AN ABSTRACT OF THE THESIS OF

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Title: An Evaluation of the Invasion Potential of *Kochia prostrata* (forage kochia) in Southwestern Idaho, USA

Abstract approved:

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Purposeful introductions of exotic species for rehabilitation efforts following wildfire are common on rangelands in the western U.S., though potential ecological impacts of introduced species in novel environments are often poorly understood. One such introduced species, *Kochia prostrata* (L.) Schrad (forage kochia) has been seeded on over 200,000 ha throughout the Intermountain West to provide fuel breaks and forage, and to compete with invasive plants. Despite its potential benefits, it has been reported to spread from some seeded areas, and no studies have addressed its potential interactions with native species. A systematic investigation is needed to increase understanding of the extent to which *K. prostrata* spreads from seeded areas, the environmental conditions under which it spreads, and its potential interactions with the associated plant communities. I sampled 28 *K. prostrata* post-fire rehabilitation and greenstrip seedings in southwestern Idaho, which ranged from 3 to 24 yr since seeding. I analyzed cover of *K. prostrata* and the associated plant community in paired seeded and unseeded areas, and quantified extent of spread from seeded areas. Sites seeded with *K. prostrata* had lower species richness and Shannon-Wiener diversity than adjacent unseeded areas. Abundance of *K. prostrata* was negatively associated with that of most plant functional groups, including all native species, but was positively associated with abundance of exotic annual forbs. *Kochia prostrata* spread to unseeded areas on 89 % of sampled sites; distances of the farthest individual from the seeding boundary were greater than those previously reported, ranging from 0 to 710 m, with a mean distance of 208 m. Further, while spread increased with time since seeding, it was apparently independent of the composition of communities into which spread occurred. This study improves understanding of the ecological implications of seeding *K. prostrata*, demonstrating potential to spread from

seeded areas into native plant communities, and that *K. prostrata* does not seem to enhance native species diversity in seeded areas over time. These findings will enable land managers to make scientifically-based decisions regarding use of this species.

An Evaluation of the Invasion Potential of *Kochia prostrata* (forage kochia) in Southwestern Idaho, USA

by
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INTRODUCTION

What ecological implications are associated with the introduction of a highly competitive, exotic plant species for management purposes? What potential costs and benefits should be considered when selecting species for widespread seeding? What impacts might introduced species have on nearby native plant communities? Land managers faced with the restoration and rehabilitation of highly-degraded, exotic species-invaded, sagebrush steppe rangelands of the western U.S. need answers to these questions. I addressed these issues using a case study of *Kochia prostrata* (L.) Schrad (forage kochia), which is an introduced species widely seeded throughout western rangelands to stabilize soils, provide fuel breaks and forage, and compete with invasive annual plant species. I assessed its spread from sites onto which it had been seeded in Idaho, and its relationships to associated plant communities in seeded and adjacent unseeded areas. Results contribute to current understanding of ecological implications of seeding *K. prostrata* and will enhance the ability of land managers to make scientifically-based management decisions about its use.

Sagebrush steppe communities in the Intermountain West once covered a vast expanse including the Great Basin and Columbia Plateau. Over the past century, systems that were historically described as "sagebrush sea" have been altered and degraded into exotic grasslands (Meinke et al. 2009); it is estimated that 55 % of historical sagebrush steppe remains (Miller et al. 2011). Prior to the 1800's, wildfire was often lightning caused and the fire-return interval is estimated to have been approximately 20-100 years (Whisenant 1990; Brooks and Pyke 2001). European settlement in the mid-nineteenth century brought with it exotic plant species and uncontrolled grazing by non-native ungulates. A combination of heavy grazing and droughts in the early 1900's opened the landscape to invasion by exotic species (Pyke 1999; Brooks and Pyke 2001). Aggressive annual species such as *Bromus tectorum* L. (cheatgrass) and *Taeniatherum caput-medusae* (L.) Nevski (medusahead)

outcompeted native perennial bunchgrasses for limited water and nutrients and invaded interspaces between bunchgrasses and shrubs, areas historically occupied by biological soil crusts (D'Antonio and Vitousek 1992; Pyke 1999). This created a continuous and highly flammable fuel base (Brooks et al. 2004), resulting in more frequent wildfire than the historical regime (Brooks and Pyke 2001). This annual grass/fire cycle can perpetuate itself, with grasses providing fine fuels and the fire providing open spaces for invasion (D'Antonio and Vitousek 1992). Currently, much of the Great Basin and Columbia Plateau is composed of a patchy mosaic of differing vegetation communities resulting from unique site conditions, disturbance and management histories.

Restoration and rehabilitation efforts in the Intermountain West are met by immense challenges. The spatial extent of land converted from historic shrub steppe is vast and the resources required to attempt restoration and rehabilitation on such a scale are lacking (Wisdom and Chambers 2009). Rangeland managers, instead, typically implement localized projects focused on post-fire and fuels management treatments (Pellant et al. 2004). Success of rehabilitative efforts that involve seeding can be limited by water availability (Jessop and Anderson 2007), invasion by exotic species, frequent fire (Eiswerth et al. 2009), and problems with seeding equipment (James and Svejcar 2010). Each year, post-fire rehabilitation efforts occur on thousands of hectares of public lands in this region, the majority of which are managed by the United States Department of the Interior (USDI) Bureau of Land Management (BLM).

In an attempt to prevent the rapid conversion of recently burned areas to highly-flammable exotic grasslands, the BLM modified its Emergency Fire Rehabilitation (EFR) Program in 1985 to focus on post-fire seeding to promote re-vegetation of beneficial species and decrease soil erosion (Pellant 1994; McArthur 2004; Eiswerth et al. 2009). Perennial exotic species, such as *Agropyron cristatum* (L.) Gaertn. (crested wheatgrass) were often seeded in EFR projects due to their greater establishment success and seed availability when compared to natives (Jones 1999; Brooks and Pyke 2001; McArthur 2004), and to their high forage value (Pellant and Monsen 1993). While use of native species in post-fire rehabilitation is advocated, exotic species continue to be seeded regularly because the supply of native seed is limited and seed is expensive (Pellant and Monsen 1993; Richards et al. 1998). For each proposed rehabilitation plan, managers must justify use of exotic species;

justifications must account for their potential impacts on biodiversity and must document their inability to spread from seeded areas or to interbreed with native species (USDI BLM 2007).

In an attempt to prevent spread and impact of wildfires on public land, the Idaho BLM initiated the Greenstripping Program, in which relatively fire-resistant vegetation was planted to create fuel breaks along roads or areas of concern. The Greenstripping Program occurred from 1985 to 1993, and involved seeding 726 km of greenstrips on BLM land with fire resistant species (Pellant 1994). Species were selected for greenstripping based upon their fire resistance, drought tolerance, capability of establishing and persisting, and palatability (Pellant 1994).

Kochia prostrata (also known as *Bassia prostrata* [L.] A.J. Scott), a long-lived exotic semi-evergreen sub-shrub, has been seeded in post-fire rehabilitations and greenstripping programs throughout the Intermountain West (Harrison et al. 2000). *Kochia prostrata* is native to arid and semi-arid regions of central Asia, eastern Europe and north Africa (Balyan 1972; USDA-ARS GRIN 2001). The variety "Immigrant" (*K. prostrata* ssp. *virescens*) was acquired in 1966 as a plant potentially capable of outcompeting *Halogeton glomeratus* (Bieb.) C.A. Mey (halogeton), an exotic species, on xeric sites (Harrison et al. 2000). "Immigrant" *K. prostrata* was first planted in the U.S. in 1968 and was released for use by the United States Department of Agriculture (USDA) Agricultural Research Service (ARS) for soil stabilization in 1984 (Waldron et al. 2001). By 2004, *K. prostrata* had been seeded on an estimated 202,000 ha throughout the Intermountain West (Harrison 2004). *Kochia prostrata* is adapted to a variety of plant community types including salt desert, sagebrush steppe, and pinyon-juniper woodlands (Stevens et al. 1985; Romo and Haferkamp 1987). It can grow under a wide range of moisture regimes, though it does best in areas receiving 150 to 300 mm of precipitation annually (Stevens et al. 1985). Hardy and drought tolerant, *K. prostrata* can withstand temperatures from -32 to 40°C and elevations ranging from 490 to 2,200 m in the Great Basin (Harrison 2004). *Kochia prostrata* occurs on many soil types and can tolerate highly saline soils (Stevens et al. 1985; Romo and Haferkamp 1987).

Kochia prostrata has many attributes that make it a candidate for use in rangeland rehabilitation efforts. It is most well known for its abilities to germinate in late winter (Balyan 1972), establish on degraded sites, and compete with aggressive exotic species such as *B. tectorum* and *T. caput-medusae* (McArthur et al. 1990; Monaco et al. 2003). Though its

competitive mechanisms are not fully understood, *K. prostrata* germinates earlier than most native species and has an extensive fibrous root system and a taproot that can extend up to 6.5 m in depth, which may enable it to utilize early season soil moisture (Romo and Haferkamp 1987; Waldron et al. 2010a). *Kochia prostrata* produces thousands of seeds per plant shortly after establishment, though full seed production often begins 3 yr thereafter (Harrison et al. 2000). Though the duration of seed viability is limited, *K. prostrata* can germinate asynchronously, potentially increasing its chance of successful establishment (Kitchen and Monsen 2001). Seeds are wind dispersed, and recruitment is often in the direction of prevailing winds, within interspaces between other plants, or in soil depressions (Stevens et al. 1985; Stevens and McArthur 1990; Harrison et al. 2000). As a semi-evergreen species, *K. prostrata* is palatable to cattle and wildlife such as mule deer, primarily during fall and winter when other forage species are unavailable (Gray 2004; Waldron et al. 2010a). Some consider it beneficial to seed *K. prostrata* along with grasses, as cattle may prefer a mixed diet (Waldron et al. 2010b). Its verdant nature contributes to its fire resistance; it is thought to withstand fire and, though it is reported to burn when surrounded by ignitable fuels, it has potential to re-sprout quickly (Pellant 1994, Harrison et al. 2002).

Although use of *K. prostrata* in rehabilitation offers potential benefits, there are also concerns associated with the introduction of a highly competitive, exotic species into severely degraded rangelands. All of the mechanisms that enable *K. prostrata* to be an aggressive competitor against exotic annuals could potentially incur negative impacts on native species. *Kochia prostrata*'s spatial and temporal capacity for water uptake (Romo and Haferkamp 1987) is thought to prevent establishment of annuals and some perennials (Harrison et al. 2000), and its prolific seed production enables recruitment in both annual- and perennial-dominated communities (Stevens et al. 1985). Though its competitive ability against invasive annuals on disturbed sites is reported (see Monaco et al. 2003; Harrison et al. 2000), little is known about its competitive interaction with native perennial species (Harrison et al. 2000). It has been suggested that *K. prostrata* will not spread into perennial plant communities (Harrison et al. 2000; Monaco et al. 2003; Waldron et al. 2010a), and that native species may re-establish in areas dominated by *K. prostrata* over time (Harrison et al. 2000; Waldron et al. 2001), however these suggestions are based on anecdotal observations. *Kochia prostrata* has been observed to dominate droughty sites, where it might inhibit native species recruitment

(Harrison 2004). Some scientists and managers have recommended that *K. prostrata* seedings be limited to greenstrips and fuels management projects and that it not be seeded in large scale rehabilitation efforts until further research is completed on its interaction with native vegetation (Participants of the Snake River Birds of Prey NCA habitat restoration workshop 1999).

Other concerns are associated with the wildlife habitat and forage value provided by *K. prostrata.* Its structure as a sub-shrub differs greatly from that of the shrubs in historic sagebrush communities, and it may not provide the structural support needed for shrub nesting birds (J. Klott, ID BLM, personal communication, 2 November 2010). Likewise, its short stature limits its provision of wildlife habitat in areas with high snowfall (ZoBell et al. 2003). It has been recommended that *K. prostrata* not be seeded in treatments intended to enhance habitat for the endangered pygmy rabbit (*Brachylagus idahoensis*), as *K. prostrata* does not provide the structural characteristics needed for this species or for other sagebrush-obligate wildlife (Lee 2008). Though it has been suggested that mule deer, elk, and antelope may feed on *K. prostrata* (Gray 2004; Wood 2006), no peer-reviewed studies have investigated its forage or habitat value for wildlife in the Intermountain West (T. Black, Utah State University, personal communication, 1 November 2010; J. Klott, ID BLM, personal communication, 2 November 2010).

Kochia prostrata has been described as invasive in slickspot and playa communities in southern Idaho. Slickspots are small, highly saline areas that retain moisture late into the season due to their clay subsurface composition (White and Robertson 2009) and are generally devoid of vegetation. It has been suggested *K. prostrata* can move into slickspots, where it might displace the endemic *Lepidium papilliferum* (L.F. Hend.) A. Nelson & J.F. Macbr. (slickspot peppergrass) and *Lepidium davisii* Rollins (Davis' peppergrass) (Tuason 2005; Federal Register 2009; USDI FWS 2010), both previously listed as Idaho sensitive species (Pellant et al. 2004); *L. papilliferum* was listed in 2009 as threatened under the Endangered Species Act (Federal Register 2009). Currently, the Idaho BLM Twin Falls District suggests that use of *K. prostrata* be "restricted" in *L. papilliferum* habitat (USDI BLM 2010), and the Idaho BLM Jarbidge and Four Rivers Field Offices suggest that *K. prostrata* be seeded in areas adjacent to slickspot habitat as a last resort, and only if its benefits are demonstrated to outweigh potential risks to *L. papilliferum* (USDI BLM 2009c).

Several studies have reported spread of *K. prostrata* from seeded areas (e.g. McArthur et al. 1990; Harrison et al. 2000), though most unpublished and published reports suggest that claims that it might be invasive are "largely unfounded" (Harrison et al. 2000; Waldron et al. 2001; Harrison 2004). In one case, *K. prostrata* was found on both seeded and adjacent unseeded sites, and over 5 yr, cover of *K. prostrata*, *B. tectorum*, and *Pseudoregnaria spicata* (Pursh) A. Löve (bluebunch wheatgrass) in the seeded site increased while that of *Phlox hoodii* Richardson (spiny phlox), a native perennial, declined (Eldridge and Wicklow-Howard 2004). In another case, *K. prostrata* spread more than 400 m over 18 yr, from a seeded area across an abandoned *B. tectorum*-invaded field and into the fringe of a sagebrush community (McArthur et al. 1990). It has been suggested that spread of *K. prostrata* from seeded areas may be particularly likely when adjacent areas are droughty, highly disturbed and low in productivity, conditions that often occur on low elevation sites with saline soils (Harrison et al. 2000).

Introduced species pose serious threats to biodiversity and are a significant driver of global change worldwide (Vitousek et al. 1997; Dukes and Mooney 1999). Many accidental or well-intended introductions of exotic species have resulted in invasion of new areas after a lag period (Mack et al. 2000; Brooks and Pyke 2001). *Kochia prostrata* is a relatively new species in the Great Basin and Columbia Plateau, though its use has been widespread (Harrison 2004). Given that the Great Basin is considered one of the most endangered biomes in the United States, a comprehensive, quantitative study of *K. prostrata*'s invasion potential is necessary to increase understanding of its recruitment patterns throughout degraded lands that are targeted for rehabilitation (e.g. Participants of the Snake River Birds of Prey NCA habitat restoration workshop 1999; Harrison et al. 2000; Monaco et al. 2003). Currently, no peer-reviewed studies have addressed the long-term ecological implications of use of *K. prostrata* on western rangelands.

I conducted a systematic investigation of *K. prostrata*'s invasion potential by sampling post-fire rehabilitation and greenstrip seedings 3 – 24 yr after seeding on BLM land throughout the western Snake River Plain and surrounding uplands, Idaho. I collected data on cover of *K. prostrata* and associated plant species within seeded and adjacent unseeded areas. In sites where *K. prostrata* had spread from the seeded area, I measured distances of spread. These data allowed me to describe quantitatively *K. prostrata*'s spread and to describe and

interpret its spatial patterns in associated plant communities. I created models that represented environmental and site history factors most strongly associated with the spread of *K. prostrata* from seeded sites in the study area; these model should aid land managers' planning future rehabilitation efforts. I asked the following questions:

1. What similarities or differences in plant communities characterize paired seeded and unseeded areas?
2. Is abundance of *K. prostrata* correlated with environmental variables or abundance of individual plant species or functional groups?
3. Does *K. prostrata* spread from seeded sites, and, if so, to what extent?
4. Does the extent of spread differ among plant communities?
5. What site history or environmental attributes are most strongly associated with extent of its spread?

This study provides insight into *K. prostrata*'s invasion potential within a subset of the range of site conditions under which it has been seeded. Results contribute to the current understanding of benefits and risks associated with the use of *K. prostrata* in rehabilitation efforts throughout the western rangelands.

METHODS

Study Area

Field research was conducted in the summer of 2010 (mid-June through late July) on 28 USDI BLM post-fire rehabilitation and greenstrip seedings located within the western Snake River Plain and surrounding uplands in southwestern Idaho (approximately 43º53´ to 42º34´N and 116º49´ to 115º2´W; Fig. 1). Low elevation sites were located primarily within the flat plains of the Snake River Birds of Prey National Conservation Area (NCA), while higher elevation sites were on the dissected uplands south of the Snake River, between the towns of Mountain Home and Twin Falls; elevation ranged from 770 to 1242 m. Climate in southwestern Idaho is semi-arid with hot, dry summers and long, cold winters. Mean annual precipitation ranges from 18-38 cm (PRISM 2006), most of which falls as winter snow and early spring rain (Paige and Ritter 1999). At a centrally located weather station in Mountain Home, mean annual precipitation for the sampling year (2010) was 26.8 cm, which was

slightly higher than the historical mean of 25.3 cm (Western Regional Climate Center 2008).
The long-term mean summer maximum temperature was 46 ℃ while the mean winter
minimum was 6 ℃ (Western Regional Climate Center 2008).   Strong winds are common and
often originate from the Pacific Ocean or from the eastern end of the Snake River Plain. An
active geologic history combined with millions of years of Snake River drainage has resulted
in soils which are largely well-drained silty to stony loams; parent material consists of silty
alluvium and volcanic ash over basalt-derived bedrock (USDA NRCS 2008).

28 study sites located throughout the western Snake River Plain and surrounding uplands, Idaho, USA
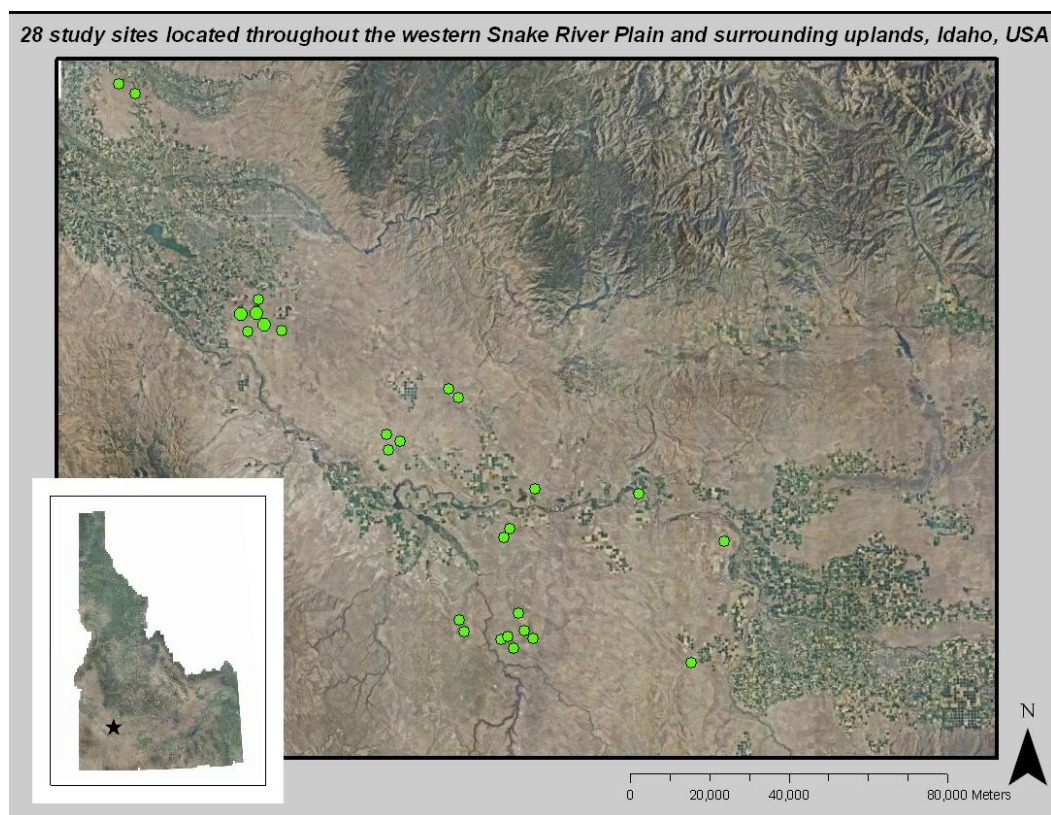
Figure 1. Study area in the Snake River Plain of southwestern Idaho, USA. Study sites are
represented by green circles, all of which are located on public land managed by the USDI
Bureau of Land Management.

Vegetation dynamics throughout the Snake River Plain are influenced by soils,
climate, topography, disturbance history, and past management. Plant species must be
adapted to a harsh, short growing season and some can tolerate alkaline to saline soils.
Changes in disturbance regimes have greatly influenced community composition. For

example, intensive cattle grazing during the early to mid 1900's caused the selective removal of understory bunchgrasses and forbs, and the trampling of biological soil crusts (Fleischner 1994). This, coupled with frequent and intense wildfire fueled by invasion by *B. tectorum*, has greatly diminished cover of sagebrush steppe communities, opening the landscape to further invasion by exotic species (D'Antonio and Vitousek 1992; Brooks and Pyke 2001). Post-fire rehabilitation and restoration efforts performed by the BLM and other land management agencies have focused primarily on attempting to control invasion by exotic species and decrease potential fuels (Pellant 1994; Eiswerth et al. 2009).

Plant communities varied greatly among study sites and between seeded and adjacent unseeded areas but were primarily composed of sagebrush steppe and salt desert shrub community types (Miller and Eddleman 2001). *Bromus tectorum* and *Poa secunda* J. Presl (Sandberg's bluegrass) were typically the dominant plant species. Commonly encountered exotic species included *Ceratocephala testiculata* [Crantz] Roth (burr buttercup); *Sisymbrium altissimum* L. (tall tumblemustard); *Salsola kali* L. (Russian thistle); and *Lepidium perfoliatum* L. (clasping pepperweed). *Artemisia tridentata* Nutt. spp. *wyomingensis* Beetle & Young (Wyoming big sagebrush; hereafter *A. tridentata*) and *Atriplex confertifolia* (Torr. & Frém) S. Watson (shadscale), both native shrub species, were often present in unseeded areas. Seeded species commonly present included *K. prostrata*, *Pseudoregnaria spicata*, and *Agropyron cristatum*.

Field Methods

Study sites differed with respect to elevation, topography, and precipitation and were selected to represent the range of environmental characteristics over which *K. prostrata* has been seeded throughout southwestern Idaho (Appendix A, Table 1). Sites areas were selected using the United States Geological Survey (USGS) Land Treatment Digital Library (Unpublished data, U.S. Geological Survey Land Treatment Digital Library, provided on 8 October 2010) and based on communication with Idaho BLM and USGS employees. Sites were defined as post-fire rehabilitation and greenstrip seedings that contained *K. prostrata* in the seeding mix, were seeded by the BLM between 1986 and 2007 (Appendix A, Table 2), and had known seeding boundaries determined largely from shapefiles of seeding treatments. Restricting sites to ground seedings and greenstrips enabled me to verify the seeding boundary

in the field by noting remnant drill rows or obvious changes in species composition. One site had been aerially seeded and one had been both aerially and drill seeded with *K. prostrata,* however seeding boundaries were verified by agency personnel for these sites. If no shapefile was available, sampling locations were targeted by determining the seeding boundary visually, based upon drill rows or natural barriers such as roads and fences. Idaho 2009 1-m NAIP imagery (USDA-FSA-APFO 2009) was also used to supplement field determination of seeding boundaries when no shapefile was available. Seeding boundaries were often associated with disturbance such as herbicide application, roadsides, or seeding efforts, including fire lines for post-fire rehabilitation sites. If sites included obviously different plant community types, topographical features, or soil characteristics in the field, they were split and treated as two study sites.

Three plots were sampled at each study site (Fig. 2); plots were defined as paired seeded and unseeded areas for sampling plant community composition and spread of *K. prostrata* from the seeded area. Plot locations were determined *a priori* using geographic information systems (GIS) shapefiles. Multiple random points were created in ArcMap (ESRI 2009) along the seeding boundary and these waypoints were uploaded onto a handheld global positioning system (GPS). In the field, three of these points were randomly chosen for sampling, within the following constraints: accessible adjacent seeded and unseeded areas, width of the seeded area ≥ 65 m, cover of *K. prostrata* within the seeded area $\geq 5\%$, and ≥ 200 m distance from another plot. If *a priori* plot locations did not fulfill criteria or were inaccessible, plots were positioned by randomly choosing additional locations along the seeding boundary that did fulfill them.

For each plot, a centerpoint on the seeding boundary was marked with a handheld GPS, and 15-m buffers, perpendicular to that boundary, were measured into the seeded area and into the unseeded area (Fig. 2). The buffer was established to accommodate uncertainty about the exact location of the seeding boundary, whether based on GIS shapefiles or visual determination. Within the seeded area, a 50-m transect was established perpendicular to the seeding boundary with the 0-m point on the seeded edge of the 15-m buffer. For each transect, aspect and waypoints for the 0-m and 50-m points were recorded for future reference. I used line point intercept to record number of "hits" for each species, sampling at 1-m intervals along the transect; number of hits were later converted to percent cover taking into account the

number of hits relative to the total number of points measured along the transect. Substrates (including material present under the vegetative canopy), including rock, moss, biological crust, soil, cow manure, and badger mounds, and slickspots were also recorded at 1-m intervals using line point intercept. Within the unseeded area, a similar 50-m transect was established running 180º from the first, with the 0-m mark beginning on the edge of the 15-m buffer into the unseeded area. Sampling along this transect was as previously described.
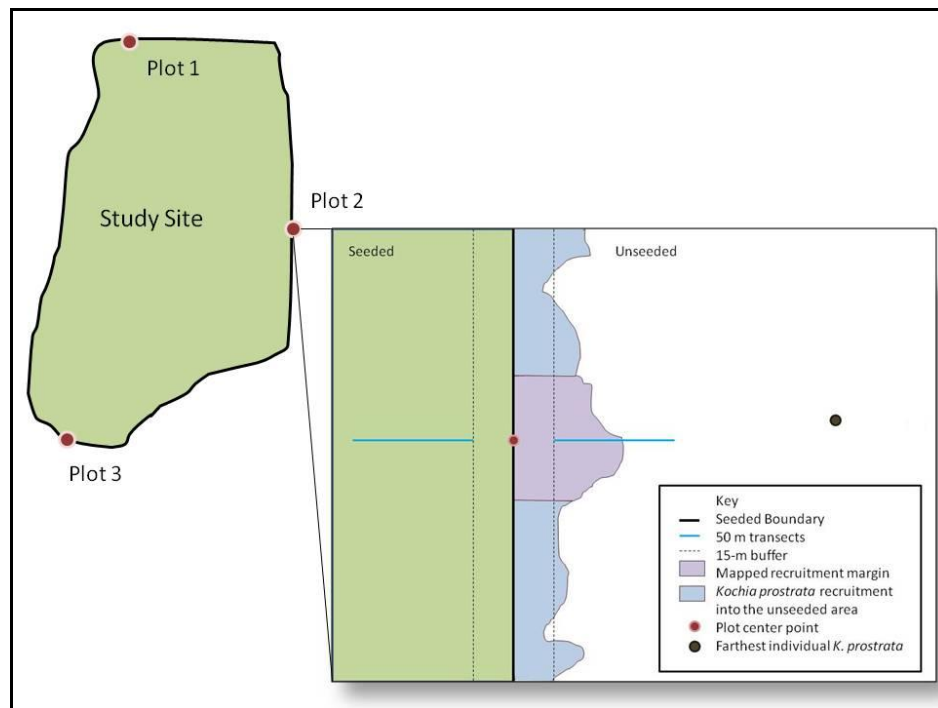
Figure 2. Schematic sampling design for study sites located throughout the western Snake River Plain, Idaho. This schematic illustrates a site on which spread of *K. prostrata* from the seeded area had occurred.

On sites where *K. prostrata* occurred beyond the seeding boundary, I mapped the "recruitment margin" along an ≈ 50 m section of the seeding boundary adjacent to each plot (Fig. 2). I defined the recruitment margin as the outer boundary of the area that supported *K. prostrata* with a density of ≥ one plant per m². From the centerpoint of each plot, I walked ≈ 25 m along the seeding boundary and recorded a waypoint using a handheld GPS. From that waypoint I enabled the GPS tracking function to map my path and walked into the unseeded

area, parallel to the transect, into the unseeded area. Once the recruitment margin was reached (*K. prostrata* density became < one plant per m^2), I walked the recruitment margin back towards the transect line and beyond it for ≈ 25 m in the opposite direction. If *K. prostrata* had not spread beyond the seeding boundary or if its density was < one plant per m², no recruitment margin was recorded for that plot.

The distance of the farthest *K. prostrata* individual from the seeding boundary was determined for each plot by walking into the unseeded area beyond the end of the 50-m transect, following the same compass bearing as the transect, and conducting an ocular survey for *K. prostrata* across the landscape. A search was conducted with a maximum distance cutoff of ≈ 800 m and, if a plant was not immediately visible, for a maximum of 15 min before stopping. Often, small populations or individuals of *K. prostrata* were present in low lying areas or along roads, so these areas were targeted in the ocular scan. Generally, the locations of three to four plants per plot vicinity were marked, using GPS; the distances for each were determined later and data for the farthest of these plants was used in analyses.

Statistical Analysis

Data preparation

I gathered site information from the LTDL and project files at Idaho BLM field offices. Soils data were downloaded from the Soil Data Mart (USDA NRCS 2008) from which I used Map Unit Descriptions (descriptions of the soil taxonomy and topographic positions; Duniway et al. 2010) in conjunction with GIS shapefiles to determine the dominant ecological site characteristics for each plot (Appendix A, Table 1; Appendix B, Table 1). Ecological sites are groupings of soil map units that have similar potential to support plant communities and respond to management and disturbance (Duniway et al. 2010). I acquired site-level values for precipitation (PRISM 2006), elevation (Idaho Geospatial Data Clearinghouse 2004), and fire history (USDI BLM 2009b) using shapefiles in ArcMap (ESRI 2009); center-point values for the three plots per site were used to obtain a mean site-level value. Fire history was defined as the maximum number of fires that occurred across one or more plots from 1937-2008, and fire history since seeding was defined as the maximum number of fires that occurred across one or more plots from the year after seeding to 2008 (USDI BLM 2009b).

Information on actual livestock use (based on reports submitted by permittees, indicating number and kind of livestock, and dates of use) and seasonality were obtained from Idaho BLM field offices. I identified, from shapefiles, the grazing allotment boundaries that encompassed each site and calculated the total area (ha) of each allotment (USDI BLM 2009a; Appendix A, Table 3). Grazing is generally restricted for 2 yr after post-fire rehabilitation treatments or until objectives have been met (USDI BLM 2007), so I calculated total annual animal unit months (AUM = amount of forage necessary to sustain one cow or equivalent for a period of one month) per allotment from 2 yr post-treatment to the time of sampling (summer, 2010). I then calculated the mean total annual AUMs for each site for that time period, and divided that by the area of the allotment (ha) to enable comparisons across sites. In cases where AUM data were missing for a particular year, means were calculated across available data. The coefficient of variation (standard deviation/mean; CV) of total annual AUMs was used to indicate variability in yearly grazing intensity for each site. Where actual use data were not available, I used billed use (AUMs the permittee was billed for) to determine approximate AUMs; actual and billed use are positively correlated (D. Pyke, USGS, personal communication, 6 October 2010). When grazing allotments covered significantly more area than I sampled, and grazing records specified detail to the pasture level (smaller areas of an allotment), grazing intensity metrics were calculated at the pasture level.

I calculated two metrics of *K. prostrata* spread, as follows. Using the mapped *K. prostrata* recruitment margin in ArcMap (ESRI 2009), I calculated the area (m²) of *K. prostrata* spread and divided it by the actual distance I had walked along the seeding boundary (≈ 50 m, measured in ArcMAP) for each plot to determine the mean distance (m) of the recruitment margin from the seeding boundary.   The three plot-level distances of the recruitment margin were averaged to yield one site-level mean distance. Distance of the farthest *K. prostrata* individual (GPS point) from the closest point on the seeding boundary for each plot was measured using ArcMap. The maximum distance of farthest individual across three plots was used to represent the distance of the farthest individual for the site.

Percent cover for each species and surface substrate was calculated for each transect; these were averaged to yield site level values. A species matrix (56 sample units X 43 species) was created, which contained mean cover values for each species in paired seeded and unseeded areas. Species data were analyzed in relation to an environmental matrix (56 sample

units X 28 variables), which included site history information, environmental variables, and cover of substrates (Appendix B, Table 1). Species were sorted into functional groups by provenance (native or exotic) and life history (annual or perennial; forb, grass, or shrub) characteristics using information from the PLANTS database (USDA NRCS 2011; Appendix C, Table 1), and total cover and proportions for each group were calculated for each site. A functional group matrix (56 sample units X 11 functional groups) was created which included total cover for each functional group category.

Data transformations and all multivariate analyses were implemented using PC-ORD version 6.0 (McCune and Mefford 2011). Species that occurred in fewer than 5 % of sample units (n =21) were removed from the species matrix, resulting in a matrix of 56 sample units by 22 species. Species and functional group data were square root transformed to reduce emphasis on the few dominant species or groups. In the environmental matrix, *K. prostrata* spread distances were $\log10(x + 1)$ transformed, and precipitation values (mm) were $\log10$ transformed to reduce skewness. Surface substrate cover values were square-root transformed to make them comparable to data in the species matrix. Two sample units were identified as outliers (variance >2 SD from the mean distance between all sample units); both were low elevation sites with low diversity, but with high cover for dominant species. Because removal of these outliers did not substantially affect results, and resulted in identification of additional outliers, they were retained in analyses.

Community differences: Seeded versus unseeded areas

I tested the null hypothesis of no difference in plant community composition between paired seeded and unseeded areas, using permutation-based multivariate analysis of variance (PerMANOVA; Anderson 2001). PerMANOVA is a non-parametric procedure for testing group differences much like parametric ANOVA, however it does not assume normality or linearity in the data, sums of squares are based on mean distances among sample units within groups rather than on means of a single response (Peck 2010), and statistical significance is assessed on the basis of a pseudo-F ratio and permutation tests (McCune and Grace 2002). PerMANOVA also allows for tests to be based on Sørensen distance (Bray and Curtis 1957), a proportionate city-block distance measure that I used in further analyses of these plant communities. PerMANOVA was run on the transformed species matrix, omitting *K.*

prostrata, as its inclusion would likely have produced differences between seeded and unseeded areas, given that it was planted in seeded areas. Group membership was indicated in the environmental matrix, using "site" as the blocking variable, and "seeded" and "unseeded" as the pairing variable.

I explored the contribution of individual species to differences between paired seeded and unseeded areas using Blocked Indicator Species Analysis (Root et al. 2010) implemented in PC-ORD version 6.0 (McCune and Mefford 2011). Indicator species analysis combines relative abundance and relative frequency of a species in defined groups, and produces indicator values (IVs), which are the percentage of perfect indication for a species within a particular group (McCune and Grace 2002). Statistical significance of indicator values (*p*-value) is evaluated using a Monte Carlo method of randomizations; 1000 randomizations were run to determine the proportion of random trials that gave an indicator value equal to or greater than the observed. I calculated differences between seeded and unseeded areas in species richness, Shannon-Wiener diversity, proportions of cover by *K. prostrata, B. tectorum*, native and exotic species, and used the Wilcoxon Singed-Rank test to test for statistical significance (at $p \leq 0.05$) of these differences, using R version 2.10.0 (R Development Core Team 2009).

Patterns of *K. prostrata* abundance in relation to the associated plant community and environmental variables

I identified major patterns in community composition and *K. prostrata* abundance, and environmental factors associated with these patterns, using non-metric multidimensional scaling (NMS) ordinations (Kruskal 1964). NMS is an ordination method that is well-suited for community analyses, in which data are often non-normal and involve non-linear relationships (McCune and Grace 2002).   I ordinated sample units on the basis of both the transformed species matrix and the transformed functional group matrix, using the autopilot setting "slow and thorough" mode (250 runs with real and randomized data, random starting configuration, no penalty for ties) and the Sørensen distance measure. I then assessed relationships (Pearson's correlation) of individual species cover, functional group cover, and environmental variables to the primary gradients in plant communities (ordination axes).

Patterns of *K. prostrata* spread from seeded sites

To determine if *K. prostrata* had spread from seeded sites, I used distances to the farthest *K. prostrata* individual and mean distances to the recruitment margin to estimate spread at the individual plant and stand levels, respectively. I graphed the mean distances from the seeding boundary to the farthest *K. prostrata* individual (m) and to the recruitment margin (m) relative to time since seeding (yr) to determine whether distances differed by time since seeding. The rates of spread were estimated as the slope of linear trendlines (distance/yr). To determine whether *K. prostrata* spread differed between communities that currently support or do not support sagebrush-steppe vegetation, I graphed spread metrics in unseeded sites with and without *A. tridentata* relative to time since seeding (yr). Though sites that supported *A. tridentata* were not intact remnant *A. tridentata* communities (many had *B. tectorum* and an abundance of other exotic species), they were the closest representation of historic communities within sampled sites. To further explore the dynamics of spread in relation to plant communities and time since seeding, I graphed native species richness for each unseeded site relative to time since seeding (yr), and superimposed distances to the farthest *K. prostrata* individual for each. To determine whether *K. prostrata* spread differed over the range of elevations sampled, I graphed spread metrics in unseeded sites relative to elevation (m).

Modeling the spread of *K. prostrata* in relation to environment

To assess whether environmental variables could be used to predict extent of *K. prostrata* spread, I used non-parametric multiplicative regression (NPMR; McCune 2006) in HyperNiche v. 2.10 (McCune and Mefford 2009). NPMR accommodates the fact that species respond to multiple interacting environmental factors by incorporating interactions among predictors and it also assumes nothing about the shape of the response surface (McCune 2006). NPMR uses an iterative process to create multiple potential models of many combinations of predictors and selects those of best fit, using cross-validation to avoid overfitting.

The response variables (in separate models) were mean distance to recruitment margin (m) and distance of the farthest *K. prostrata* individual (m), both square-root transformed, and predictors were drawn from the transformed environmental matrix described previously. To

select candidate predictors, I examined relationships between environmental variables and the measures of *K prostrata* spread using scatterplots, selecting those with seemingly non-random relationships (Appendix B, Table 1); candidate predictors included time since seeding (yr), elevation (m), mean annual precipitation (mm), number of fires (1937-2008), years of grazing since seeding, mean AUMs/ha, variability in annual grazing intensity (CV), and cover values (%) for litter, moss, rock, biological crust, soil and badger mounds. Models were created using the local mean setting, which estimates the response as an average of observed values, with an automatic minimum neighborhood size (N*). N* indicates the amount of data used to obtain a point estimate on the response surface. In NPMR, the estimate for a point on the response surface is weighted depending on its proximity to a data point, with these weights diminishing according to a Gaussian smoothing parameter. Best fit models were selected based on the leave-one-out cross-validated statistic for fit (xR^2) which is similar to the conventional R^2, however point data are excluded when calculating the mean, resulting in the possibility of a negative xR^2 for a weak model (McCune 2006). Predictors were evaluated using sensitivity analysis, in which sensitivity of a predictor is determined by the magnitude of change in response of the model caused by altering that predictor; predictors with higher sensitivity have more influence on model response. Statistical significance of the whole model is evaluated using a randomization test to determine if the fit of the selected model is better than that expected by chance (McCune 2006).

RESULTS

Community differences: Seeded versus unseeded areas

Across 28 sites and 168 transects (84 seeded and 84 unseeded), 43 plant species were recorded (Appendix C, Table 1). Twenty two species were native and 17 were exotic, with 4 unknown species. Across seeded and unseeded areas, site level mean total plant cover was 110.2 % (cover can exceed 100% when point-intercept sample pins encounter multiple layers of vegetation), with mean species richness (S) of 7.1 and, mean Shannon-Wiener (H') diversity of 1.3. Mean species richness and Shannon-Wiener diversity were greater in unseeded than in seeded areas (S = 7.9 and 6.2, respectively; Wilcoxon Signed-Rank Test, Z = 2.72, *p* = 0.006; H' = 1.39 and 1.14, respectively; Z = 3.76, *p* < 0.001). Total plant cover was

higher in seeded than in unseeded areas (116.8 and 103.6 %, respectively; $Z = 2.51$, $p = 0.01$); mean plant canopy cover was 81.5 % across seeded and unseeded areas, and tended to be slightly higher in unseeded than in seeded areas (83.9 and 79.1 %, respectively), but this difference was not statistically significant ($p = 0.06$).

Seeded and unseeded sites differed in proportions of cover by various functional groups and in overall community composition. Total proportions of cover by exotic species (omitting *K. prostrata*, as it was seeded) within unseeded areas tended to be higher than in seeded areas (means = 0.54 and 0.48, respectively), but this difference was not statistically significant ($p = 0.07$). Seeded and unseeded sites did not differ significantly in proportions of *B. tectorum* cover (means = 0.22 and 0.26, respectively; $p = 0.09$). Total proportions of *K. prostrata* cover ranged from 0.1 to 0.72 in seeded areas and 0 to 0.68 in unseeded areas; mean proportions of *K. prostrata* cover were greater within seeded than in unseeded areas (means = 0.27 and 0.09, respectively; $Z = -4.62$, $p < 0.001$). Native species comprised higher proportions of cover within unseeded than within seeded areas (means = 0.37 and 0.25, respectively; $Z = 2.49$, $p = 0.01$). Paired seeded and unseeded areas did differ in overall community composition (excluding *K. prostrata*; PerMANOVA; $p < 0.001$; Table 1).

Table 1. Permutation based multivariate analysis of variance (PerMANOVA) tests of the null hypothesis of no difference in plant communities between paired seeded and unseeded areas, omitting *K. prostrata*. "Block" = site; "Seed" = seeded or unseeded sample units.

Source	d.f.	SS	MS	F	*p*
Block	27	9.18	0.340	4.95	0.0002
Seed	1	0.53	0.534	7.78	0.0002
Residual	27	1.85	0.069		
Total	55	11.56			

Indicator species analysis provided information on species that were primarily abundant in and faithful to either seeded or unseeded areas. As expected, *K. prostrata* was an indicator for seeded sites (IV =78.1, $p = 0.001$) along with the introduced species *Agropyron cristatum* (IV = 46.1, $p = 0.002$). Indicator species for unseeded areas were the exotic species *Sisymbrium altissimum* (IV = 52, $p = 0.05$) and *Salsola kali* (IV = 35.1, $p = 0.039$), and the native shrub, *A. tridentata* (IV = 37.2, $p = 0.002$).

Patterns of *K. prostrata* abundance in relation to the associated plant community and environmental variables

An NMS ordination of sample units in functional group space resulted in a 3-dimensional stable solution (final stress = 11.1, final instability = 0.0000, Fig. 3). A randomization test confirmed that the final stress was lower than expected by chance (p = 0.004). Cumulatively, the three axes explained 95 % of the variance in functional group cover, with Axis 1 explaining the majority (73%), and Axes 2 and 3 sharing the remainder (10% and 12%, respectively). Seeded and unseeded sample units tended to be intermixed in the ordination space. *Kochia prostrata* was the only exotic perennial shrub, and it had a strong and positive correlation with Axis 1 (r = 0.75), as did cover of exotic annual forbs (r = 0.77). Exotic perennial grasses (r = -0.70), exotic annual grasses (*B. tectorum*; r = -0.73), native perennial grasses (r = -0.73), and native perennial shrubs (r = -0.41) all had strong negative relationships with Axis 1. Axis 1 was also negatively correlated with elevation and cover by fine litter and positively correlated with yearly variation in grazing intensity and biological crust cover.

Axis 2 was positively correlated with cover by native perennial grasses (r = 0.51) and native annual forbs (r = 0.45); while exotic perennial shrubs (r = -0.44) and exotic annual grasses (r = -0.43) were negatively correlated with this axis. Cover by moss was the only environmental variable strongly correlated with this axis (|r| = 0.54; all others < 0.4). Axis 3 was positively correlated with cover of exotic annual grasses (r = 0.47), however it was not strongly correlated with any of the environmental variables (maximum r = 0.36).
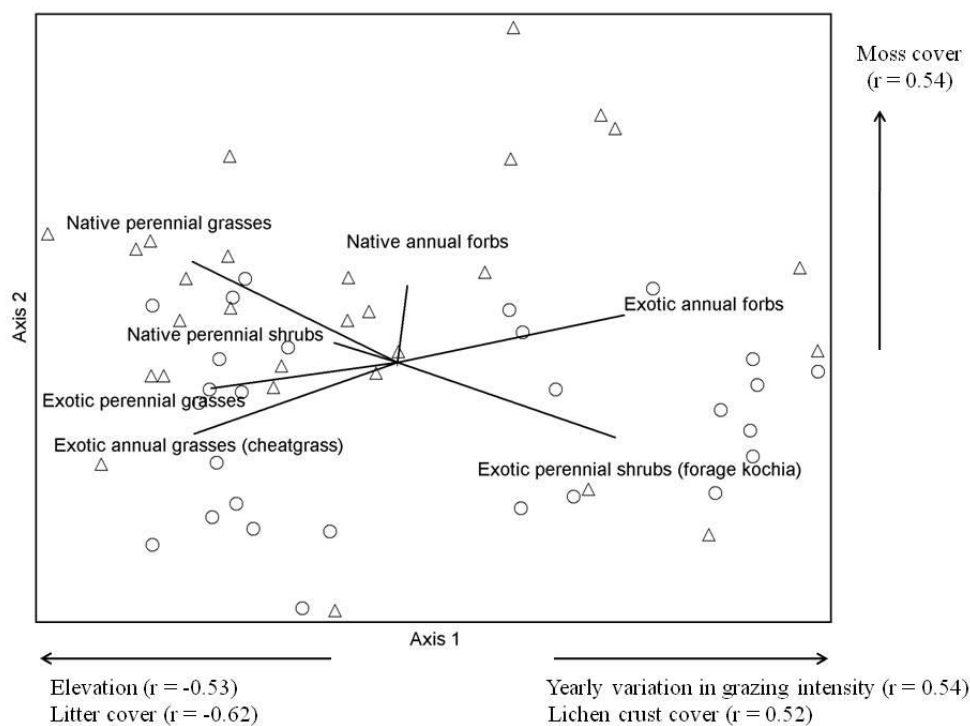
Figure 3. NMS ordination of sample units in functional group space, with overlays of functional groups. Points represent sample units, and distance between points indicates their similarity in functional group composition. Circles represent seeded sample units and triangles represent unseeded sample units. Functional groups with notable correlations ($|r| >$ 0.40) with ordination axes are indicated by vector lines on the ordination diagram, with the length of the line representing the strength of the correlation; environmental variables correlated ($|r| > 0.40$) with ordination axes are indicated along the margins.

The NMS ordination of sample units in species space (Appendix B, Fig. 1 & Table 2) indicated similar relationships to those revealed by the functional group ordination, with intermixing of seeded and unseeded sample units and a primary axis strongly correlated with *K. prostrata* cover (r = 0.83). This ordination, however, identified additional environmental correlates; years since seeding (r = 0.49), years of grazing since seeding (r = 0.49), number of fires since seeding (r = 0.42), and silty soils (r = 0.45) were positively correlated with the primary axis, while mean grazing intensity (r = -0.50) and mean annual precipitation (-0.41) were negatively correlated with this axis. Species negatively correlated with this axis included the exotic *B. tectorum* (r = -0.74), seeded *Agropyron cristatum* (r = -0.62), and natives *Poa secunda* (r = -0.70), and *A. tridentata* (r = -0.43). Species positively correlated with this axis

included the exotic forbs *Ceratocephala testiculata* (r = 0.85) and *Lepidium perfoliatum* (r = 0.41).

Patterns of *K. prostrata* spread from seeded sites

Kochia prostrata spread from 89 % of sampled seeded sites. Distances of the farthest *K. prostrata* individual from the seeding boundary ranged from 0 - 710 m with a mean distance of 208 m (Fig. 4A). Distance increased with time since seeding ($r^2 = 0.47$), with an estimated mean rate of spread of 25 m/yr. Distances to the recruitment margin ranged from 0 - 197 m, with a mean distance of 30 m (Fig. 4B). The distance to the recruitment margin also increased with time since seeding ($r^2 = 0.50$) with an estimated mean rate of expansion of the initial seeding of 4 m/yr.   Seventy five percent of sampled sites had a measureable recruitment margin. *Kochia prostrata* spread into unseeded areas with and without *A. tridentata* (Fig. 4). In the 12 unseeded areas in which *A. tridentata* was present, distance to the recruitment margin ranged from 0 - 80 m, with an average of 11 m. Distance of the farthest *K. prostrata* individual in these sites ranged from 0-357 m with an average of 110 m. Due to differences in ranges of years since seeding between sites with *A. tridentata* and those without (10 to 18 yr and 3 to 24 yr, respectively), direct comparisons of spread between the two types of communities cannot be made; no differences within the 10 to 18 yr range were apparent. No clear patterns of spread in relation to unseeded areas were apparent. Native species richness in unseeded areas ranged from 0 to 9 species, and distances to the farthest *K. prostrata* individual varied widely across these sites (Fig. 5); greatest distances of spread occurred in sites that were seeded longest ago, which also had relatively low native species richness. Both metrics of spread were greatest at elevations < 1000 m (Fig. 6).
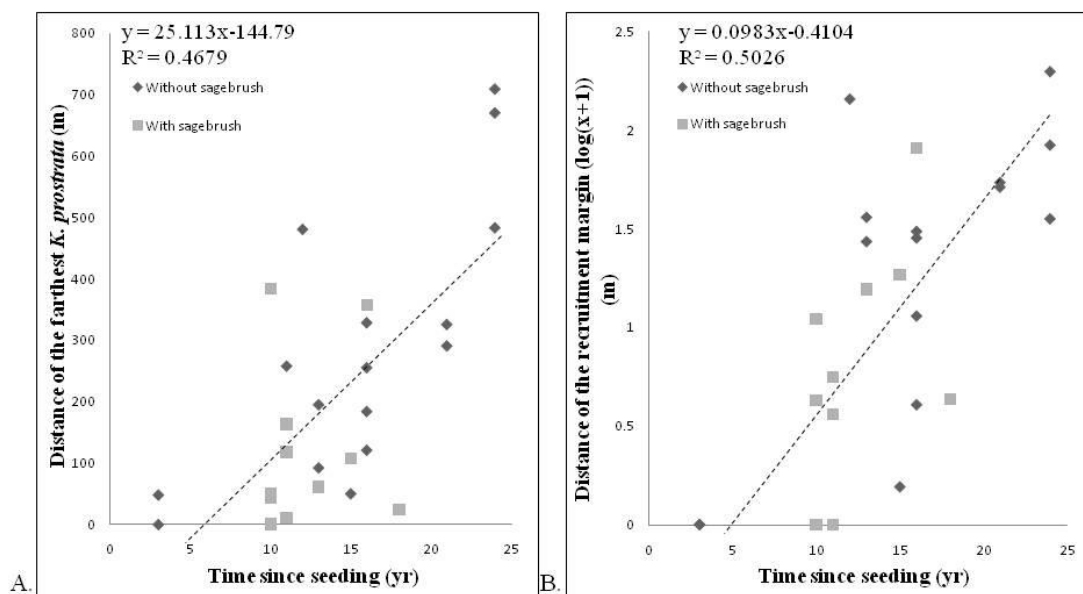
Figure 4. Distance of the farthest *K. prostrata* plant (m) from the seeding boundary (A.) and mean distance of the recruitment margin (m) (log x+1 scaled) from the seeding boundary (B.), relative to time since seeding. Points represent unseeded sample units with and without *A. tridentata*.
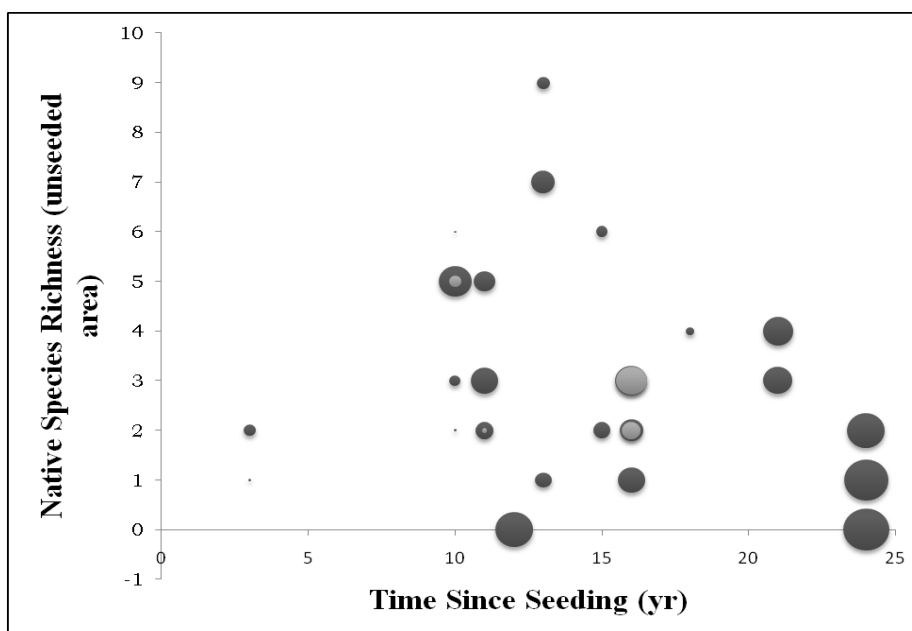
Figure 5. Native species richness in unseeded areas relative to time since seeding of adjacent seeded areas. Points represent unseeded sample units and the size represents distance of the farthest *K. prostrata* individual (m) for each sample unit. Some points represent two sample units with differing spread differences (noted by light and dark gray points overlapping).

A.   B.
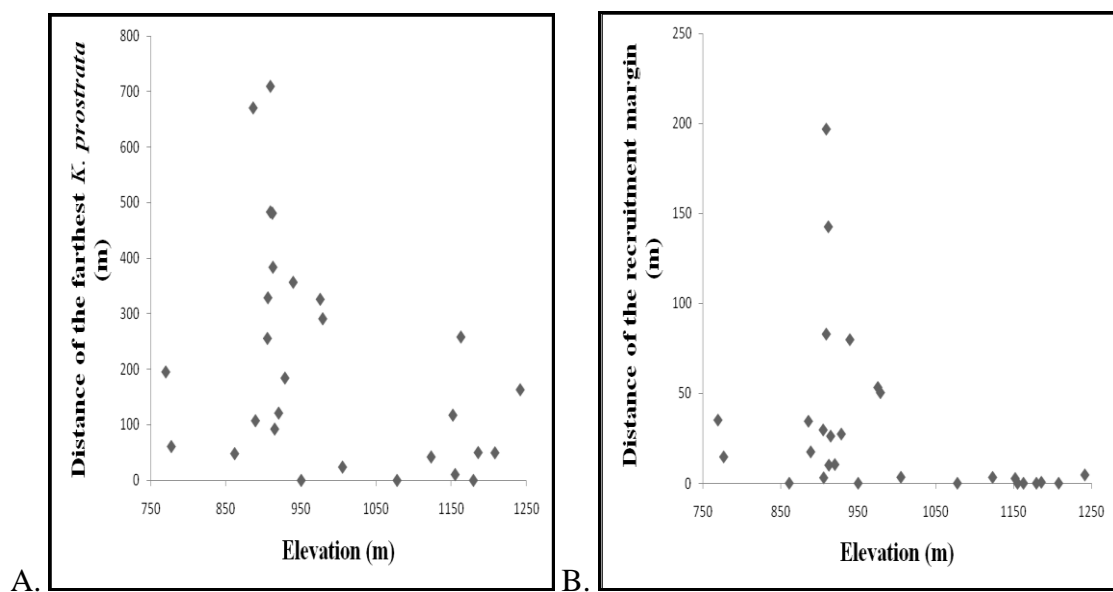
Figure 6. Distance of the farthest *K. prostrata* plant (m) from the seeding boundary (A.), and distance of the recruitment margin (m) from the seeding boundary (B.), in relation to elevation (m). Points represent unseeded sites.

Modeling the spread of *K. prostrata* in relation to environment

NPMR models for spread of *K. prostrata* (mean distances of recruitment margin and to farthest *K. prostrata* individuals from the seeding boundary) had fits that were better than expected by chance (Table 2; Appendix B, Fig. 2), and both explained a large percentage of the variation in spread distances. The strongest model for spread in terms of distance to the recruitment margin had three predictors, including variability in yearly grazing intensities, cover of soil, and elevation. The strongest model for distance of the farthest *K. prostrata* individual included predictors of years since seeding, rock cover, and badger mound cover.

Table 2. Non-parametric multiplicative regression models of *K. prostrata* spread (mean distances of recruitment margin and of farthest *K. prostrata* individuals from the seeding boundary) in relation to environmental predictors. xR^2 =leave-one-out cross-validated statistic for fit. Average N* = amount of data used to obtain a particular point estimate. Sensitivity = relative strength of each predictor, indicating the magnitude of change in model response from changing a predictor value; high values indicate greater magnitude of response. Tolerance = the range of predictor space over which data values are used to estimate response (sensitive to scale of data); predictors with narrow tolerances have greater effects on the model than do those with broad tolerances.

Measure of Spread	Recruitment margin (m)	Distance to farthest *K. prostrata* (m)
xR^2	0.81	0.64
p	0.01	0.02
Average N*	1.67	2.24
Sensitivity (Tolerance)		
Badger mound cover (%)	-	0.05 (0.32)
CV of yearly grazing	1.07 (2.77)	-
Elevation, m	0.04 (169)	-
Rock cover (%)	-	0.12 (0.83)
Soil cover (%)	0.32 (0.57)	-
Years since seeding	-	0.63 (1.47)

DISCUSSION

Community differences: Seeded versus unseeded areas

Differences in plant communities between paired seeded and unseeded areas are likely a result of diverse disturbance and management histories, with management including seeding with *K. prostrata*. The lack of pre-seeding data on these communities limits inference about effects of seeding on them; results, however, may provide insights into outcomes of post-fire rehabilitation and greenstripping treatments that include seeding with *K. prostrata*. Specific objectives for Emergency Fire Rehabilitations include restoring or establishing healthy ecosystems to provide habitat for sagebrush-obligate wildlife species, while deterring establishment of invasive exotic species (Eiswerth et al. 2009). Higher Shannon-Wiener diversity, species richness, and proportions of natives (including *A. tridentata* as an indicator species) in unseeded than in seeded areas suggest that seeding efforts may not result in diverse plant communities, at least over the time scale encompassed by this study. Objectives for greenstrip seedings differ from those for post-fire rehabilitations, and include reducing fire-suppression costs and losses of plant diversity and shrub cover, properties and structures (Pellant 1994). Though greenstrips may be effective in reducing fuel continuity (Pellant 1994; Harrison et al. 2002), those seeded with *K. prostrata* may not fulfill objectives related to maintaining plant diversity within seeded areas.

Kochia prostrata was present in both seeded and unseeded areas and its abundance varied across sites. The species dominated some seeded areas, predominantly in low elevation sites (850-1000 m); those with the greatest proportions of cover by *K. prostrata* (≥ 0.5; n=5) also included large proportions of cover by exotic annual forbs, with only small proportions of native species present. Though *K. prostrata* is seeded in part to compete with invasive annual grasses (McArthur et al. 1990; Harrison et al. 2000; Monaco et al. 2003), proportions of *B. tectorum* cover did not differ significantly between seeded and unseeded areas. In one site, *K. prostrata* dominated the unseeded area (proportion of cover = 0.68), with the remainder of vegetation also composed exclusively of exotic species. Although the mean proportions of *K. prostrata* were greater in seeded than in unseeded areas, the mean proportion across the unseeded areas was substantial (1/3 of that in the seeded area), indicating potential to dominate even unseeded areas. Disturbance associated with fire history and seeding activities

may have depleted species' populations in the seeded areas, and likely affected species composition in these areas. Lack of other species could increase the potential for *K. prostrata* to dominate some seeded sites.

Patterns of *K. prostrata* abundance in relation to the associated plant community and environmental variables

Success rates of *Kochia prostrata* seedings have been variable, and can depend on seeding conditions, seed viability, and precipitation (McArthur et al. 1996; Kitchen and Monsen 2001; Stewart et al. 2001). While I sampled 28 sites with established *K. prostrata* populations, I also visited multiple sites that had been seeded with *K. prostrata,* but on which *K. prostrata* cover was less than the minimum (5 %) required for sampling, suggesting a lack of seeding success on these sites.

Kochia prostrata abundance was positively associated with silty soils, and negatively associated with elevation and precipitation. Similar environmental associations have been noted in other low elevation, salt desert shrub habitats on saline, fine-textured soils (Waldron et al. 2001; Waldron et al. 2010b). *Kochia prostrata* may be able to persist on low elevation, low precipitation sites on which other species may be moisture-limited (Monaco et al. 2003; Newhall et al. 2004), due to its drought tolerance and ability to utilize soil moisture early in the season (Romo and Haferkamp 1987; Leonard et al. 2008). Though many species in western rangelands utilize C_3 metabolism, *K. prostrata* is a C_4 species, which enhances its drought tolerance and water use efficiency at high temperatures (Pyankov et al. 2001; Leonard et al. 2008). As a halophyte, *K. prostrata* is salt-tolerant during multiple stages of its life cycle (Romo and Haferkamp 1987) and is able to accumulate salts, that increase its internal osmotic potential, further enhance its drought tolerance, and enable it to persist in highly saline soils (Karimi et al. 2005). *Kochia prostrata* has lower water potential than *A. tridentata*, which may confer a competitive advantage in arid climates; relatively low plant water potential confers the ability to tolerate relatively low levels of soil moisture (Romo and Haferkamp 1988; Lambers et al. 1998). Though *K. prostrata* can draw down available soil moisture earlier in the season than many other species using its deep tap root and fibrous root system (Leonard et al. 2008), it has higher stomatal conductance and transpiration rates than *A. tridentata*, that results in relatively high rates of water loss from its stems and leaves. High

gas exchange rates, despite the accompanying high water loss rates, are needed for the species to reconstruct most of its above-ground biomass annually as compared to *A. tridentata* which retains most of its leaf area (Romo and Haferkamp 1988).

For many known competitive and invasive plant species in the Intermountain West, the ability to utilize soil moisture early in the season is considered to confer an important competitive advantage (Chambers et al. 2007). *Kochia prostrata* shares this ability with other highly competitive species such as *Agropyron cristatum, B. tectorum* and *T. caput-medusae* who, by preempting resources, can adversely affect native species establishment (Eissenstat and Caldwell 1988; Melgoza and Nowak 1991; Chambers et al. 1994). Numerous studies have linked water extraction capabilities with ability to compete against native species, including *Pseudoroegneria spicata* and *A. tridentata* (Harris 1977; Eissenstat and Caldwell 1988; Melgoza and Nowak 1991; Chambers et al. 1994; Henderson and Naeth 2005). Though no studies have directly investigated the roles of water relations in *K. prostrata*'s interspecific interactions, future studies should address the possibility that its water relations could negatively affect native species growth and establishment. The relatively low abundance of *K. prostrata* at higher elevation and precipitation sites could be the result of increased competition from more prevalent native species, and lessened competitive advantage associated with its drought tolerance and water extraction capabilities (Chambers et al. 2007).

While there are examples of *K. prostrata* competing with and reducing abundance of *B. tectorum* (McArthur et al. 1990; Monaco et al. 2003), little information exists on interactions between *K. prostrata* and native species. Though I did not study competition explicitly, trends in occurrence of *K. prostrata* and various plant functional groups in ordination space suggest potential interactions. *Kochia prostrata* abundance was negatively correlated with that of most functional groups, including native perennial grasses, native perennial shrubs, exotic perennial grasses, and exotic annual grasses (*B. tectorum*), and was positively correlated only with abundance of exotic annual forbs. These patterns in relative abundance of most functional groups could suggest competitive interactions between *K. prostrata* and other species, both native and exotic. While some reports speculate that *K. prostrata*'s competitive influence affects only exotic annuals (Harrison et al. 2000; Monaco et al. 2003), my results did not indicate differences between exotic annual grasses' and native species' abundance in relation to *K. prostrata*. On the contrary, *K. prostrata*'s positive

relationship to exotic annuals forbs in functional group space may suggest a lack of competitive interactions, or that other forces are more influential than such interactions. For example, the most positive species association that I observed occurred between *K. prostrata* and *Ceratocephala testiculata*, an exotic annual forb, whose seedlings (based on early spring [March 2010] field observations at low elevation sites) were commonly present under the canopy of *K. prostrata*. This could be a case of *K. prostrata* facilitating an early-season annual, although experimental studies would be necessary to elucidate causation. Though these general patterns of species occurrence that I documented may be caused by interspecific interactions, they could also result from disturbance prior to and during seeding and proximity, or lack thereof, to native seed sources. Experimental studies of interactions between *K. prostrata* and native species would increase understanding of its effects on them.

Time was an important factor related to *K. prostrata* abundance; its abundance increased both with number of years since seeding and years of grazing since seeding. Years since seeding and years grazing since seeding were strongly positively correlated, and it is likely that the association of *K. prostrata* abundance and years of grazing since seeding may simply reflect time since seeding. Though *K. prostrata* is thought to live up to 15 yr (Balyan 1972), sites that were seeded up to 24 yr prior to my sampling supported high *K. prostrata* abundance, suggesting that it has established self-perpetuating populations. Within seeded areas and in the recruitment margin, *K. prostrata* appeared to be self-limiting; early in the season many seedlings had established around reproductive individuals, however by late summer many of the seedlings had died. A similar phenomenon has been noted in salt-desert habitats (Monaco et al. 2003). This species is likely influenced by both interspecific and intraspecific competition; those species highly influenced by intraspecific competition often have strong competitive effects (Pyke and Archer 1991).

My data suggest that grazing intensity by livestock may affect patterns in *K. prostrata* abundance. Inferences about patterns in *K. prostrata* abundance in relation to grazing based on my study must, however, be made cautiously, given that my field data are observational and that grazing data were derived from actual use data provided by the BLM. While actual use data are often used to estimate grazing pressure (e.g., Lym and Kirby 1987; Hickman 2004), they are self-reported by permittees and also do not take into account livestock forage preferences or heterogeneity in grazing depending on water availability, which could affect

species' abundances and responses to grazing pressures. *Kochia prostrata* abundance tended to be higher on sites that had relatively long term post-seeding grazing histories and variable intensities of grazing, but its abundance tended to be lower on sites that had been grazed at relatively high intensities. Pre-grazing biomass and branch density have been found to be strong predictors of *K. prostrata* biomass consumption in cattle preference studies (Waldron et al. 2010b), and year-to-year variability in grazing could allow for recovery of branch density and biomass, and hence cover, after high intensity years. *Kochia prostrata* biomass increased with grazing intensity in comparisons of grazed and ungrazed sites in Mongolia (Chen et al. 2005), suggesting that some grazing likely stimulates growth, as has been noted for other arid-land shrubs (Bilbrough and Richards 1993). On the other hand, consistently high grazing intensity could decrease branch density to the point where seed production and growth of *K. prostrata* were inhibited. Decline in *K. prostrata* abundance has been observed in studies of increasing grazing pressure in northwestern China (Zhao et al. 2007) and negative effects are thought to occur when *K. prostrata* is grazed to a height of less than 5 cm (USDA NRCS 2011). Although consistently high intensity grazing may negatively affect *K. prostrata* abundance, its abundance tended to increase with years of grazing since seeding in my study; because, however, years of grazing since seeding and time since seeding were strongly positively correlated, the relationship of abundance to years of grazing could be spurious.

Though effects of seasonality of grazing on *K. prostrata* abundance were difficult to assess because most sites had been grazed in similar seasons, its cover could be reduced if high intensity grazing occurred in early spring when the plant allocates energy to flowering and fruiting (Harrison et al. 2000; USDA NRCS 2011). Cattle selectively graze inflorescences prior to consuming leaves and vegetative stems (Davenport 2005), which could affect abundance by diminishing recruitment of new plants. Some have noted that *K. prostrata* is rarely grazed in the spring, when other species are preferred (USDA NRCS 2011). Though this may be the case in communities that support other forage species, on sites lacking perennial grasses and on which *K. prostrata* is dominant, spring grazing could potentially have large effects on its abundance and spread. Experimental investigations on effects of grazing intensities and seasonality on *K. prostrata* abundance would enhance understanding of consequences of alternative grazing management strategies for this species.

Disturbance influences *K. prostrata* abundance (McArthur et al. 1990; Waldron et al. 2001), and I found that its abundance and the number of fires since seeding were positively associated. Number of fires since seeding and time since seeding were not correlated, so the relationship of *K. prostrata* abundance to numbers of fires is not simply an artifact of more time having allowed for more fires. Increased abundance of *K. prostrata* with increasing fire could result at least partly from its reported fire tolerance, ability to re-sprout after fire (McArthur et al. 1990; Harrison et al. 2002), and release from less fire tolerant competitors. Though fire is a natural disturbance in western rangelands, fire frequencies and intensity have greatly increased in the last century (Brooks and Pyke 2001). Many native perennial species can survive fire because they have perennating buds below the soil surface, but probability of survival lessens with repeated, high intensity fires (Brooks and Pyke 2001; Pyke et al. 2010). *Artemisia tridentata* cannot re-sprout after fire (West and Young 2000), though, after low-intensity fires it can re-establish from the seedbank (Schlatterer 1973; McArthur et al. 1977; Howard 1999) and from surrounding seed sources (Bushey 1987; Howard 1999). While *K. prostrata* does not have a persistent seed bank and has been noted to burn when surrounded by fine fuels in high intensity fires, its ability to re-spout after fire confers the ability to withstand repeated fire and, potentially, to increase in abundance after consecutive fires. These attributes may enable it to dominate plant communities such as salt-desert shrub, which were not historically adapted to fire but which now burn more frequently than in the past (Chambers et al. 2009), as well as plant communities that were adapted to a regime of less frequent fire than now prevails (Brooks and Pyke 2001).

Kochia prostrata abundance was positively associated with biological crust cover and negatively associated with litter and moss cover; all might reflect disturbance within the community and interactions with other species. Presence of *B. tectorum* increases cover of fine litter (Belnap and Phillips 2001; Evans et al. 2001; Sperry et al. 2006), and this, coupled with the negative relationship I observed between abundances of *K. prostrata* and *B. tectorum* across sample units, could account for the negative association between *K. prostrata* and litter cover. The positive relationship between cover of biological crust and *K. prostrata* may reflect gradients in grazing intensity or abundance of *B. tectorum*. Biological soil crusts are sensitive to physical disturbance (Belnap and Eldridge 2001; Ponzetti and McCune 2001; Sperry et al. 2006), and I found lower abundance of *K. prostrata* on sites that were grazed

most intensely, which also had lower cover by crusts. Biological crust and *B. tectorum* cover have also been reported to be negatively related (Ponzetti et al. 2007), which could also account for the positive association between crust and *K. prostrata* cover that I observed. *Kochia prostrata* is thought to recruit on bare soil and in the interspaces between plants (Harrison et al. 2000), which might explain the negative relationship of *K. prostrata* cover with that of moss, which also occupies these regions.

Patterns of *K. prostrata* spread from seeded sites

The measured distance of the farthest *K. prostrata* individual from the seeding boundary (710 m) is likely a conservative estimate of potential spread. I limited the search distance to 800 m owing to time constraints, and it is likely that other individuals occurred past the marked individuals. Further, there were often large relatively isolated patches of *K. prostrata* beyond the mapped recruitment margin, indicating that this measurement also likely underestimated expansion of the initial seeding. In many exotic species invasions, spread can occur via satellite patches that become new seed sources for invasion (Moody and Mack 1988; Archer and Pyke 1991). This pattern was observed in unseeded areas, where recruitment of *K. prostrata* occurred not only as spread from the immediately adjacent seeded area but also from more disjunct patches, that could eventually increase in density and area. Individuals marked as the farthest *K. prostrata* often occurred in small satellite populations, sometimes far from the seeding boundary.

Distances over which *K. prostrata* spread were greater than previously reported by others in research reports and proceedings. For example, sites located throughout the Great Basin (10-30 yr after seeding) had mean and maximum distances to a single plant in unseeded areas of 28 m and 386 m, respectively (Harrison et al. 2000), whereas in my study they were 208 m and 710 m, respectively. In another study, *K. prostrata* spread > 400 m across a cheatgrass-invaded field and into an *A. tridentata* community over 17 yr (McArthur et al. 1990); this distance is more comparable to distances that I measured. Differences in reported spread could result from differences in site conditions or accuracy of determining seeding boundaries. Though many reports claim that concerns about *K. prostrata* as an invasive species are "largely unfounded," (Harrison et al. 2000) it is clear that *K. prostrata* does spread into areas where it was not seeded.

Both metrics of spread increased linearly with time, indicating continued potential for further spread. Though the 4 m/yr rate of increase in distance of the recruitment margin from the seeding boundary may seem slow compared to more rapid invasions, continued spread at this rate could substantially increase the extent of *K. prostrata* on the landscape, with greater proportional increases for small seeded areas. For example, for a hypothetical circular seeded site with area equal to the smallest seeded area (8 ha) whose radius increased by 4 m/yr, the extent of *K. prostrata* would be twice that of the original seeded area within 17 yr and within 25 yr, its extent would equal nearly 265% of the original seeded area. For a hypothetical circular area with area equal to the largest seeded area sampled (1841 ha) and from which *K. prostrata* was spreading at 4 m/yr, the overall size of the area occupied by *K. prostrata* would equal approximately 108% of the original seeded area within 25 yr. These scenarios demonstrate clearly that spread of *K. prostrata* may, over time, greatly increase the area that it occupies compared to the original size of the seeding, with greater effect on smaller seedings.

My data suggest that spread may be independent of species composition in the unseeded community, however because spread distances were affected by time since seeding, trends in relation to species composition must be interpreted cautiously. Spread occurred into unseeded sites containing *A. tridentata*, those with relatively high abundance of native perennial grasses, as well as those dominated by exotic annuals. These results are contrary to reports that suggest that *K. prostrata* will not spread into perennial plant stands (Waldron et al. 2001; Monaco et al. 2003; Waldron et al. 2010a). Many report also that *K. prostrata* will spread into disturbed sites only when competition is limited (e.g. Waldron et al. 2001), however my results suggest that spread was not restricted to exotic-species invaded, highly degraded areas that supported particularly low plant cover.

Kochia prostrata spread into unseeded areas on sites containing slickspots, that are habitat for the federally threatened *L. papilliferum*, and Idaho sensitive species *L. davisii*, both endemic to these harsh saline habitats (Moseley 2005; Federal Register 2009; White and Robertson 2009). Others have suggested that invasion of *K. prostrata* into highly saline slickspots may negatively impact abundance of both *Lepidium* species (Tuason 2005; Federal Register 2009; USDI FWS 2010). I found highest abundances of *K. prostrata* on low elevation sites with silty soil, which are also areas that slickspots and playas occur in southern Idaho; slickspots could often be located visually in unseeded areas due to their high abundance

of *K. prostrata.* Currently, the BLM recommends restricted use of *K. prostrata* when seeding around slickspot habitat to protect populations of *L. papilliferum* and *L. davisii* (USDI BLM 2010), and BLM emergency fire rehabilitation plans do not allow planting of *K. prostrata* within a 402 m buffer surrounding slickspots. My results demonstrate that *K. prostrata* can spread greater distances than 402 m under similar environmental conditions, and indicate that its potential to spread great distances from the seeding boundary with time should be taken into account when planning to use this species.

I was able to model *K. prostrata* spread as both distance to the recruitment margin and to the farthest individual plant ($xR^2 = 64\%$ and 81%, respectively). Distance to the recruitment margin was most strongly predicted by variability of yearly grazing intensity, suggesting that grazing affects not only *K. prostrata* abundance but also its spread. Soil cover was also a predictor of spread, which may reflect that bare soil is necessary for its establishment. Finally, elevation tended to be a negative predictor of *K. prostrata* spread, consistent with its pattern of lower abundance at higher elevations. The strongest predictor for distance of the farthest individual was years since seeding, corroborating that time since seeding has important influence not only on abundance but also on spread into unseeded sites. Cover by rock and badger mounds were also positive predictors of spread. Relatively high rock cover might indicate areas with less vegetation and more differentiated microsites in which *K. prostrata* could establish. American badger (*Taxidea taxus*) mounds are highly abundant throughout southwestern Idaho and create ubiquitous disturbance (Eldridge 2004), into which *K. prostrata* is thought to preferentially spread (Harrison et al. 2000).

Humans have facilitated many plant invasions (e.g., Brooks and Pyke 2001; Mack et al. 2000) and anthropogenic forces should be considered when addressing invasion potential of a purposefully introduced exotic species. Though *K. prostrata* seed is primarily dispersed by wind, I found individual plants along roadsides and vehicle tracks in unseeded areas. These may have originated from seed conveyed by vehicles and their establishment may have been enhanced by disturbance caused by vehicles; roads are known conduits for invasion of unwanted species in semi-arid landscapes (Gelbard and Belnap 2003; Pyke and Knick 2003). Use of off-road vehicles in sites seeded with *K. prostrata* and adjacent unseeded areas could increase spread in these highly disturbed landscapes. Other anthropogenic influences could also affect *K. prostrata* abundance and spread. For example, the BLM issues seed collection

permits for use in seeded sites that are primarily composed of *K. prostrata* (C. Fritz, ID BLM personal communication, 01 April 2011). The species' abundance and spread may be lowered on sites from which seed is harvested often.

It has been suggested that *K. prostrata* could be useful in assisted succession on severely degraded rangelands (Monaco et al. 2003). The concept is that seeded perennial vegetation would compete with annual grasses following fire and promote native species establishment by lessening competition from exotic species (Cox and Anderson 2004). Though use of *K. prostrata* is often justified on this basis, evidence in support of its facilitating establishment of native species is anecdotal, and no published studies have previously assessed the potential of native species to become established in *K. prostrata* seedings. I found that native vegetation occurred only sparsely in *K. prostrata* seedings that were 3-24 yr old. Native species have been included in seed mixes with *K. prostrata* in some plantings, however little information exists on the success of these seedings; native establishment has been limited when sites were seeded with *Agropyron cristatum,* another exotic species often seeded for rehabilitation efforts (Thompson et al. 2006). More research should be conducted on the ability of native species to establish naturally in *K. prostrata* seedings on a range of sites, and on the potential for native species to be seeded successfully into established *K. prostrata* stands.

Many well-intended introductions, such as those of *Agropyron cristatum*, have resulted in unwanted interactions with native species. Establishing native species in *Agropyron cristatum* monocultures has become a restoration priority and control methods for the introduced species are being explored (Hulet et al. 2010). *Kochia prostrata* is considered a highly adaptable species (Waldron et al. 2001), and though it has been seeded widely, it is still a relatively new species on the landscape. *Kochia prostrata* has demonstrated its adaptability to a variety of ecosystem types and conditions and little is known about how that adaptation may affect diversity and resilience of these systems, or of other ecosystems into which it may be introduced in the future (Thorpe et al. 2011). Much like many Eurasian species that have become problematic in the United States, *K. prostrata* behaves differently in its introduced range than it does in its native habitat, where it reportedly does not grow in pure stands or spread from seeded areas (Waldron et al. 2001). These differences in population dynamics

may indicate potential for further adaptation in its novel environment, with unknown consequences for native species.

Management implications

When selecting species for rehabilitation efforts, land managers must consider potential benefits and costs, including tradeoffs concerning species diversity and ecosystem functioning (Chambers et al. 2009). Federal agencies now emphasize the use of native species in rehabilitation efforts, and managers must justify use of exotic species, planting them only if doing so will not diminish plant diversity and the species will remain confined within the management area (USDI BLM 2007). My study demonstrated that *K. prostrata* did not remain confined to seeded areas across the majority of sampled sites, nor was its spread restricted to plant communities dominated by annuals. Further, plant diversity was lower on sites seeded with *K. prostrata* than in adjacent unseeded areas up to 24 yr post-planting. Given the highly disturbed nature of many of the sites into which *K. prostrata* has been introduced, a strong rationale is necessary to justify use of this exotic species, particularly given that peer-reviewed literature addressing it potential competitive interactions with native species is lacking. Land managers should be especially cautious under the environmental conditions that seem to be most conducive to its spread, including low elevation areas with silty soils, particularly on sites containing slickspot habitat and native species abundance. Managers should also take into account the low abundance of *K. prostrata* in seeded areas at higher elevations, which suggests lack of seeding success and potential inefficiency of seeding efforts involving this species on such sites. Demonstrated increases in spread with time since seeding suggest that, although negative effects on plant communities in adjacent unseeded areas might not occur instantaneously, they may develop over time. Native species are critical for the perpetuation of resilient ecosystem functioning (Richards et al. 1998); expanded research on effects of highly competitive introduced species such as *K. prostrata* on native plants is needed to inform rehabilitation efforts. Knowledge of environmental and disturbance variables associated with increased abundance and spread, coupled with enhanced understanding of effects on native plant communities, will enable land managers to make ecologically-informed decisions regarding use of *K. prostrata* on the landscape. If

management objectives include increasing species or functional diversity or native species abundance, use of *K. prostrata* may not be effective.

BIBLIOGRAPHY

Anderson, M. J. 2001. A new method for non-parametric multivariate analysis of variance. *Austral Ecology* 26: 32-46.

Archer, S. and D. A. Pyke. 1991. Plant-animal interactions affecting plant establishment and persistence on revegetated rangeland. *Journal of Range Management* 44: 558-565.

Balyan, G. A. 1972. Prostrate summer cypress and its culture in Kirghizia. Isdatel'stvo, Frunze, Kirghizstan. (Translated from Russian, 1979). National Technical Information Service TT77- 59026.

Belnap, J., and D. J. Eldridge. 2001. Disturbance and recovery of biological soil crusts. *In*: J. Belnap and O. L. Lange [EDS.]. Biological soil crusts: structure, function, and management. New York, New York, USA: Springer Verlag. p. 363-384.

Belnap, J. and S. L. Phillips. 2001. Soil biota in an ungrazed grassland: response to annual grass (*Bromus tectorum*) invasion. *Ecological Applications* 11: 1261-1275.

Bilbrough, C. J. and J. H. Richards. 1993. Growth of sagebrush and bitterbrush following simulated winter browsing: Mechanisms for tolerance. *Ecology* 74: 481-492.

Bray, J. R. and J. T. Curtis. 1957. An ordination of the upland forest communities in southern Wisconsin. *Ecological Monographs* 27: 325-349.

Brooks, M. L. and D. A. Pyke. 2001. Invasive plants and fire in the deserts of North America. *in:* K. E. M. Galley and T. P. Wilson [EDS.]. Proceedings of the invasive species workshop: The role of fire in the control and spread of invasive species; Tall Timbers Research Station, Tallahassee, FL, USA: the First National Congress on Fire Ecology, Prevention, and Management. Miscellaneous Publication No. 11. p. 1-14.

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.

Bushey, C. L. 1987. Short-term vegetative response to prescribed burning in the sagebrush/grass ecosystem of the northern Great Basin; three years of postburn data from the demonstration of prescribed burning on selected Bureau of Land Management districts. Systems for Environmental Management Final Report. Missoula, MT, USA: Cooperative Agreement 22-C-4-INT-33. p. 77.

Chambers, J. C., R. W. Brown, and B. D. Williams. 1994. An evaluation of reclamation succession on Idaho's phosphate mines. *Restoration Ecology* 2: 4-16.

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

Chambers, J. C., E. Leger, and E. Goergen. 2009. Cold desert fire and invasive species management: Resources, strategies, tactics, and response. *Rangelands* 31: 14-20.

Chen, S. P., Y. F. Bai, G. H. Lin, Y. Liang, and X. G. Han. 2005. Effects of grazing on photosynthetic characteristics of major steppe species in the Xilin River Basin, Inner Mongolia, China. *Photosynthetica* 43: 559-565.

Cox, R. D. and V. J. Anderson. 2004. Increasing native diversity of cheatgrass-dominated rangeland through assisted succession. *Rangeland Ecology and Management* 57: 203-210.

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.

Davenport, B. W. 2005. Cattle utilization of forage kochia (*Kochia prostrata*) and its relation to forage quality and plant morphological characteristics [thesis]. Logan, UT, USA: Utah State University. 75 p.

Dukes, J. S. and H. A. Mooney. 1999. Does global change increase the success of biological invaders? *Trends in Ecology & Evolution* 14: 135-139.

Duniway, M. C., B. T. Bestelmeyer, and A. Tugel. 2010. Soil processes and properties that distinguish ecological sites and states. *Rangelands* 32: 9-15.

Eissenstat, D. M., and M. M. Caldwell. 1988. Competitive ability is linked to rates of water extraction. *Oecologia* 75: 1-7.

Eiswerth, M. E., K. Krauter, S. R. Swanson, and M. Zielinski. 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.

Eldridge, D. 2004. Mounds of the American badger (*Taxidea taxus*): Significant features of North American shrub-steppe ecosystems. *Journal of Mammalogy* 85: 1060-1067.

Eldgridge, D. and M. Wicklow-Howard. 2004. Post-Fire recovery of microbiotic soil crusts. Final year monitoring results for two sites on the Lower Snake River District, BLM. Boise, ID, USA: USDI Bureau of Land Management and Boise State University.

ESRI Incorporated. 2009. ArcMap GIS Software, Version 9.3.1. Environmental Systems Research Institute, Inc., Redlands, CA, USA.

Evans, R. D., R. Rimer, L. Sperry, and J. Belnap. 2001. Exotic plant invasion alters nitrogen dynamics in an arid grassland. *Ecological Applications* 11: 1301-1310.

Federal Register. 2009. Endangered and threatened wildlife and plants; Listing *Lepidium papilliferum* (slickspot peppergrass) as a threatened species throughout its range; Final rule October 8, 2009. US Department of Interior, Fish and Wildlife Service, Volume 74, Number 194 p. 52013-52064. Available at: http://frwebgate.access.gpo.gov/cgi-bin/getdoc.cgi?dbname=2009_register&docid=fr08oc09-17. Accessed 1 October 2010.

Fleischner, T. L. 1994. Ecological costs of livestock grazing in western North America. *Conservation Biology* 8: 629-644.

Gelbard, J. L. and J. Belnap. 2003. Roads as conduits for exotic plant invasions in a semiarid landscape. *Conservation Biology* 17: 420-432.

Gray, K. J. 2004. The use of forage kochia to benefit a north central Nevada mule deer herd. *In:* D. Zobell, B. Waldron, and K. Wood [compilers]. Proceedings: Forage kochia workshop and tour. 9-10 November, 2004. Utah State University, Logan, UT, USA. p. 50.

Harris, G. A. 1977. Root phenology as a factor of competition among grass seedlings. *Journal of Range Management* 30: 172-177.

Harrison, R. D., N. J. Chatterton, B. L. Waldron, B.W. Davenport, A. J. Palazzo, W. H. Horton, and K. H. Asay. 2000. Forage kochia: Its compatibility and potential aggressiveness on intermountain rangelands. Utah Agricultural Experiment Station Research Report 162. Utah State University, Logan, Utah, USA. 66p.

Harrison, R. D., B. L. Waldron, K. B. Jensen, R. Page, T. A. Monaco, W. H. Horton, and A. J. Palazzo. 2002. Forage kochia helps fight range fires. *Rangelands* 24: 3-7.

Harrison, R.D. 2004. Adaptation and compatibility of forage kochia. *In:* D. Zobell, B. Waldron, and K. Wood [compilers]. Proceedings: Forage kochia workshop and tour. 9-10 November, 2004. Utah State University, Logan, UT, USA. p. 16-18.

Henderson, D. C. and M. A. Naeth. 2005. Multi-scale impacts of crested wheatgrass invasion in mixed-grass prairie. *Biological Invasions* 7: 639-650.

Hickman, K. R. 2004. Grazing management effects on plant species diversity in tallgrass prairie. *Journal of Range Management* 57: 58-65.

Howard, Janet L. 1999. *Artemisia tridentata* subsp. *wyomingensis*. *In:* Fire Effects Information System, [Online]. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fire Sciences Laboratory (Producer). Available at: http://www.fs.fed.us/database/feis/. Accessed 1 April 2011.

Hulet, A., B. A. Roundy, and B. Jessop. 2010. Crested wheatgrass control and native species establishment in Utah. *Rangeland Ecology and Management* 63: 450-460.

Idaho Geospatial Data Clearinghouse. 2004. Digital Elevation of Tile 07, Idaho with a Horizontal Grid Spacing of 10-meters: Idaho Geospatial Data Clearinghouse, Moscow, ID, USA.
Available at:
http://data.insideidaho.org/data/IGDC/archive/elev10m/elev10m_tile07_igdc.tgz.
Accessed 10 January 2010.

James, J. J., and T. Svejcar. 2010. Limitations to postfire seedling establishment: The role of seeding technology, water availability, and invasive plant abundance. *Rangeland Ecology and Management* 63: 491-495.

Jessop, B. D., and V. J. Anderson. 2007. Cheatgrass invasion in salt desert shrublands: Benefits of postfire reclamation. *Rangeland Ecology and Management* 60: 235-243.

Jones, T. A. 1999. Use of native plants for sagebrush steppe restoration. *In*: Entwistle, P. G., A. M. Debolt, J. H. Kaltenecker, and K. Steenhof [EDS.]. Proceedings: Sagebrush steppe ecosystems symposium; 21-23 June 1999; Boise, ID, USA: Boise State University. p. 73-77.

Karimi, G., M. Ghorbanli, H. Heidari, R. A. Khavari Nejad, and M. H. Assareh. 2005. The effects of NaCl on growth, water relations, osmolytes and ion content in *Kochia prostrata*. *Biologia Plantarum* 49: 301-304.

Kitchen, S. G. and S. B. Monsen. 2001. *K. prostrata* seed germination response to storage time and temperature. *Journal of Range Management* 54: 299-306.

Kruskal, J. B. 1964. Nonmetric multidimensional scaling: a numerical method. *Psychometrika* 29: 115-129.

Lambers, H., F. S. Chapin III, and T. L. Pons. 1998. Plant Physiological Ecology. New York, New York, USA: Springer-Verlag. 550 p.

Lee, J. E. 2008. Pygmy rabbit (*Brachylagus idahoensis)* habitat use, activity patterns and conservation in relationship to habitat treatments [thesis]. Provo, UT, USA: Department of Plant and Wildlife Sciences, Brigham Young University. 46 p.

Leonard, E. D., T. A. Monaco, J. M. Stark, and R. J. Ryel. 2008. Invasive forb, annual grass, and exotic shrub competition with three sagebrush steppe growth forms: Acquisition of a spring [15]N Tracer. *Invasive Plant Science and Management* 1: 168-177.

Lym, R. G. and D. R. Kirby. 1987. Cattle forage behavior in leafy spurge (*Euphorbia esula*)-infested rangeland. *Weed Technology* 1: 314-318.

Mack, R. N., D. Simberloff, W. M. Lonsdale, H. Evans, M. Clout and F. A. Bazzaz. 2000. Biotic invasions: causes, epidemiology, global consequences, and control. *Ecological Applications* 10: 689-710.

McArthur, E. D., B. C. Giunta, and A. P. Plummer. 1977. Shrubs for restoration of depleted range and disturbed areas. *Utah Science* 35: 28-33.

McArthur, E. D., A. C. Blauer, and R. Stevens. 1990. *K. prostrata* competition with cheatgrass in central Utah. *In*: E. D. McArthur, E. M Romney, S. D. Smith, P. T. Tueller [compilers]. Proceedings: Symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management; 5-7 April, 1989. Las Vegas, NV, USA. Gen. Tech. Report INT-276. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Research Station. p. 351.

McArthur, E. D., S. C. Sanderson, and J. N. Davis. 1996. Adaptation of forage kochia accession across an environmental gradient in Rush Valley, Utah. *Arid Soil Research and Rehabilitation* 10: 125-138.

McArthur, T.O. 2004. Emergency fire rehabilitation of BLM sands in the Great Basin: Revegetation & monitoring [thesis]. Corvallis, OR, USA: Oregon State University. 68 p.

McCune, B. and J. B. Grace. 2002. Analysis of Ecological Communities, 2nd Printing. Gleneden Beach, Oregon, U.S.A.: MjM Software Design. 300 p.

McCune, B. 2006. Non-parametric habitat models with automatic interactions. *Journal of Vegetation Science* 17: 819-830.

McCune, B. and M. J. Mefford. 2009. HyperNiche. Nonparametric Multivariate Habitat Modeling. Version 2.10. Gleneden Beach, OR, USA: MjM Software.

McCune, B. and M. J. Mefford. 2011. PC-ORD. Mulitvariate Analysis of Ecological Data. Version 6.0. Gleneden Beach, OR, USA: MjM Software.

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.

Melgoza, G. and R. S. Nowak. 1991. Competition between cheatgrass and two native species after fire: Implication from observations and measurements in root distribution. *Journal of Range Management* 44: 27-33.

Miller, R. F. and L. L. Eddleman. 2001. Spatial and temporal changes of sage grouse habitat in the sagebrush biome. Oregon State University Agricultural Experiment Station Technical Bulletin 151. Corvallis, OR, USA. 39 p.

Miller, R. F., S. T. Knick, D. A. Pyke, C. W. Meinke, S. E. Hanser, M. J. Wisdom, and A. L. Hild. 2011. Characteristics of sagebrush habitats and limitations to long-term conservation. Pp. 145-184 in S. T. Knick and J. W. Connelly (EDS). Greater Sage-Grouse: ecology and conservation of a landscape species and its habitats. Studies in Avian Biology (vol. 38), University of California Press, Berkeley, CA.

Monaco, T. A., B. L. Waldron, R. L. Newhall, and W. H. Horton. 2003. Re-establishing perennial vegetation in cheatgrass monocultures. *Rangelands* 25: 26-29.

Moody, M. E., and R. N. Mack. 1988. Controlling the spread of plant invasions: The importance of nascent foci. *Journal of Applied Ecology* 25: 1009-1021.

Moseley, R. K. 2005. Report on the conservation status of *Lepidium davisii.* Idaho Department of Fish and Game, Conservation Data Center, Boise, ID.

Newhall, R. L., T. A. Monaco, W. H. Horton, R. D. Harrison, and R. J. Page. 2004. Rehabilitating salt-desert ecosystems following wildfire and wind erosion. *Rangelands* 26: 3-7.

Paige, C., and S. A. Ritter. 1999. Birds in a sagebrush sea: managing sagebrush habitats for bird communities. Partners in Flight Western Working Group, Boise, ID, USA. 47 p.

Participants of the Snake River Birds of Prey NCA habitat restoration workshop. 1999. Snake River Birds of Prey NCA habitat restoration summary and recommendations. *In*: Entwistle, P. G., A. M. Debolt, J. H. Kaltenecker, and K. Steenhof [compilers]. Proceedings: Sagebrush steppe ecosystems symposium; 21-23 June 1999; Boise State University, Boise, ID, USA. p. 91-97.

Peck, J.E. 2010. Multivariate Analysis for Community Ecologists: Step-by-Step using *PC-ORD*. Gleneden Beach, OR, USA: MjM Software Design. 162 p.

Pellant, M. and S. B. Monsen. 1993. Rehabilitation on public rangelands in Idaho, USA: a change in emphasis from grass monocultures. Pages 778-779 *In*: anonymous [compilers], Proceedings of the XVII International Grassland Congress, 8-21 February 1993, Palmerston North, Hamilton, and Lincoln, New Zealand and Rockhampton, Australia. New Zealand Grassland Association, Tropical Grasslands Society of Australia, New Zealand Society of Animal Production, Australian Society of Animal Production – Queensland Branch, and New Zealand Institute of Agricultural Science. Keeling & Mundy, Palmerston North, New Zealand.

Pellant, M. 1994. History and applications of the Intermountain greenstripping program, p. 63–68. *In*: S.B. Monsen and S.G. Kitchen [EDS.] Proceedings of the Ecology and Management of Annual Rangelands, USDA. Forest Service, General Technical Report INT-313.

Pellant, M., B. Abbey, and S. Karl. 2004. Restoring the Great Basin desert, USA: Integrating science, management, and people. *Environmental Monitoring and Assessment* 99: 169-179.

Ponzetti, J. M. and B. P. McCune. 2001. Biotic soil crusts of Oregon's shrub steppe: Community composition in relation to soil chemistry, climate, and livestock activity. *The Bryologist* 104: 212-225.

Ponzetti, J. M., B. McCune, and D. A. Pyke. 2007. Biotic soil crusts in relation to topography, cheatgrass, and fire in the Columbia Basin, WA. *The Bryologist* 113: 292-307.

The PRISM Climate Group, Oregon State University. 2006. United States Average Monthly or Annual Precipitation, 1971-2000. Corvallis, OR, USA. Available at: http://www.prism.oregonstate.edu/products/viewer.phtml?file=/pub/prism/us_30s/grids/ppt/Normals/us_ppt_1971_2000.14.gz&year=1971_2000&vartype=ppt&month=14&status=final. Accessed 15 September 2010.

Pyankov, V. I., E. G. Artyusheva, G. E. Edwards, C. C. Black Jr., and P. S. Soltis. 2001. Phylogenetic analysis of tribe Salsoleae (Chenopodiaceae) based on ribosomal ITS sequences: Implications for the evolution of photosynthetic types. *American Journal of Botany* 88: 1189-1198.

Pyke, D. A. and S. Archer. 1991. Plant-plant interactions affecting plant establishment and persistence on revegetated rangeland. *Journal of Range Management* 44: 550-557

Pyke, D. A. 1999. Invasive exotic plants in sagebrush ecosystems of the Intermountain West. *In*: Entwistle, P. G., A. M. Debolt, J. H. Kaltenecker, and K. Steenhof [EDS.]. Proceedings: Sagebrush steppe ecosystems symposium; 21-23 June 1999; Boise State University, Boise, ID, USA. p. 43-54.

Pyke, D. A., M. L. Brooks, and C. D'Antonio. 2010. Fire as a restoration tool: A decision framework for predicting the control or enhancement of plants using fire. *Restoration Ecology* 18: 274-284.

Pyke, D. A. and S. T. Knick. 2003. Plant invaders, global change and landscape restoration. *In*: Allsopp, N., A. R. Palmer, S. J. Milton, K. P. Kirkman, G. I. H. Kerley, C. R. Hurt, and C. J. Brown [EDS.]. Proceedings: VIIth International Rangelands Congress; 26 July – 1 August 2003; Durban, South Africa. p. 278-288.

R Development Core Team. 2009. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. ISBN 3-900051-07-0, Available at: http://www.R-project.org. Accessed 20 April 2011.

Richards, R. T, J. C. Chambers, and C. Ross. 1998. Use of native plants on federal lands: Policy and practice. *Journal of Range Management* 51: 625-632.

Romo, J. T. and M. R. Haferkamp. 1987. Forage kochia germination response to temperature, water stress, and specific ions. *Agronomy Journal* 79: 27-30.

Romo, J. T. and M. R. Haferkamp. 1988. Comparative water relations in *Artemisia tridentata* Nutt. subsp. *wyomingensis* and *Kochia prostrata* (L.) Shrad. *Journal of Arid Environments* 15: 53-64.

Root, H. T., B. McCune, and P. Neitlich. 2010. Lichen habitat may be enhanced by thinning treatments in young *Tsuga heterophylla- Pseudotsuga menziesii* forests. *The Bryologist* 113: 292-307.

Schlatterer, E. F. 1973. Sagebrush species and subspecies. *Range Improvement Notes* 18: 1-11.

Sperry, L. J., J. Belnap, and R. D. Evans. 2006. *Bromus tectorum* invasion alters nitrogen dynamics in an undisturbed arid grassland ecosystem. *Ecology* 87: 603-615.

Stevens, R., K. R. Jorgensen, E. D. McArthur, and J. N. Davis. 1985. "Immigrant" forage kochia. *Rangelands* 7: 22-23.

Stevens, R. and E. D. McArthur. 1990. "Immigrant" forage kochia competition with halogeton following various seeding techniques. *In:* McArthur, E. D., E. M. Romney, S. D. Smith, P. T. Tueller [compilers] Proceedings: Symposium of cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management; 5-7 April 1989. USDA Forest Service, Intermountain Forest and Range Experiment Station. Ogden, UT, USA. p. 175-180.

Stewart, A., V. J. Anderson, and S. G. Kitchen. 2001. 'Immigrant' forage kochia seed viability as impacted by storage methods. *Journal of Range Management* 54: 396-399.

Thompson, T. W., B. A. Roundy, E. D. McArthur, B. D. Jessop, B. Waldron, and J. N. Davis. 2006. Fire rehabilitation using native and introduced species: A landscape trial. *Rangeland Ecology and Management* 59: 237-248.

Thorpe, A. S., E. T. Aschehoug, D. Z. Atwater, and R. M. Callaway. 2011. Interactions among plants and evolution. *Journal of Ecology*: (in press).

Tuason, T. 2005. 1987-2004 Monitoring results of *Lepidium davisii* in the Mountain Home District. USDI BLM Boise District, Boise, ID, USA.

USDA ARS Germplasm Resources Information Network (GRIN). 2001. GRIN Taxonomy for Plants: *Bassia prostrata* (L.) A.J. Scott. Available at: http://www.ars-grin.gov/cgi-bin/npgs/html/taxon.pl?409953. Accessed 27 September 2010.

USDA-FSA-APFO Aerial Photography Field Office. 2009. Digital Orthoimagery Series of Idaho, 2009, 1-meter, Natural Color and False Color series. USDA-FSA-APFO Aerial Photography Field Office, Salt Lake City, UT, USA. Available at: http://cloud.insideidaho.org/data/imageryBaseMapsEarthCover/orthoimagery/2009_1m_idaho/2009naturalColorAndIR_1m_idaho.lyr. Accessed 20 July 2010.

USDA NRCS. 2008. Soil Survey Geographic (SSURGO) database. Soil Surveys for Ada, Elmore, Payette, Owyhee, and Twin Falls Counties, Idaho. Fort Worth, TX, USA. Available at: http://greatbasin.wr.usgs.gov/ltdl/Default.aspx. Accessed 2 August 2010.

USDA NRCS. 2011. The PLANTS Database (http://plants.usda.gov, 22 February 2011). National Plant Data Center, Baton Rouge, LA, USA. Accessed 3 January 2011.

USDI BLM. 2007. Burned Area Emergency Stabilization and Rehabilitation Handbook- (Public) H-1742-1. Pp 38. Available at: http://www.blm.gov/pgdata/etc/medialib/blm/wo/Information_Resources_Management/policy/blm_handbook.Par.52739.File.dat/h1742-1.pdf. Accessed 27 September 2010.

USDI BLM, Idaho State Offices, Engineering and Geographic Sciences. 2009a. Range Allotments of Idaho: Bureau of Land Management (BLM), Boise, ID, USA. Available at: http://data.insideidaho.org/util/zip.ashx?fn=http://data.insideidaho.org/data/BLM/downdata/statewide/rngAllotment_id_blm.zip. Accessed 1 June 2010.

USDI BLM, Boise District, Twin Falls District Office, Idaho Falls District Office, Idaho State Office. 2009b. Historic Wildfire Perimeters [1937-2008] on or Adjacent to Bureau of Land Management Administered Lands in Idaho: U.S. Bureau of Land Management (BLM), Boise, ID, USA Available at: http://data.insideidaho.org/util/zip.ashx?fn=http://data.insideidaho.org/data/BLM/downdata/statewide/wildfire_id_blm.zip. Accessed 25 September 2010.

USDI BLM. 2009c. Slickspot peppergrass (*Lepidium papilliferum*): Conservation Measures and Implementation Actions for the Jarbidge and Four Rivers Field Offices.

USDI BLM. 2010. Twin Falls District Wildlife Tracts Habitat Enhancement. Environmental assessment ID-210-2008-EA-248. p. 11.

USDI FWS. 2010. Biological opinion on the effects of Bureau of Land Management ongoing livestock grazing actions in Idaho on the slickspot peppergrass (*Lepidium papilliferum*). Boise, ID, USA. 14420-2010-F-0025.

Vitousek, P. M., C. D. Antonio, L. L. Loope, M. Rejmánek, and R. Westbrooks. 1997. Introduced species: A significant component of human caused global change. *New Zealand Journal of Ecology* 21:1-16.

Waldron, B. L., R. D. Harrison, N. J. Chatterton, B. W. Davenport. 2001. Forage kochia: Friend or foe. USDA Forest Service Proceedings RMRS-P-21. p. 210-215.

Waldron, B. L., J.-S. Eun, D. R. Zobell, and K. C. Olson. 2010a. Forage kochia (*Kochia prostrata*) for fall and winter grazing. *Small Ruminant Research* 91: 47-55.

Waldron, B. L., B. W. Davenport, J. C. Malechek, and K. B. Jensen. 2010b. Relative cattle preference of 24 forage kochia (*Kochia prostrata*) entries and its relation to forage nutritive value and morphological characteristics. *Crop Science* 50: 2112-2123.

Western Regional Climate Center. 2008. Mountain Home Idaho, Period of Record Monthly Climate Summary : 8/ 1/1948 to 12/31/2005. Available at: http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?idmoun. Accessed 15 September 2010.

West N. E. and J. A. Young. 2000. Intermountain valleys and lower mountain slopes. In: Barbour M. G. and W. D. Billing [EDS.]. North American terrestrial vegetation. Cambridge, United Kingdom: Cambridge University Press. p. 255–284.

Whisenant, S.G. 1990. Changing fire frequencies on Idaho's Snake River Plains: Ecological and management implications. *In:*   E. D. McArthur, E. M. Romney, S. D. Smith, and P. T. Tueller [compilers]. Proceedings—symposium on cheatgrass invasion, shrub die-off, and other aspects of shrub biology and management. USDA Forest Service, Intermountain Research Station, Ogden, UT, USA: General Technical Report INT-276. p. 4-10.

White, J. P. and I. C. Robertson. 2009. An unusual life history strategy in *Lepidium papilliferum* (Brassicaceae), a rare mustard endemic to southwestern Idaho. *Northwest Science* 83: 287-290.

Wisdom, M. J. and J. C. Chambers. 2009. A landscape approach for ecologically based management of Great Basin Shrublands. *Restoration Ecology* 17: 740-749.

Wood, M. 2006. Nutritious treat for cattle and wildlife: Forage kochia! *Agricultural Research* 54: 18-19.

Zhao, W. Y., J. L. Li, and J. G. Qi. 2007. Changes in vegetation diversity structure in response to heavy grazing pressure in the northern Tianshan Mountains, China. *Journal of Arid Environments* 68: 465-479.

ZoBell, D. R., B. L. Waldron, K. C. Olson, R. D. Harrison, and H. Jensen. 2003. Forage kochia for fall/winter grazing. Utah State University and USDA-ARS, AG-2003-07. Logan, UT, USA.

APPENDICES

Appendix A. Characteristics of 28 study sites located throughout the Snake River Plain and surrounding uplands in southwestern Idaho, USA. All sites were seeded in post-fire rehabilitation or greenstrips by the USDI BLM.

Table 1. Location and environmental characteristics of sampled sites.

Site Name	Idaho BLM Field Office	UTM Coordinates	Elevation (m)	Average annual Precipitation (mm) 1971-2000	Soils: Ecological Site (NRCS)
Apple Valley Greenstrip 1	Four Rivers	11N 0513997, 4862278	770	252	Loamy 8-12 Artrw8/pssps-Acth7
Apple Valley Greenstrip 2	Four Rivers	11N 0518236, 4858318	777	234	Loamy 8-12 Artrw8/pssps-Acth7
Big Draw	Jarbidge	11N 0617520, 4720890	1242	377	Loamy 8-12 Artrw8/pssps-Acth7
Brown Dune N	Four Rivers	11N 0612291, 4749830	861	196	Sandy Loam 8-12 Artrw8/achy
Brown Dune S	Four Rivers	11N 0611493, 4747951	950	198	Sandy Loam 8-12 Artrw8/achy
Canyon Creek Greenstrip	Four Rivers	11N 0584144, 4772477	915	192	Silty 7-10 Krla/achy
Cassia Gulch	Jarbidge	11N 0666148, 4746634	1005	197	Loamy 8-12 Artrw8/pssps-Acth7
Disc Chain	Four Rivers	11N 0583397, 4771192	909	196	Silty 7-10 Krla/achy
Disc Chain SS	Four Rivers	11N 0582826, 4771682	909	204	Loamy 8-12 Artrw8/pssps-Acth7
Deadman Gulch	Jarbidge	11N 0661273, 4714862	1186	355	Loamy 8-12 Artrw8/pssps-Acth7
Echo Water Tank	Jarbidge	11N 0610454, 4721183	1155	221	Loamy 8-12 Artrw8/pssps-Acth7
Flat Broke Drill	Bruneau	11N 0600467, 4722587	1123	219	Sandy Loam 8-12 Artrw8/achy
Flat Broke Greenstrip	Bruneau	11N 0599581, 4725759	1078	214	Sandy Loam 8-12 Artrw8/achy

Table 1 (Continued)

Site Name	Idaho BLM Field Office	UTM Coordinates	Elevation (m)	Average annual Precipitation (mm) 1971-2000	Soils: Ecological Site (NRCS)
Frog Holler	Jarbidge	11N 0646355, 4757491	912	188	Loamy 8-12 Artrw8/pssps-Acth7
Fritz Spur	Jarbidge	11N 0616035, 4722336	1208	366	Loamy 8-12 Artrw8/pssps-Acth7
Fritz Spur Low	Jarbidge	11N 0613320, 4720408	1179	302	Loamy 8-12 Artrw8/pssps-Acth7
I84 MP85	Four Rivers	11N 0596861, 4785139	975	201	Loamy 8-12 Artrw8/pssps-Acth7
I84 MP85 E	Four Rivers	11N 0598759, 4782761	979	211	Loamy 8-12 Artrw8/pssps-Acth7
Little Big South	Jarbidge	11N 0614087, 4727180	1152	232	Loamy 8-12 Artrw8/pssps-Acth7
Little Big West	Jarbidge	11N 0611113, 4721335	1163	223	Loamy 8-12 Artrw8/pssps-Acth7
Orchard	Four Rivers	11N 0555588, 4799441	889	181	Loamy 8-12 Artrw8/pssps-Acth7
Poen	Four Rivers	11N 0549524, 4803107	939	207	Loamy 8-12 Artrw8/pssps-Acth7
RRMP	Four Rivers	11N 0618634, 4760120	886	189	Loamy 8-12 Artrw8/pssps-Acth7
Simco Restoration	Four Rivers	11N 0583889, 4771516	912	176	Silty 7-10 Krla/achy
Swan Falls Greenstrip 2	Four Rivers	11N 0547593, 4803954	920	206	Loamy 8-12 Artrw8/pssps-Acth7
Swan Falls Greenstrip 3	Four Rivers	11N 0547424, 4800479	905	178	Loamy 8-12 Artrw8/pssps-Acth7

Table 1 (Continued)

Site Name	Idaho BLM Field Office	UTM Coordinates	Elevation (m)	Average annual Precipitation (mm) 1971-2000	Soils: Ecological Site (NRCS)
Swan Falls Greenstrip 4	Four Rivers	11N 0546379, 4802283	906	198	Loamy 8-12 Artrw8/pssps-Acth7
Swan Falls Greenstrip 5	Four Rivers	11N 0550211, 4801997	928	212	Stony 10-12 Artrw8/pssps

Table 2. Seeding treatments on sampled sites. "YS" = year of seeding.

Site Name	Year of Seeding	No. of Fires since seeding (YS - 2008)	Herbicide	Treatment Method
Apple Valley Greenstrip 1	1997	0	Oust®	Greenstrip: Drill Seeding
Apple Valley Greenstrip 2	1997	0	Oust®	Greenstrip: Drill Seeding
Big Draw	1999	0	unknown	Ground Seeding: Drill
Brown Dune N	2007	0	unknown	Hand Planting
Brown Dune S	2007	0	unknown	Hand Planting
Canyon Creek Greenstrip	1997	1	Oust®	Greenstrip: Drill Seeding
Cassia Gulch	1992	0	unknown	Ground Seeding: Broadcast
Disc Chain	1986	1	Oust®	Greenstrip: Drill Seeding
Disc Chain SS	1986	1	unknown	Greenstrip: Drill Seeding
Deadman Gulch	1995	2	Oust®	Greenstrip: Drill Seeding
Echo Water Tank	1999	0	unknown	Ground Seeding: Drill
Flat Broke Drill	2000	0	Oust®	Ground Seeding: Drill
Flat Broke Greenstrip	2000	0	Oust®	Ground Seeding: Drill
Frog Holler	2000	0	unknown	Ground Seeding: Drill & Aerial
Fritz Spur	2000	0	unknown	Ground Seeding: Drill
Fritz Spur Low	2000	0	unknown	Ground Seeding: Drill
I84 MP85	1989	2	unknown	Ground Seeding: Drill
I84 MP85 E	1989	2	unknown	Ground Seeding: Drill
Little Big South	1999	0	unknown	Ground Seeding: Drill
Little Big West	1999	0	unknown	Ground Seeding: Drill
Orchard	1995	1	Oust®	Aerial seeding
Poen	1994	1	Oust®	Ground Seeding: Drill & Aerial
RRMP	1986	0	Oust®	Greenstrip: Drill Seeding
Simco Restoration	1998	0	Oust®	Greenstrip: Drill Seeding
Swan Falls Greenstrip 2	1994	1	Oust®	Greenstrip: Drill Seeding

Table 2
(Continued)

Site Name	Year of Seeding	No. of Fires since seeding (YS - 2008)	Herbicide	Treatment Method
Swan Falls Greenstrip 3	1994	2	Oust®	Greenstrip: Drill Seeding
Swan Falls Greenstrip 4	1994	1	Oust®	Greenstrip: Drill Seeding
Swan Falls Greenstrip 5	1994	1	Oust®	Greenstrip: Drill Seeding

Table 3. Grazing characteristics of sampled sites. Dates of use varied by year for some sites.

Site Name	Grazing Allotment	First Post-seeding Grazing Year	Mean AUMs/Ha (Actual use)	Predominant Season	Approximate Dates of Use
Apple Valley Greenstrip 1	Black Canyon	1999	0.09	Winter, Spring	12/15-1/31, 3/1-4/16
Apple Valley Greenstrip 2	Black Canyon	1999	0.09	Winter, Spring	12/15-1/31, 3/1-4/16
Big Draw	Flat Top	2001	0.28	Winter, Spring	3/1-6/18, 12/18-2/28
Brown Dune N	Brown's Gulch	2009	0.70	Fall, Winter, Spring	3/1-5/5, 10/1-2/28
Brown Dune S	Brown's Gulch	2009	0.70	Fall, Winter, Spring	3/1-5/5, 10/1-2/28
Canyon Creek Greenstrip	Airbase	1999	0.19	Winter	11/10-2/28
Cassia Gulch	none	1994	0.00	none	none
Disc Chain	Sunnyside Winter	1988	0.10	Winter	12/16-2/28
Disc Chain SS	Sunnyside Winter	1988	0.10	Winter	12/16-2/28
Deadman Gulch	Devil Creek/Balanced Rock	1997	0.19	Winter, Spring	3/1-5/21, 11/15-2/28
Echo Water Tank	Bruneau Hill	2001	0.23	Winter	11/1-2/28
Flat Broke Drill	Center	2002	0.14	Winter, Spring	11/1-5/31
Flat Broke Greenstrip	Center	2002	0.14	Winter, Spring	11/1-5/31
Frog Holler	Thompson	2002	0.30	Year round	3/1-2/28
Fritz Spur	Flat Top	2002	0.28	Winter, Spring	3/1-6/18, 12/18-2/28
Fritz Spur Low	Bruneau Hill	2002	0.23	Winter	11/1-2/28
I84 MP85	Martha Avenue (3 & 4)	1991	0.19	Winter, Spring	12/15-2/8, 3/1-3/31

Table 3 (Continued)

Site Name	Grazing Allotment	First Post-seeding Grazing Year	Mean AUMs/Ha (Actual use)	Predominant Season	Approximate Dates of Use
I84 MP85 E	Martha Avenue (3 & 4)	1991	0.19	Winter, Spring	12/15-2/8, 3/1-3/31
Little Big South	Canyon View	2001	0.66	Fall, Winter, Spring	rotational
Little Big West	Bruneau Hill	2001	0.23	Winter	11/1-2/28
Orchard	Sunnyside Spring/Fall	1997	0.05	Spring, Fall	4/1-6/25, 11/20-1/20
Poen	Sunnyside Spring/Fall	1996	0.05	Spring, Fall	4/1-6/25, 11/20-1/20
RRMP	Chalk Flat (S of I84)	1988	0.35	Winter, Spring	3/1-4/10, 10/10-2/28
Simco Restoration	Sunnyside Winter	2000	0.09	Winter	12/16-2/28
Swan Falls Greenstrip 2	Sunnyside Spring/Fall	1996	0.05	Spring, Fall	4/1-6/25, 11/20-1/20
Swan Falls Greenstrip 3	Sunnyside Winter	1996	0.10	Winter	12/16-2/28
Swan Falls Greenstrip 4	Sunnyside Spring/Fall	1996	0.05	Spring, Fall	4/1-6/25, 11/20-1/20
Swan Falls Greenstrip 5	Sunnyside Spring/Fall	1996	0.05	Spring, Fall	4/1-6/25, 11/20-1/20

Appendix B. Environmental and site variables assessed in relation to *K. prostrata* abundance and spread, NMS ordination of sample units in species space including correlation coefficients (r) of species and environmental variables with predominant axes, and response surfaces for NPMR models of spread metrics.

Table 1. Environmental and site treatment variables used in NMS ordinations and NPMR models of *K. prostrata* spread.

Environmental Variables (Quantitative)	Minimum	Maximum	Mean	Candidate Predictor for NPMR?
Recruitment margin, m (unseeded only; site level mean distance of margin from seeding boundary)	0	197	30	-
Distance of the farthest *K. prostrata* from seeding boundary m (unseeded only)	0	709	208	-
Time since seeding, yr	3	24	14	Y
Elevation, m	770	1242	990	Y
Precipitation, mm (Mean Annual, 1971-2008)	176	377	226	Y
No. of fires (1937-2008)	1	7	2.36	Y
No. of fires since seeding (through 2008)	0	2	0.6	-
Years of grazing since seeding (2 yr post-seeding through summer 2010)	1	22	12	Y
Mean AUMs/Hectare (2 yr post-seeding through summer 2010)	0.21	0	0.7	Y
Coefficient of variation in yearly grazing (STDEV/MEAN*100)	0	69	33	Y
Litter cover (%)	7.33	92	41	Y
Rock cover (%)	0	7.33	0.94	Y
Moss cover (%)	0	32.67	6.02	Y
Biological crust cover (%)	0	25	3.3	Y
Soil cover (%)	46.67	100	88.5	Y
Badger mound cover (%)	0	8.67	0.89	Y
Cow manure cover (%)	0	1.33	0.35	Y
Slickspot cover (%)	0	14	2.38	-
Environmental Variables (Categorical)				
Treatment method: (1) Greenstrip: Drill seeding, (2) Ground seeding: Drill, (3) Hand planting, (4) Ground seeding: Broadcast, (5) Ground seeding: Drill & aerial, (6) Aerial				-
Herbicide: (1) Oust®, (2) Unknown				-

Table 1 (Continued)

Environmental Variables (Categorical)

Predominant grazing season:(1) Winter, (2) Winter, Spring, (3) Spring, Fall, (4) Fall, Winter, Spring, (5) None, (6) Year round   -

Soils (Ecological Site): (1) Loamy 8-12 Artrw8/pssp-Acth7, (2) Sandy Loam 8-12 Artrw8/achy, (3) Silty 7-10 Krla/achy, (4) Stony 10-12 Artrw8/pssp   -
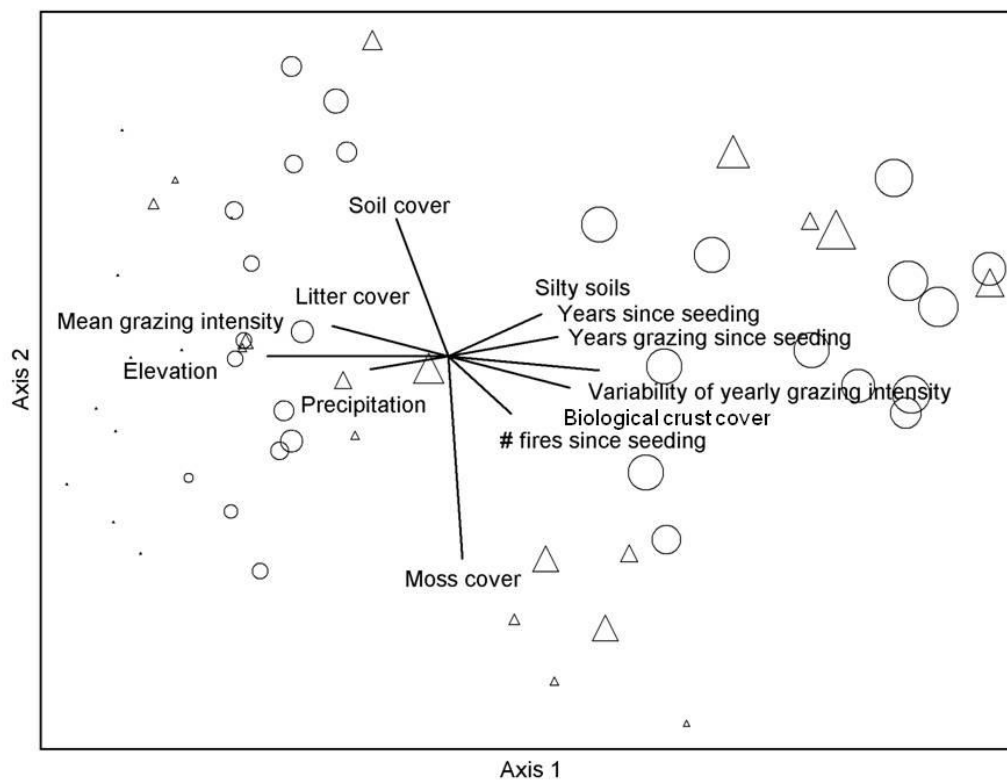
Figure 1. NMS ordination of sample units in species space across all sites. Points represent sample units and distance between points indicates similarity of community composition in sample units. Size of the points indicates abundance of *K. prostrata* in each sample unit; circles represent seeded sample units and triangles represent unseeded sample units. Environmental variables with notable correlations with the ordination axes (|r| >0.40) are indicated by vector lines, with the length of the line representing the strength of the correlation with axes.

Table 2. Pearson's correlation coefficients (r) of species and environmental variables with NMS axes in the ordination of sample units in species space. Only those with |r| ≥ 0.40 with at least one axis (in bold) are shown.

	Axis 1	Axis 2	Axis 3
Variance explained by axes (%)	**70**	**11**	**10**
Pearson's correlation coefficients (r)			
Species cover			
Kochia prostrata	**.833**	.195	.280
Bromus tectorum	**-.736**	**-.562**	.038
Agropyron cristatum	**-.615**	-.100	**-.509**
Poa secunda	**-.695**	**.608**	-.161
Artemisia tridentata ssp. *wyomingensis*	**-.432**	-.003	.312
Ceratocephala testiculata	**.848**	.108	.124
Lepidium perfoliatum	**.405**	-.074	.274
Environmental variables			
Distance to farthest *Kochia prostrata* individual (m)	-.068	-.188	**-.497**
Years since seeding	**.486**	.210	-.084
Elevation (m)	**-.626**	-.031	.213
Mean annual precipitation (mm)	**-.412**	-.162	-.120
No. of fires since seeding (through 2008)	**.418**	-.126	.048
Years grazing since seeding	**.486**	.210	-.084
Mean grazing intensity (Mean AUMs/ha)	**-.502**	.023	.167
Variability in yearly grazing intensity (CV)	**.571**	-.169	-.309
Litter cover	**-.501**	.261	-.170
Moss cover	.177	**-.661**	.000
Biological crust cover	**.512**	-.260	.007
Soil cover	-.337	**.545**	-.061
Silty soil ecological sites	**.450**	.304	-.047

A.

Recruitment margin (sqrt transformed)

Soil cover

Variability in yearly grazing intensity (CV)
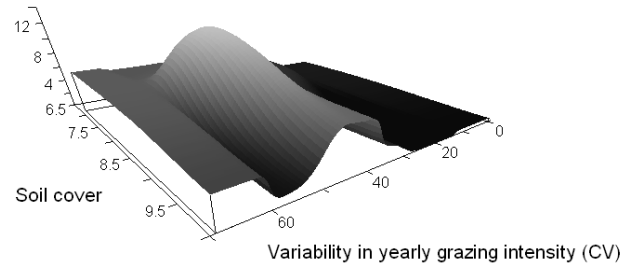
B.

Distance to farthest forage kochia (sqrt transformed)

Rock cover (sqrt transformed)

Years since seeding
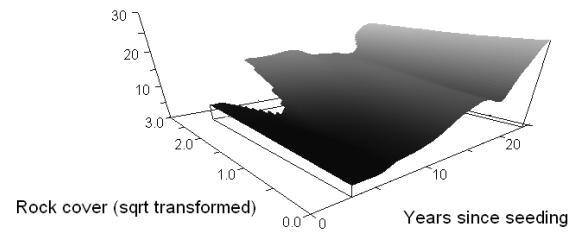
Figure 2. Response surfaces of NPMR models for (A.) distance of the *K. prostrata* recruitment margin from the seeding boundary (minimum N* = 0.2) and (B.) distance of the farthest *K. prostrata* from the seeding boundary (minimum N*=0). Distances are plotted in relation to the two most influential predictors for each model.

Appendix C. Species sampled across all sites.

Table 1. Species sorted by functional groups across all sampled sites. Scientific names, common names, and authorities were derived from the USDA PLANTS database (USDA 2011).

Species by Functional Group	Authority	Common Name
Exotic Annual Forbs		
Bassia scoparia	(L.) A.J. Scott	burningbush
Ceratocephala testiculata	(Crantz) Roth	curveseed butterwort
Descurainia sophia	(L.) Webb ex Prantl	herb sophia
Halogeton glomeratus	(M. Bieb.) C.A. Mey.	saltlover
Holosteum umbellatum	L.	jagged chickweed
Lactuca serriola	L.	prickly lettuce
Lepidium perfoliatum	L.	clasping pepperweed
Salsola kali	L.	Russian thistle
Sisymbrium altissimum	L.	tall tumblemustard
Tragopogon dubius	Scop.	yellow salsify
Native Annual Forbs		
Amsinckia tessellata	A. Gray	bristly fiddleneck
Descurainia pinnata	(Walter) Britton	western tansymustard
Epilobium brachycarpum	C. Presl	tall annual willowherb
Gayophytum sp.	A. Juss.	groundsmoke
Lappula occidentalis	(S. Watson) Greene	flatspine stickseed
Exotic Perennial Forbs		
Chondrilla juncea	L.	rush skeletonweed
Native Perennial Forbs		
Achillea millefolium	L.	common yarrow
Linum sp.	L.	flax
Phlox. sp.	Richardson	phlox
Exotic Annual Grasses		
Bromus tectorum	L.	cheatgrass
Native Annual Grasses		
Vulpia sp.	Nutt.	fescue
Exotic Perennial Grasses		
Agropyron cristatum	(L.) Gaertn.	crested wheatgrass
Agropyron intermedium	(Host) P. Beauv.	intermediate wheatgrass
Poa bulbosa	L.	bulbous bluegrass
Psathyrostachys juncea	(Fisch.) Nevski	Russian wildrye

Table 1 (Continued)

Species by Functional Group	Authority	Common Name
Native Perennial Grasses		
Achnatherum hymenoides	(Roem. & Schult.) Barkworth	Indian ricegrass
Elymus elymoides	(Raf.) Swezey	squirreltail
Hesperostipa comata	(Trin. & Rupr.) Barkworth	needle and thread grass
Leymus cinereus	(Scribn. & Merr.) A. Löve	basin wildrye
Pascopyrum smithii	(Rydb.) A. Löve	western wheatgrass
Poa secunda	J. Presl	Sandberg bluegrass
Pseudoroegneria spicata	(Pursh) A. Löve	bluebunch wheatgrass
Exotic Perennial Shrubs		
Kochia prostrata (now *Bassia prostrata*)	(L.) Schrad	forage kochia
Native Perennial Shrubs		
Artemisia rigida	(Nutt.) A. Gray	scabland sagebrush
Artemisia tridentata ssp. *wyomingensis*	Nutt., Beetle & Young	Wyoming big sagebrush
Atriplex canescens	(Pursh) Nutt.	fourwing saltbrush
Atriplex confertifolia	(Torr. & Frém.) S. Watson	shadscale saltbrush
Chrysothamnus viscidiflorus	(Hook.) Nutt.	yellow rabbitbrush
Picrothamnus desertorum	Nutt.	bud sagebrush

Appendix D. Rapid assessment of *Kochia prostrata* abundance in slickspots in the Poen fire rehabilitation area, Snake River Birds of Prey National Conservation Area, Idaho.

I conducted a rapid assessment of *Kochia prostrata* abundance within slickspots in the Poen fire rehabilitation area in the Snake River Birds of Prey National Conservation Area, Idaho. Prior to a fire in 1994, the site had been dominated by *Artemisia tridentata* ssp. *wyomingensis* with a *Bromus tectorum* understory, and provided habitat for many small mammal species that were food sources for birds of prey. The site also contained slickspots, which provide habitat for the federally threatened endemic mustard, *Lepidium papilliferum*. The site burned in summer, 1994, was aerially and drill seeded with *K. prostrata* and other species in the fall of 1994, and burned once again in 1996 (Appendix A, Table 2). After the initial post-fire treatment in 1994, slickspots were monitored, and results suggested that populations of the endangered *Lepidium papilliferum* had decreased in years following seeding with *K. prostrata* and the subsequent fire. Some studies have indicated that *K. prostrata* is invasive in slickspots (Harrison et al. 2000), and my study was intended as a baseline for further monitoring of *K. prostrata* abundance in these sensitive habitats. For more site information, see "Paradise Lost, Seeded, and Lost Again" by John Doremus (ID State Office Files, USDI BLM, 2010 pers. comm.)

Methods

Sampling occurred on July 21, 2010 in the Poen Fire rehabilitation seeded site and surrounding unseeded area (Fig. 1). The site was accessed near Initial Point (43º22´N, 116º23´W, elevation = 939 m), south of Kuna, ID. I determined the approximate seeding boundary using a shapefile uploaded onto a handheld GPS unit. I followed a randomly selected azimuth into the seeded area, and randomly selected slickspots within it for sampling. Slickspots occurred in low lying areas with relatively smooth soil surfaces and lack of vegetation compared to the surrounding areas.

Once slickspots were located, their boundaries were determined (primarily by soil structure observations), and two 20-m perpendicular transects were established, with center points in the visually-determined center of the slickspot (Fig. 2). Endpoints of each transect were marked using a handheld GPS and the azimuths of each transect (looking toward the slickspot center) were recorded. Along each transect, starting at the 0 m mark, I measured *K. prostrata* cover using a gap intercept method. Gaps were areas with a minimum distance of 5

cm between *K. prostrata* plants, and I required that *K. prostrata* cover at least 5 contiguous cm if it was to be recorded (due to many small seedlings and a shortage of time). When transects extended beyond the slickspot, I recorded locations of slickspot edges in cm. After I sampled a slickspot, I walked a large circular pattern, attempting to locate equal numbers of slickspots in the seeded area and in the unseeded area, with a minimum distance of 50 m between slickspots sampled. Potential slickspots were noted, assigned numbers, and a random number was chosen to determine which were sampled, with the constraint in the unseeded area that only slickspots that supported *K. prostrata* were sampled. GPS waypoints of transect start and end points were uploaded into ArcGIS (ESRI 2009) and shapefiles of points were created for future reference. Ultimately, I sampled eight slickspots in the seeded area and ten in the unseeded area. Cover and total proportions of *K. prostrata* within slickspots and outside of slickspots were then calculated for the seeded and unseeded areas.
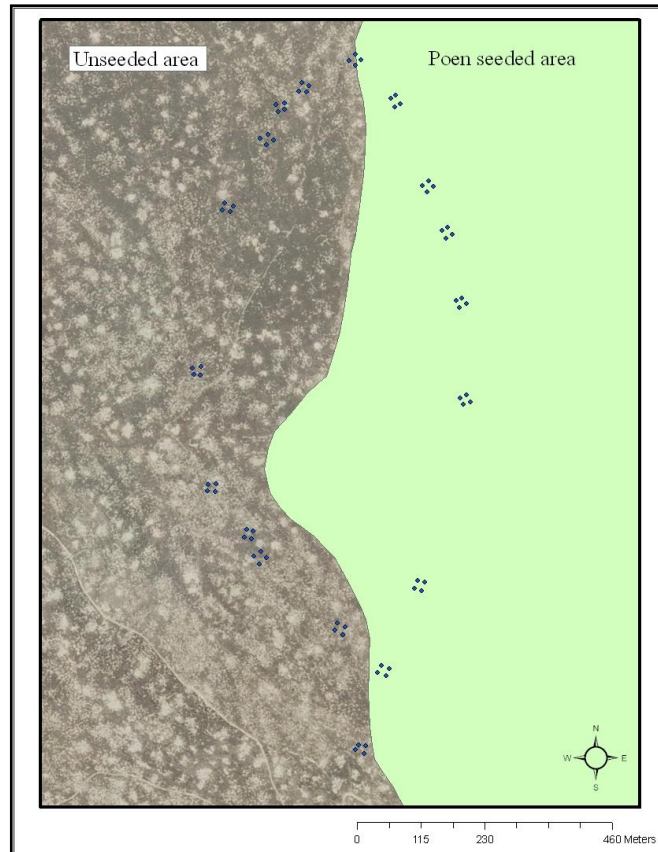
Figure 1. Sampling sites for rapid assessment of *K. prostrata* in slickspots in the Poen seeded area and adjacent unseeded areas. Groups of four points represent a single sampled slickspot, with a total of 18 sampled.
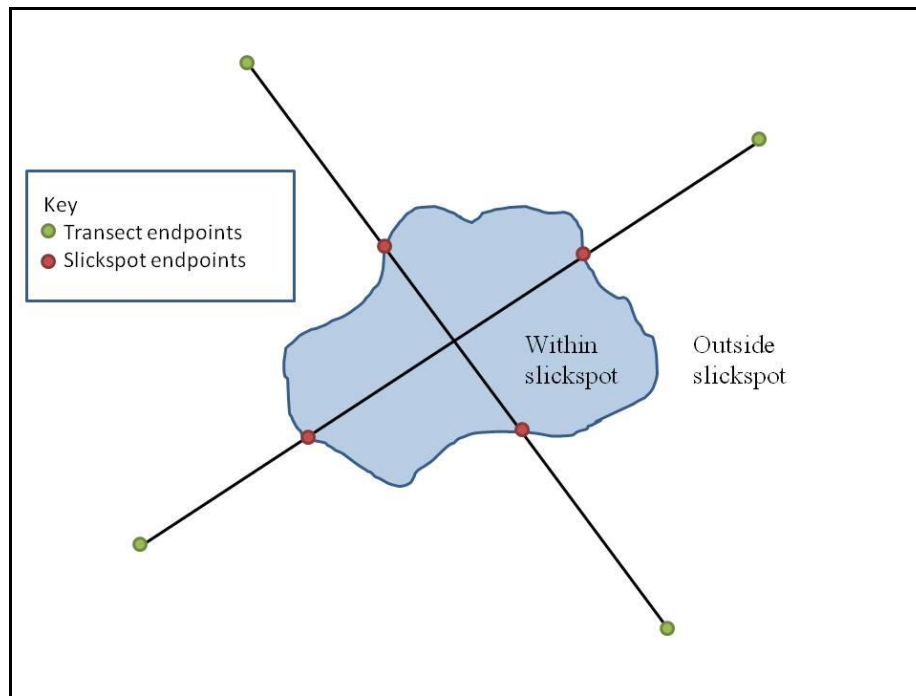
Figure 2. Schematic of slickspot sampling within the Poen seeded area and adjacent unseeded areas.

Results & Discussion

Total *K. prostrata* cover tended to be greater in the seeded than the unseeded area (19 % and 11%, respectively). In the seeded area, *K. prostrata* cover outside of slickspots tended to be greater than that within slickspot (12% and 7%, respectively; Fig. 3). In the unseeded area, however, *K. prostrata* cover outside of slickspots tended to be less than that within slickspots (3.7% and 7%, respectively). Thus, within slickspots, *K. prostrata* cover was approximately equal in seeded and in unseeded areas, however, outside of slickspots, its cover was approximately three times higher in seeded than in unseeded areas.
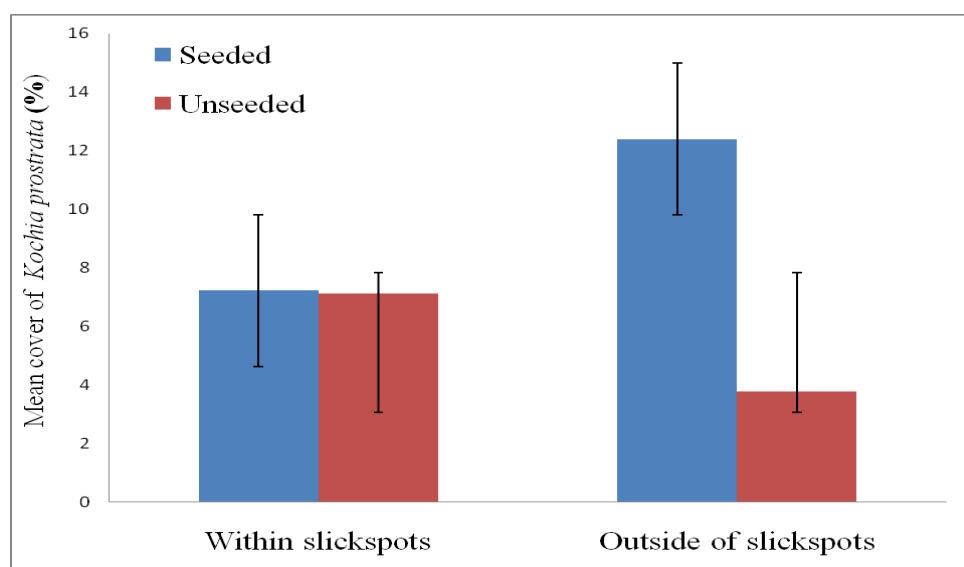
Figure 3. Comparison of mean *K. prostrata* cover (%) within slickspots and outside of slickspots. Bars indicate one standard error.

The fact that *K. prostrata* was present within both seeded and unseeded areas suggests that it had spread from the seeded into the unseeded area during the 17 yr since seeding. Though this site was both drill and aerially seeded, the seeding boundary was determined by a shapefile provided by the BLM and was confirmed in the field by changes in vegetation; *K. prostrata* occurrence in unseeded areas was likely not an artifact of mistakes in defining this boundary.

The observation that *K. prostrata* cover tended to be greater within slickspots than outside of them in unseeded areas may suggest that, when it spreads into unseeded areas, it is more likely to establish within slickspots. This apparently preferential recruitment in slickspots in the unseeded area may be at least partly a result of escape from competitors, as surrounding areas had much higher *B. tectorum* cover with patchy *Agropyron cristatum* abundance than did slickspots.

The greater cover of *K. prostrata* outside of slickspots than within them in the seeded area could be a result of disturbance caused by fire and seeding efforts, as *K. prostrata* establishes well in highly disturbed areas (Waldron et al. 2001). The seeded area had been burned multiple times, and fires tend to burn around slickspots due to lack of fuels within them. Also, slickspots are considered stressful environments, and, though *K. prostrata* is tolerant of salts, the apparently enhanced recruitment and establishment outside of slickspots

in seeded areas could be due to lessened stress outside of slickspots. The fact that *K. prostrata* cover within slickspots in both seeded and unseeded areas was approximately equal may suggest that *K. prostrata* populations in slickspots are eventually limited by competition or environmental factors that are independent of seeding.

This rapid assessment provides further evidence that *K. prostrata* actively recruits in slickspots in unseeded areas. This, coupled with its well-documented competitive interactions with other species (Waldron et al. 2001), suggest that it has potential to negatively impact slickspot-associated threatened species, such as *Lepidium papilliferum*. Though this assessment was a snapshot in time, it provides pertinent information for future studies and monitoring within the Poen Rehabilitation area. This rapid assessment was intended to provide quantitative data on patterns of *K. prostrata* abundance within and outside of slickspots, in a seeded and an unseeded area. Interpretations of patterns based on this study should be made with caution, given that it involved only one site. A future study involving a wider range of sites would enable wider inference and more robust understanding of *K. prostrata* dynamics in and around slickspots.

Bibliography

Harrison, R. D., N. J. Chatterton, B. L. Waldron, B.W. Davenport, A. J. Palazzo, W. H. Horton, and K. H. Asay. 2000. Forage kochia: Its compatibility and potential aggressiveness on intermountain rangelands. Utah Agricultural Experiment Station Research Report 162. Utah State University, Logan, Utah, USA. 66p.

Waldron, B. L., R. D. Harrison, N. J. Chatterton, B. W. Davenport. 2001. Forage kochia: Friend or foe. USDA Forest Service Proceedings RMRS-P-21. p. 210-215.

Appendix E: Survey of *Kochia prostrata* presence in playas near Simco Road, Mountain Home, Idaho.

      Kochia prostrata has been reported to invade playas located throughout the Snake River Plain in southwestern Idaho and is thought to negatively impact populations of *Lepidium davisii*, a rare species found in playas throughout the Mountain Home Desert (Tuason 2005). For example, one such playa has been invaded by *K. prostrata* from an adjacent seeding and a population of *Lepidium davisii* that occurred on this playa is thought to have been extirpated within 17 yr of the seeding potentially due to the presence of *K. prostrata* (Tuason 2005). On July 20, 2010, I surveyed seven playas at the southern end of Simco Road near Mountain Home, Idaho (43º5´N, 115º59´W; Fig. 1) that had not been seeded with *K. prostrata* for evidence of its spread from adjacent seeded areas. This survey was intended to increase understanding of the current state of *K. prostrata* presence within these sensitive areas, and its spread from adjacent seeded areas. The adjacent seeded areas were sampled in the primary investigation (Appendix A: Disc Chain, Disc Chain SS, and Simco Restoration), and these exhibited some of the greatest distances of spread of *K. prostrata* from the seeded boundary (maximum distance = 710 m). The playas surveyed were selected *a priori* using ArcMap and were located in low elevation, salt-desert shrub communities. At each playa I monumented locations of individuals and small populations of *K. prostrata* with a handheld GPS and recorded observations on characteristics of *K. prostrata* and the surrounding environment.
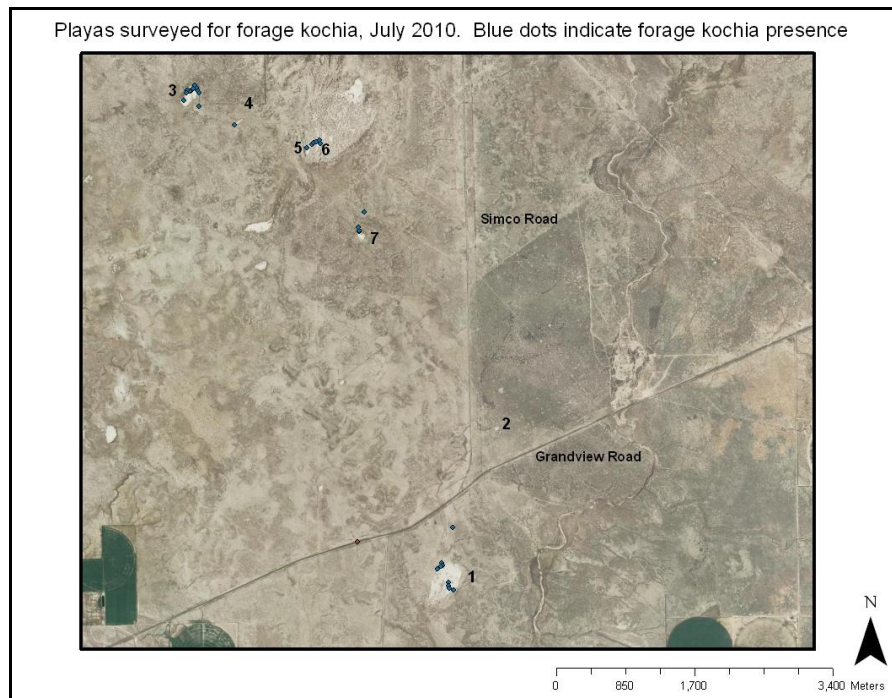
Figure 1. Playas visited in a rapid assessment for *K. prostrata* presence. Numbers represent playas visited, and blue dots represent *K. prostrata* individuals located within and in close proximity to playas.

Six of the seven playas did support *K. prostrata*. *Kochia prostrata* presence tended to be along the periphery of many of the playas, with recruitment often seeming to occur within the playas (Figs. 2-5). *Kochia prostrata* plants were also found along roadsides surrounding the playas, which also had not been seeded. Though *K. prostrata* within playas often occurred in satellite populations with multiple individuals, a few of the playas visited had healthy individual plants located great distances (estimated to be at least 200 m) from others, suggesting that seeds can travel at least that far, and that individual plants can persist when distant from source populations. The maximum recorded distance of a playa-inhabiting *K. prostrata* individual from the nearest seeding was 925 m, which was greater than any distance to farthest individual *K. prostrata* found in the primary analysis (710 m), though time since seeding is unknown. Waypoints and photos were saved for future use and monitoring, and are available from the author. This survey provides evidence that *K. prostrata* established within playas from adjacent rehabilitation seedings in the Mountain Home Desert, some at greater distances than previously known.

Figure 2. *Kochia prostrata* establishment along roadsides in an unseeded area in close proximity to playas.

Figure 3. *Kochia prostrata* recruitment in a playa near Mountain Home, ID. Note seedlings surrounding established individuals.

Figure 4. *Kochia prostrata* recruitment along the edge of a playa near Mountain Home, ID. This playa contains a population of *Lepidium davisii*, a rare endemic mustard.

Figure 5. A playa that previously contained a population of *L. davisii,* now colonized by *K. prostrata*.

Bibliography

Tuason, T. 2005. 1987-2004 Monitoring results of *Lepidium davisii* in the Mountain Home
District. USDI BLM Boise District, Boise, ID, USA.