Service (2007). http://www.ocs.oregonstate.edu/indexhtml

Pellant M, Abbey B, Karl S (2004) Restoring the Great Basin desert, U.S.A.: integrating science, management and people. Environmental Monitoring and Assessment 99:169–179

Pyke DA, Knick ST (2005) Plant invaders, global change and landscape restoration. African Journal of Range and Forage Science 22:75–83

Ratzlaff TD, Anderson JE (1995) Vegetal recovery following wildfire in seeded and unseeded sagebrush steppe. Journal of Range Management 48:386–391

Richards RT, Chambers JC, Ross C (1998) Use of native plants on federal lands: policy and practice. Journal of Range Management 51:625–632

SAS Institute Inc. (1999) SAS procedures guide, release 8.0. SAS Institute. Cary

Sheley RL, Bingham BS, Svejcar TJ (2008) Crested wheatgrass defoliation intensity and season on medusahead invasion. Rangeland Ecology and Management 61:211–217

Stringham TK, Krueger WC, Shaver PL (2003) State and transition modeling: an ecological process approach. Journal of Range Management 56:106–113

Technicon Instrument Corporation (1977) Individual/simultaneous determinations of nitrogen and (or) phosphorus in BD acid digests: industrial method No. 329–74W. TIC, Tarrytown

U.S. Department of the Interior (2004) Burned area emergency stabilization and rehabilitation. http://elips.doi.gov/app_DM/act_getfiles.cfm?relnum=3610

Vasquez E, Sheley R, Svejcar T (2008a) Creating invasion resistant soils via nitrogen management. Invasive Plant Science and Management 1:304–314

Vasquez E, Sheley R, Svejcar T (2008b) Nitrogen enhances the competitive ability of cheatgrass (*Bromus tectorum*) relative to native grasses. Invasive Plant Science and Management 1: 287–295

Whisenant SG (1990) Changing fire frequencies in Idaho's Snake River Plains: ecological and management implications. In: McArthur ED, Romney EM, Smith SD, Tueller PT (eds) Symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. Las Vegas, NV

Young K, Mangold J (2008) Medusahead outperforms squirreltail through interference and growth rate. Invasive Plant Science and Management 1:73–81