Recovery and Rehabilitation of Vegetation on the Fitzner-Eberhardt Arid Lands Ecology Reserve, Hanford Reach National Monument, Following the 24 Command Fire

FINAL REPORT: 2001 - 2004

The Nature Conservancy

Recovery and Rehabilitation of Vegetation on the Fitzner-Eberhardt Arid Lands Ecology Reserve, Hanford Reach National Monument, Following the 24 Command Fire

Final Report: 2001 - 2004

Prepared by The Nature Conservancy of Washington for the U.S. Fish and Wildlife Service, Hanford Reach National Monument in partial fulfillment of Cooperative Agreement No. 13410.

February, 2005

James R. Evans

Marita P. Lih

The Nature Conservancy®
OF WASHINGTON
217 Pine St., Suite 1100

Seattle, WA 98106

CONTENTS

I. Short-Term Impacts of the 24 Command Fire on Vegetation of the Fitzner-Eberhardt Arid Lands Ecology Reserve.

II. Post-Fire Rehabilitation on the Arid Lands Ecology Reserve: Results of Year-One Monitoring

.

LIST OF TABLES

LIST OF TABLES (Continued)

LIST OF TABLES (Continued)

LIST OF TABLES (Continued)

LIST OF FIGURES

LIST OF FIGURES (Continued)

EXECUTIVE SUMMARY

I. Short-Term Impacts of the 24 Command Fire on Vegetation of the Arid Lands Ecology Reserve.

The 24 Command Fire burned across the Fitzner-Eberhardt Arid Lands Ecology (ALE) Reserve of the Hanford Reach National Monument and surrounding lands between June 27th and July 2nd, 2000. The effects of the wildfire on native and alien plant species and their communities on the ALE Reserve were monitored during spring and early summer (March-July) each year from 2001 to 2004. Monitoring efforts concentrated on the resampling of three different sets of permanent vegetation plots on the Reserve for which detailed pre-fire vegetation data were available. Additional methods, including two new series of plots, were added to sampling protocols to facilitate tracking the abundance and distribution of the alien annual cheatgrass (*Bromus tectorum*).

Invasive Species. After undergoing dramatic reductions immediately following the 24 Command Fire, measures of cheatgrass abundance matched or exceeded pre-fire levels in most of the habitats sampled by 2003 or earlier. Overall percent cover of cheatgrass in 2004 (10.3% ± 11.4 SD) was statistically similar to pre-fire cover (8.8% ± 15.4 SD; P = 0.363; Fig. S1). This pattern of dramatic decline followed by rapid recovery to levels of abundance equal to or greater than pre-fire values is a signature pattern of cheatgrass populations following wildfire. The only areas where cheatgrass cover still appeared to be reduced relative to pre-fire values lay in former big sagebrush stands on sandy soils. Overall percent frequency of cheatgrass in 2004 (68.1% ± 34.2 SD) was significantly greater than pre-fire frequency in most major habitat types, suggesting that the species was more evenly distributed within plots and was perhaps more ubiquitous on the landscape than prior to the fire (Fig. S2).

Stem density, the measure of plant stems per unit area, provides additional detail to the picture of cheatgrass responses to the 2000 fire. No pre-fire density data were available from vegetation plots to allow for the same comparisons made with percent cover and frequency. However, close correlations between cheatgrass cover and density in all other years suggest that low densities in 2001 represented a considerable reduction in this parameter compared to pre-fire conditions. Post-fire comparisons of densities between unburned refugia and adjacent burned areas support this conclusion. Densities in vegetation plots increased by 4-10 times between 2001 and 2002 and, after leveling off in 2003, increased again in 2004 (Table S3). This overall increase in density in 2004 was driven by dramatic increases in former big sagebrush and winterfat stands, which increased from 297.7 stems/m^2 (± 399.2 SD) to 547.9 stems/m^2 (± 582.0 SD) between 2003 and 2004 (P = 0.002), while density in other habitats remained fairly stable.

Cheatgrass abundance was not uniform across the ALE landscape. Cheatgrass was strikingly less abundant in vegetation plots at elevations above 1400 ft. (427 m). These sites support relatively productive native perennial plant communities that are considerably higher in total percent cover and species richness of native perennials than habitats at lower elevations. Higher precipitation and more moderate growing season temperatures in these habitats may promote greater resilience in these more nearly closed stands in the aftermath of wildfires and other disturbances. The high percent cover of native perennial vegetation was correlated with reduced cover of cheatgrass and may indicate a degree of competitive control of cheatgrass abundance.

Characteristics of the winter environment at higher elevations may also contribute to reduced performance by cheatgrass. A critical advantage of this invasive winter annual is its capacity for root growth during the cool months of winter when soil moisture is less limiting and native perennial species are dormant. The progressively lower temperatures and lengthening periods of persistent snow cover that are associated with increasing elevation probably reduce this competitive advantage. Over-winter mortality of fall-germinating seedlings, reduced productivity of survivors, and delayed germination of spring-germinating seedling cohorts may all be significant consequences of winter conditions

At lower elevations, cheatgrass abundance remained lower than pre-fire values in 2004 where big sagebrush stands occurred on sandy soils. Cheatgrass cover was still only 54 % of pre-fire values in these habitats four years after the 24 Command Fire, in contrast to other sites where cheatgrass cover had returned to pre-fire values much earlier. The coarse texture of sandy soils allows moisture from winter storms to penetrate quickly through surface layers. This characteristic renders these sites less favorable for winter growth of cheatgrass seedlings than silt loam soils, which hold moisture nearer to the soil surface where it is more available to seedling roots. While recolonization is slower on these sites, high cheatgrass abundance in pre-fire data underscores this invader's ability to dominate such sites over the long term.

The invasion of cheatgrass into shrub-steppe sites initiates changes in critical ecosystem properties such as community structure, species diversity, and moisture and nutrient regimes. Cheatgrass exploits the niches previously occupied by native annual plant species, outcompetes the seedlings of perennials, smothers microbiotic crusts, and disrupts mycorrhizal associations. Its winter annual habit alters seasonal patterns of production and adds uncharacteristically large amounts of dead above- and below-ground biomass to the ecosystem annually. This copious addition of litter contributes to smothering effects and precipitates changes in soil chemistry and in the composition and diversity of soil invertebrate and microbial communities. Areas dominated by cheatgrass are associated with declining habitat value for mammals and birds. The annual buildup of a continuous mat of dry litter contributes to increases in the frequency, extent, and severity of wildfires, which reinforces the trends outlined above. Wildfires contribute to the wholesale alteration of the ecosystem and increasing dominance of cheatgrass by altering the availability and distribution of nutrients and by further reducing the vigor and continuity of native vascular plant and microbiotic crust communities.

The interaction of cheatgrass and wildfire in the arid west resulted in the conversion of millions of acres of shrub-steppe habitats to alien annual grasslands within the last century. Once converted, these habitats have persisted as annual grasslands for a half-century or longer without exhibiting any trend towards recolonization by native species or recovery to a condition of dominance by native perennials.

Alien annual forbs are also common and widespread across all major habitat types on the ALE Reserve. The abundance and distribution of alien annual forbs increased substantially following the 24 Command Fire. While overall cover and frequency of alien annual forbs were highest in 2003, abundance values for tumble mustard (*Sisymbrium altissimum*), Russian thistle (*Salsola kali*), and storksbill (*Erodium cicutarium*) in 2004 were still significantly greater than pre-fire values, and the number of plots in which these species were recorded represented a 4- to 6-fold increase over pre-fire distributions. These species do not of themselves threaten ecosystem integrity in the way that cheatgrass does, but they are frequently cited as paving the way for the proliferation of cheatgrass in sequences of plant succession following wildfire.

Native plants and communities. The 24 Command Fire had substantial impacts at all structural levels within stands of Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) on ALE. Changes in stand structure, species abundance, and community composition were strongly evident four years after the wildfire, and the impacts of these changes will affect ecosystem processes and habitat quality for many years to come.

Both Wyoming big sagebrush and spiny hopsage (*Atriplex* [= *Grayia*] *spinosa*) were nearly entirely removed from within the footprint of the 24 Command Fire (Fig. S4), and the portions of stands that escaped the fire are small and fragmented. The 24 Command Fire consumed most of the last remaining high-quality, large, and contiguous stands of big sagebrush within the ALE Reserve, continuing a trend of the past three to five decades that has seen large-scale wildfires consume nearly all of the shrublands that once dominated the Reserve. The loss of these keystone shrub species over wide areas on the ALE Reserve is exacerbated by the lack of significant natural sources of sagebrush seed over most of the affected area, and by the vanishingly small rate of reproductive success of spiny hopsage in the Columbia Basin in recent decades.

The removal of large deep-rooted shrubs from shrub-steppe habitats alters important ecosystem functions such as water utilization and storage, biological production, and nutrient cycling, and reduces habitat value for large and small mammals, birds, and terrestrial invertebrates. Sagebrush-obligate wildlife species such as the greater sage grouse (*Centrocercus urophasianus*), sage sparrow (*Amphispeza belli*), loggerhead shrike (*Lanius ludovicianus*), and other species require big sagebrush habitat for shelter, nest sites, food, or habitat for prey species, while many other shrub-steppe species are most common and abundant in the vicinity of expansive big sagebrush habitats.

These former Wyoming big sagebrush or big sagebrush-spiny hopsage habitats on the ALE Reserve will not recover without extensive and persistent restoration efforts. The same factors that make unassisted recovery unlikely – the semiarid climate of south central Washington, the high frequency of wildfires, and the potential for increase of cheatgrass and other invasive species – will present extreme challenges to the restoration and long-term maintenance of these critical habitats.

Impacts of the 24 Command Fire within shrublands on ALE were not limited to the canopy layer. All other structural layers of the big sagebrush plant community – hemishrubs, grasses, forbs, and microbiotic crusts – exhibited evidence of decline and mortality beyond that experienced in other community types. Recovery of native perennial vegetation in former big sagebrush shrublands was still far below pre-fire levels in 2004. In a few places perennial forbs such as Carey's balsamroot (*Balsamorhiza careyana*) and Cusick's sunflower (*Helianthus cusickii*) became abundant during the springs of 2003 and 2004. In most cases, however, increases in the abundance of perennial forbs and hemishrubs amounted to only a few percent in absolute cover and failed to compensate for large declines in perennial grasses and the complete absence of dominant shrubs.

In the absence of significant cover by perennial plant species, tumble mustard and other disturbance-oriented annual forbs have colonized many of these sites, although cover remains sparse in many areas. Cheatgrass cover and frequency have increased annually since 2001 in these burned-over sagebrush stands, and density also increased in 2004, underscoring serious concerns about the trajectory of ecological succession on these habitats.

Like the loss of keystone shrubs and understory perennials, substantial loss of topsoil through erosion would represent the crossing of another ecosystem threshold from which recovery could be effected only at great expense and over a very long period of time. The potential for soil erosion on these heavily impacted sites has apparently been moderated to some degree by the establishment of annual vegetation and the slow recovery of perennials since 2000. Very few observations of dust storms, dust devils, sand over roadways, and other evidence of mass movement of soil particles were made in 2003 and 2004 compared to the two years immediately following the fire. However, the potential for long-term degradation and erosion of surface soils will remain so long as perennial vegetation and microbiotic crust do not reclaim the heavily impacted former big sagebrush stands.

Threetip sagebrush (*Artemisia tripartita*) shrublands did not exhibit the same degree of impact upon the associated herbaceous understory, except within limited areas. Fuel loads likely were lower in these stands than in the big sagebrush communities. Since these stands occur in more mesic habitats, at higher elevations and on northerly aspects, community resilience is favored in these sites compared to lower elevation shrublands. Threetip sagebrush stands may recover structural characteristics more quickly than big sagebrush stands, since *A. tripartita* is capable of resprouting from crowns following wildfire. Nevertheless, canopy cover in threetip sagebrush stands were still significantly below pre-fire levels in 2004 (Fig. S4), and Wyoming big sagebrush, historically a common associate of threetip sagebrush in these habitats, was almost entirely missing from these habitats.

Stands of winterfat (*Eurotia lanata*) in our samples suffered dramatic declines in the shrub canopy following the 24 Command Fire (Fig. S4). As in the big sagebrush habitats, shrub mortality was associated with high mortality of perennial grasses and reduced percent cover of perennial grasses and forbs in the understory. Percent cover and density of cheatgrass in winterfat stands on the ALE Reserve have increased substantially since 2001. Unlike threetip sagebrush stands, winterfat communities occur at lower elevations where the warmer, drier conditions are less favorable for the recovery of perennial vegetation. Winterfat communities are in decline in the Great Basin where winterfat reproduction appears to be threatened by competition from invasive annual plant species. Population trajectories for this species in south central Washington are not well known, but the characteristic winterfat/ Sandberg's bluegrass community is uncommon in the lower Columbia Basin, occupying less than 1100 acres on the ALE Reserve prior to the 24 Command Fire.

Bluebunch wheatgrass (*Agropyron spicatum*) is the dominant large bunchgrass over most of the middle and upper slopes of ALE. Overall cover of bluebunch wheatgrass has increased substantially since 2001 (Fig. S5), but in 2004 was still significantly below pre-fire levels. Successive wildfires at short intervals will gradually reduce the vigor of bluebunch wheatgrass and other native bunchgrasses, especially when associated with increased competition from cheatgrass and other invasive species. Cheatgrass seedlings outcompete the seedlings of native bunchgrasses, preempting the establishment of new cohorts that would replace weakened or senescent individuals.

Needle-and-thread (*Stipa comata*) and Idaho fescue (*Festuca idahoensis*) tend to be more sensitive to fire damage than bluebunch wheatgrass, and both species suffered more severe losses of above-ground cover, exhibiting only limited recovery by 2004 (Fig. S5). Sandberg's bluegrass (*Poa sandbergii*) is relatively tolerant of wildfire and was the least affected of the native bunchgrasses. This cool season perennial sustained only moderate losses following the 24 Command Fire, and had recovered to pre-fire levels by 2002. However, where fire severity was high, as in stands of big sagebrush or winterfat, even Sandberg's bluegrass exhibited mortality and post-fire declines in abundance.

The condition of the bluebunch wheatgrass grasslands varied across the Reserve, with habitats above 1000 – 1200 ft. (305 -365 m) tending to be in better condition than those below this range. The herbaceous component in many mid-elevation stands appeared to be headed towards recovery in 2004. The best of these stands may approach the condition they were in immediately before the 24 Command Fire within the next few years. Stands below 1000 ft. were in the poorest condition in terms of cheatgrass abundance and the virtual absence of microbiotic crusts. The prospect of recurring catastrophic wildfires and resultant stepwise increases in cheatgrass abundance prompt concerns about ecosystem integrity, despite the recovery of perennial bunchgrasses. From a long-term perspective, key elements of the shrub-steppe ecosystem, such as large shrubs and microbiotic crusts, are still missing or in poor condition in all of these habitats and will require extensive and persistent restoration efforts if these habitats are to again approach historical levels of productivity and wildlife habitat value.

Owing to increased moisture, lower evaporative demand, and fewer woody fuels, herbaceous plant communities at elevations greater than 1400 ft. (427 m) elevation appeared to be more resilient than those at lower elevations. This effect was especially apparent above 2000 ft (610 m). Invasive species, although still present, also occupy a smaller proportion of the landscape at higher elevations. These two factors taken together suggest that higher elevation communities have the best chance of recovery to the conditions that pertained prior to the 24 Command Fire. However, the absence of significant stands of big sagebrush or threetip sagebrush in these habitats indicates unfilled roles in the ecological community, which signal lower biological production and reduced wildlife habitat value in these sites compared to historical conditions and site potential.

Microbiotic soil crusts (MBC) are extremely sensitive to wildfire and percent cover of MBC on ALE was greatly reduced in all habitat types following the 24 Command Fire (Fig. S6). The most dramatic reductions in crust abundance occurred in shrublands, where MBC was almost entirely eliminated. Changes in the abundance of MBC following wildfire are accompanied by changes in crust community attributes such as crust thickness, biomass, species composition, and species diversity. Degradation of MBC adversely affects its critical ecological roles in soil stabilization, resistance to biological invasion, and other functions. The assessment of crust recovery based solely on visual surface cover, as in this study, fails to account for these and other critical attributes, and may greatly underestimate recovery time.

The condition of microbiotic crusts across the ALE Reserve is generally poor. Even areas that currently support extensive cover of MBC are characterized primarily by rudimentary crust assemblages in early stages of recovery from disturbance. Biological crusts exhibiting the dark coloration, species diversity, and complex microtopographic relief characteristic of mature crusts are found only in a few unburned refugia.

Recommendations. Assessment of post-fire patterns of cheatgrass abundance as outlined in this study can help to identify top priority areas where restoration will be most effective and is most urgently needed. While no habitats on the ALE Reserve appear to be immune to cheatgrass invasion, environmental and plant community characteristics may confer a degree of resistance upon some types of habitats. Following wildfires in big sagebrush stands, a brief window of opportunity may be available for restoring native perennial plant species with minimal competition from cheatgrass. Restoration efforts that take advantage of this opportunity can minimize site preparation expenses dedicated for invasive species control, while at the same time minimizing risks of herbicide effects on desirable remnant native species. Following the 24 Command Fire, this opportunity appeared to last only for a single season on finer-grained silt-loam soils. Where sagebrush stands were cleared on sandy soils, however, restoration opportunities appeared to exist even four years following wildfire.

A truly accurate portrayal of abundance and condition of microbiotic crusts on the ALE Reserve will require the work of specialized crust ecologists. No technology exists for the restoration of microbiotic crusts at a landscape scale. The few remaining areas of undisturbed crusts on ALE and across the Monument represent irreplaceable reference stands and refugia of biodiversity. Measures for the protection of selected areas of high quality biological crusts should be explicit in fire management plans and in other resource protection plans.

Monument managers must have timely information regarding habitat condition and the status of invasive species populations in order to adaptively manage at-risk habitats. Additional management intervention is likely to be required in former sagebrush stands between the Gate 117 and Gate 118 roads, and in winterfat stands on the lower slopes above the Cold Creek Valley. In these areas continued monitoring of a portion of the network of permanent plots used in this study is strongly recommended. Continued monitoring of cheatgrass abundance and the status of native perennial plant communities is necessary to provide timely indication of habitat condition and to allow managers to evaluate levels of threat to resources and to respond appropriately.

The Hanford Reach National Monument is fortunate to have a system of reference vegetation plots against which to measure ecosystem change. The analyses contained within this report would not have been possible without the existence of a network of permanent vegetation plots on the ALE Reserve and the availability of relevant pre-fire data from those plots.

A portion of this network -- permanent plots established under the Hanford Site Biological Resources Management Plan (BRMaP) -- extends across the Monument, on lands currently managed by the Department of Energy (DOE) as well as those managed by USFWS. Periodic resampling of these plots will help management to evaluate trends in habitat condition and invasive species populations, as well as to update baseline information in advance of potential wildfires or other disturbances, or climatically induced vegetation change. It is strongly recommended that the USFWS and DOE, in concert as co-managers of the Hanford Reach National Monument, make certain that these plots are located and resampled within the next one to two years, and that resampling of these plots thereafter become part of a periodic effort. Installation of additional plots may be necessary to ensure adequate coverage of the Monument's critical resources.

Effective monitoring programs are essential to the adaptive management of natural resources and contribute to budgetary efficiencies when threats to resources are identified at an early stage. A pool of capable volunteers, including individuals who made strong contributions to the data collection efforts recounted in this volume, is available in the Columbia Basin to assist in monitoring endeavors.

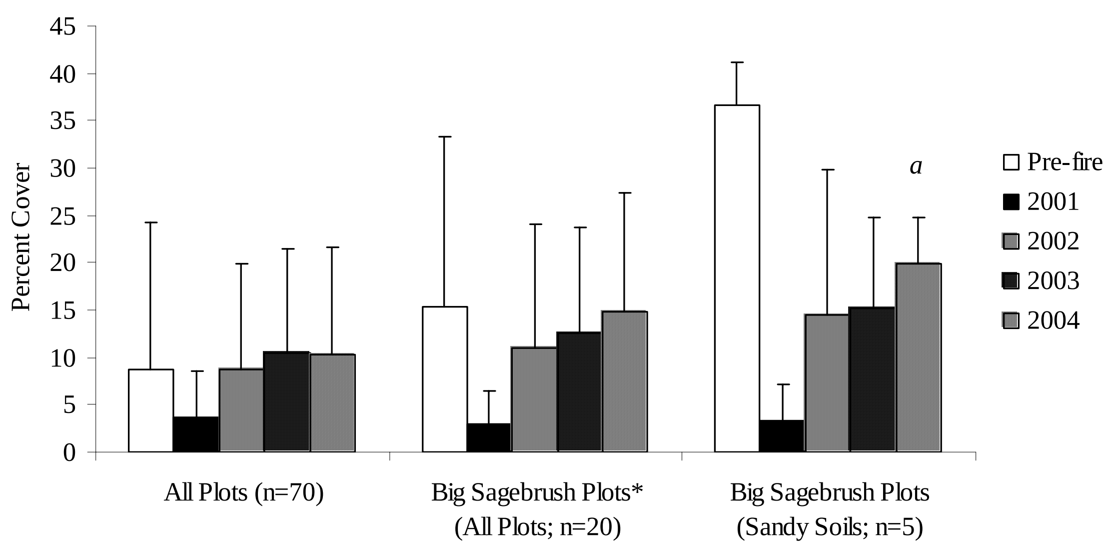


Fig. S1. Changes in percent cover of cheatgrass (*Bromus tectorum*) on the ALE Reserve following the 24 Command Fire. Bars = 1 standard deviation. 2004 values with accompanying script letter are statistically greater than pre-fire values ($P < 0.005$).
* Category includes winterfat plots (n = 3).

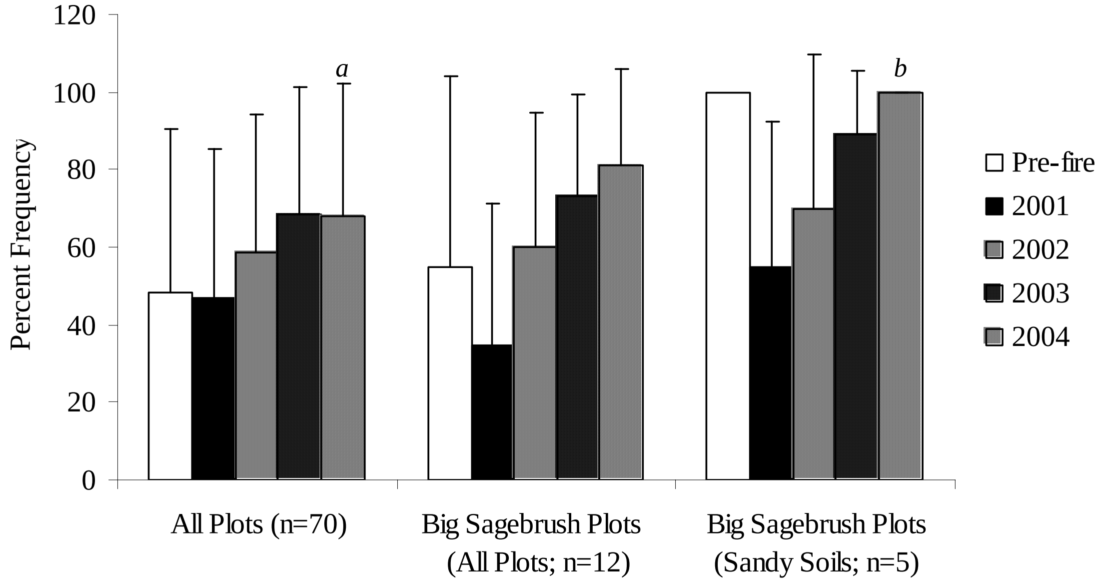


Fig. S2. Changes in percent frequency of cheatgrass (*Bromus tectorum*) on the ALE Reserve following the 24 Command Fire. Bars = 1 standard deviation. 2004 values with accompanying script letter are statistically greater than pre-fire values: *a* =$P < 0.001$; *b* = $P < 0.01$.

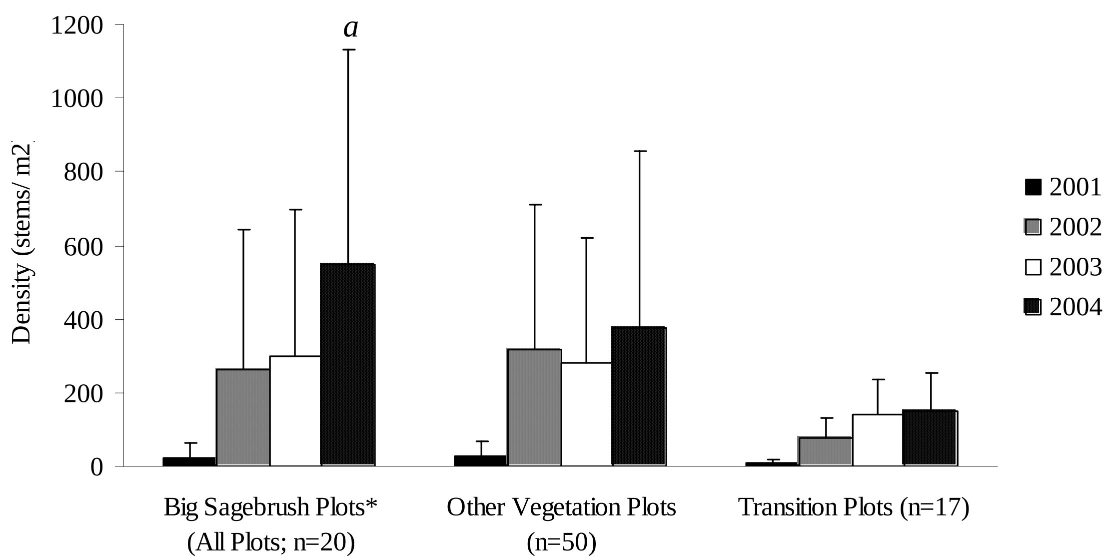


Fig. S3. Changes in density of cheatgrass (*Bromus tectorum*) on the ALE Reserve following the 24 Command Fire. Bars = 1 standard deviation. 2004 values accompanied by a script letter are significantly greater than 2003 values ($P < 0.005$).
* Category includes winterfat plots (n = 3).

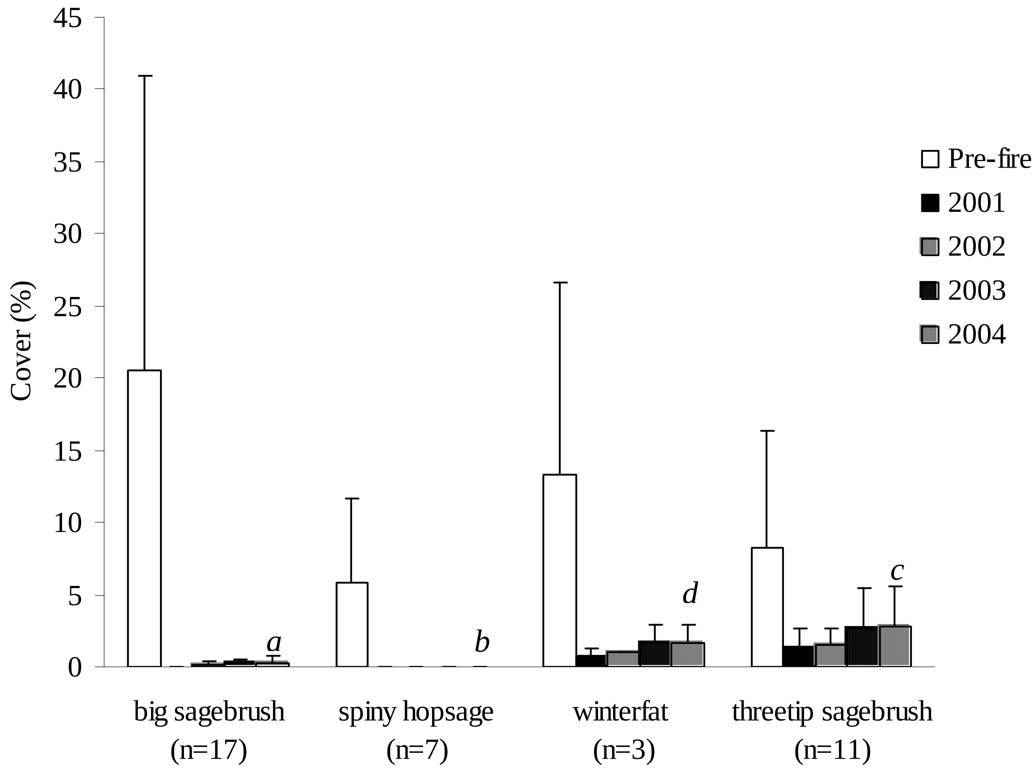


Fig. S4. Changes in percent cover of important large shrubs on the ALE Reserve following the 24 Command Fire: Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*); spiny hopsage (*Atriplex* [= *Grayia*] *spinosa*); winterfat (*Eurotia lanata*); threetip sagebrush (*Artemisia tripartita*). Bars = 1 standard deviation. 2004 values with accompanying script letter are statistically lower than pre-fire values: $a = P < 0.0001$; $b = P < 0.005$; $c = P < 0.05$; $d = P < 0.10$.

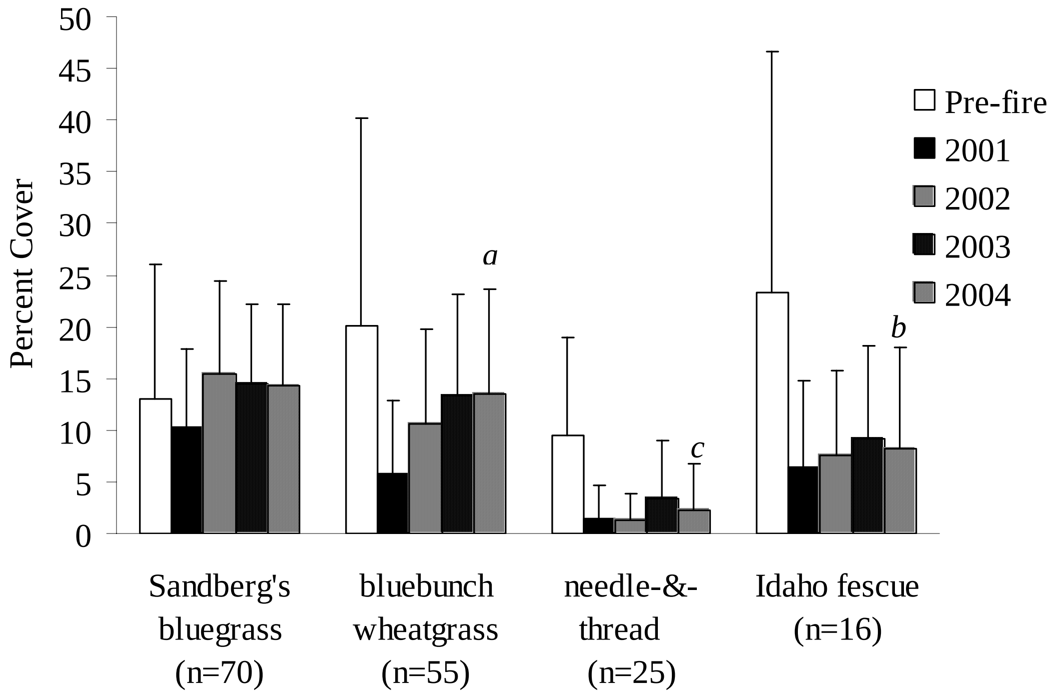


Fig. S5. Changes in percent cover of selected native perennial bunchgrasses on the ALE Reserve following the 24 Command Fire: Sandberg’s bluegrass (*Poa sandbergii*), bluebunch wheatgrass (*Agropyron spicatum*), needle-and-thread (*Stipa comata* and *S. thurberiana*), and Idaho fescue (*Festuca idahoensis*). Bars = 1 standard deviation. 2004 values with accompanying script letter are statistically lower than pre-fire values: $a = P < 0.0001$; $b = P < 0.005$; $c = P < 0.05$.

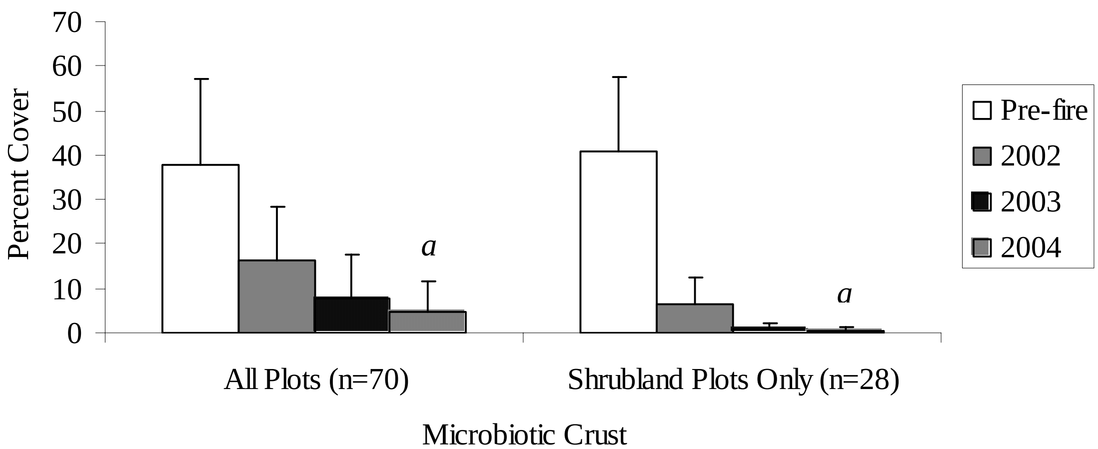


Fig. S6. Changes in percent cover of microbiotic soil crust on the ALE Reserve following the 24 Command Fire. 2001 data are suspect and are not shown. Bars = 1 standard deviation. 2004 values with accompanying script letter are statistically lower than pre-fire values: $a = P < 0.0001$.

II. Monitoring Post-fire Restoration on the Fitzner-Eberhardt Arid Lands Ecology Reserve, Hanford Reach National Monument.

Rehabilitation efforts implemented between December 2002 and February 2003 targeted 10,000 acres of the native shrub-steppe habitats on the Arid Lands Ecology (ALE) Reserve that had been most seriously affected as a result of the 24 Command Fire. Rehabilitation measures included outplanting of nursery-grown big sagebrush seedlings, herbicide treatments for the control of cheatgrass, and aerial and drill seeding of native grasses and cultivars. Monitoring of rehabilitation measures began in November 2002 and continued through fall 2004. The monitoring team made use of long-term permanent vegetation plots located throughout the rehabilitation area and installed new permanent plots as needed to assess survival of outplantings, seed density, emergence and recruitment of grass seedlings, and the effectiveness of cheatgrass control measures. The monitoring team also assessed the condition of a fire suppression swath in Upper Snively Basin that was regraded and seeded during spring 2001.

Cheatgrass abundance and effects of herbicide treatments. Effects of herbicide treatments and native seedings on cheatgrass abundance within the rehabilitation project area were slight, if any. Herbicide treatments (performed during November 2002 and again in February 2003) may have dampened increases in cheatgrass abundance somewhat, as percent cover of cheatgrass within the project area in 2003 (10.1% ± 9.9 SD) remained statistically similar to pre-treatment levels, while cover in untreated areas exhibited a significant increase of more than 2.0% during the same period ($P = 0.009$; Fig. S7). Nevertheless, cheatgrass cover and density over most of the project area (with the exception of elevations over 800 ft. in Polygon 3) were still sufficiently high to interfere with native seedlings (Fig. S8) and likely contributed to reduced emergence and recruitment at lower elevations. Glyphosate (Roundup™) is a post-emergence herbicide and has no effect on plants that germinate after treatment is applied. Periods of above-normal precipitation during the winter and spring of 2003 favored opportunistic germination of cheatgrass after herbicide treatments had been completed, and the resulting growth very likely swamped potential treatment effects.

Both cover and density increased significantly within the project area between 2003 and 2004 (Figs. S7, S8).

Big sagebrush outplantings. The overall initial planting density of nursery-grown Wyoming big sagebrush seedlings (410.2 plants/ acre ± 128.9 SD) was somewhat below project specifications (450 plants/ acre). Planting treatments consisting of bare root seedlings dipped in a hydrogel inoculated with mycorrhizal fungi (M+), bare root seedlings dipped in a hydrogel without mycorrhizal fungi (M-), and seedlings grown in 4 in.^{3} nursery tubes (without mycorrhizae) were installed in separate polygons. Overall survival of big sagebrush outplantings over two years was 36.3% (± 26.0 SD). Based on these samples, more than 254,000 of a total of more than 700,000 sagebrush outplantings have been established on ALE. Survival varied widely among polygons, ranging from as low as 4.8% (± 6.6 SD) to as high as 76.6% (± 10.5 SD; Fig. S9a). Polygons planted with M- seedlings exhibited significantly greater survival (71.6% ± 8.8% SD) than M+ seedlings (13.5% ± 11.2% SD) or seedlings grown in 4 in.^{3} nursery tubes (44.2% ± 15.5% SD; Fig. S9b).

Mycorrhizal inoculation has been shown to enhance shoot growth and increase the tolerance of Wyoming big sagebrush seedlings to soil moisture stress. It is extremely unlikely that the presence of mycorrhizae was deleterious to seedling establishment and survival on the ALE Reserve. An hypothesis regarding the method of inoculation and sagebrush root anoxia is discussed. Controlled experiments are recommended to clearly evaluate the poor performance of mycorrhizal treatments observed in this study. Pending the results of these experiments, we recommend against the use of Plant Success™ mycorrhizal hydrogel in sagebrush plantings on silt loam soils on the Hanford Reach National Monument. The results of this and other recent plantings on the ALE Reserve suggest that mycorrhizae are not required for successful establishment of Wyoming big sagebrush outplantings, at least during years of above-normal precipitation as experienced in 2003 and 2004; however, mycorrhizal inoculation via alternative methods (e.g., inoculation of growing media at the nursery, or the use of dry tablets during outplanting) is still recommended.

Most surviving sagebrush outplantings appeared to be vigorous and well-established in October 2004 and site trajectories towards the development of mature shrub canopies in these areas appear promising. Despite low survival rates in several polygons, the overall success of sagebrush outplantings resulting from these and other local efforts over the past several years provide an encouraging sign that this component of the ALE ecosystem may be restorable in this manner.

Attempts to establish big sagebrush from seed along the west side of the 1200 Ft. Rd. appeared to be unsuccessful. While above-normal precipitation during the winter and early spring of 2003 favored the emergence of a number of sagebrush seedlings, prolonged drought from late spring through fall 2003, along with competition from established native perennial grasses and invasive annuals, likely exceeded the stress tolerance of vulnerable new seedlings.

Seed applications. Aerial seeding densities exceeded the project specifications of 538.2 seeds/m^2 (50 seeds/ ft.^{2}) by 2.6 to 6.6 times in all polygons. High seeding rates may be necessary in order to achieve adequate stocking densities for habitat stabilization and recovery, as rates of seedling recruitment observed in this and other studies suggest. Seed rate specifications based on density may not be relevant when application of large quantities of seed by weight is involved, especially when mixes include large quantities of small, light seeds such as those of Sandberg's bluegrass.

The overall emergence rate (2003 seedlings/m^2 ÷ seeds/m^2) for aerially seeded grasses and forbs in this study was 3.2% (± 2.4 SD) of seed density, while the potential seedling recruitment rate (2004 seedlings/m^2 ÷ seeds/m^2) was 0.7% (± 0.4 SD). Recruitment rates are inflated by an unknown factor due to seedling emergence during fall 2003 and/or spring 2004. Recruitment was highest at the upper elevations within the project area (800 – 1000 ft./ 245 – 305 m), but differences between this and other portions of the site were not statistically significant.

Potential recruitment densities for wheatgrass species (bluebunch wheatgrass [*Agropyron spicatum*] in the high elevation seed mix, and thickspike wheatgrass [*A. dasystachyum*] in the low elevation mix) ranged between 0.5 seedlings/m^2 and 2.1 seedlings/m^2 in aerially seeded polygons (Fig. S10). Potential recruitment densities for Sandberg's bluegrass (*Poa sandbergii*) ranged between 6.0 seedlings/m^2 and 13.7 seedlings/m^2. Recruitment densities for yarrow (*Achillea millefolium*) were < 0.6 seedlings/m^2, while densities of needle-and-thread (*Stipa comata*), Indian ricegrass (*Oryzopsis hymenoides*), and squirreltail (*Sitanion hystrix*) were negligible.

Data on densities of native bunchgrasses on seeded ranges or in natural habitats are scarce in the ecological literature. Based upon one of the few rating systems available for quantitatively evaluating seeding success, overall recruitment stocking rates achieved by the BAER project on ALE (16.3 seedlings/m^2 ± 28.5 SD) may be rated as excellent, primarily due to high densities of Sandberg's bluegrass.

Overall potential recruitment densities reported in this study (16.3 seedlings/m^2) were higher than those observed for large ($\geq$ 5.0 cm diameter) mature bunchgrasses (3.9 tussocks/m^2) in post-fire bunchgrass mortality plots surveyed in 2001 in comparable sites on the ALE Reserve. Potential recruitment densities for wheatgrass seedlings in three of four rehabilitation treatment types were very similar to density values for bluebunch wheatgrass in mortality plots (2.1 tussocks/ m^2 ± 1.8 SD). Only the low elevation seed mix over silt loam soils (the treatment which covered the largest area) exhibited densities that were substantially lower than these values. These survival plot densities are not directly comparable to our recruitment densities for several reasons, but provide an indication of desirable ultimate densities for site stabilization. It remains to be seen whether the small, recently established seedlings in rehabilitation areas will continue to survive and expand to the point where the desired habitat stabilization has been achieved.

The rates of grass seedling emergence and recruitment from aerial seeding efforts observed in this study are probably typical of broadcast seeding efforts in the arid West. Under natural conditions, soil moisture availability limits seed germination, as well as seedling emergence and early seedling growth. The higher emergence rates observed at higher elevations in the project area may reflect somewhat more favorable moisture conditions for seedling emergence and early survival in these habitats. Biological factors such as seed predation and grazing by small mammals and birds, along with seed dormancy mechanisms and both inter- and intraspecific competition among plants may also affect the outcome of wildland seeding efforts. Extended seed dormancy and the redistribution of seeds by rodents are both characteristic of Indian ricegrass (*Oryzopsis hymenoides*) and may account for the near absence of this species from emergence and recruitment samples.

Potential recruitment densities of seeded species in treatment plots did not differ statistically from seedling densities in control plots, suggesting that aerial seeding had no effect on seedling density. However, control plots were outside the project area and had higher densities of established perennials as a source of seed than did the treatment plots. This factor could have been especially significant with regard to wheatgrass species, which were entirely absent below approximately 800 ft. (244m) elevation within the project area prior to seeding.

Drill seed treatment. While drill seeding is generally accepted as a much more reliable method of achieving seeding success, there were no significant differences in either emergence or recruitment in the drill seed treatment compared to aerially seeded areas on ALE. Numerical differences in performance between the three wheatgrass stocks (Anatone, Secar, and Hells Canyon) installed by drill seeding were also not significant. Low recruitment rates for all three wheatgrass stocks probably reflect the serious obstacles to seedling establishment in the extreme environment of this portion of the Columbia Basin. Drill seeding was carried out in the warmest, most arid portion of the ALE Reserve where annual precipitation averages only 6.25 inches (160 mm)/ year, well below the recommended limits for all three wheatgrass stocks.

Recovery of bulldozer fireline. Three years after rehabilitation efforts were implemented, the fire line swath in Upper Snively Basin was still significantly different from the surrounding vegetation. Percent cover of native vegetation was considerably less than in the relatively undisturbed vegetation surrounding the impacted area. Microbiotic crusts were absent from the affected area, and cover of native perennial bunchgrasses was greatly reduced, while cover and density of cheatgrass was increased (Fig. S11). Native vegetation is unlikely to recover fully within the disturbed area so long as cheatgrass and other invasive species are present in large numbers. If left untreated, cheatgrass will preempt the recovery of native vegetation and can be expected to increase within the suppression area over time. The presence of such an inoculum of invasive species within an otherwise very high-quality native plant community raises concern over the potential spread of invasives from the disturbed swath farther into the surrounding habitat.

Recommendations. Outplanting of nursery-grown stock is more reliable than direct seeding as a method for restoring shrubs to the landscape, and it is highly recommended that this practice be continued on ALE beyond the limits of the BAER program. There are tens of thousands of acres on the Reserve that are legitimate candidates for the reintroduction of Wyoming big sagebrush, along with threetip sagebrush and spiny hopsage in appropriate habitats. Plantations at higher elevations will enjoy the benefits of higher precipitation and more moderate growing season temperatures that should facilitate establishment of outplantings during average years.

One additional year of monitoring the existing array of sagebrush survival plots in 2005 is strongly recommended in order to document that the trends described in this study are firmly established. After 2005, periodic monitoring at five-year intervals will provide valuable information with which to calibrate performance objectives for future shrub introduction projects. In order to assess the impacts of restoration on habitat quality, monitoring of invertebrate populations and wildlife use of these developing sagebrush stands is also highly recommended.

Additional management intervention will be required within the BAER rehabilitation project area in order to stabilize soils, suppress invasive species, and promote recovery of these critical wildlife habitats. Continued monitoring of selected areas, such as the former sagebrush stands between the Gate 117 and Gate 118 roads and elsewhere, will assist resource managers in developing plans for further treatment of areas at risk (see Section I, above).

Future efforts to establish and maintain a greenstrip along the SR 240 corridor should carefully assess the condition of perennial vegetation along the corridor. Portions of this corridor, such as the areas between mileposts 6 and 9 and around mileposts 18 and 19, currently support fair to good quality native perennial vegetation. While neither pristine nor weed-free, these areas are reservoirs of low elevation native plant biodiversity and probably function as well as the target greenstrip vegetation in terms of firebreak effectiveness. These areas should not be disturbed unnecessarily but should be monitored and maintained as diverse natural plant communities. Excluding them from disruptive interventions will allow management resources to be focused on establishing perennials along portions of this corridor where they are presently largely absent.

Within the suppression area in upper Snively Basin, effective control of cheatgrass for at least 1-2 years will be necessary to allow native vegetation to recover, whether or not the area is enhanced with additional native seedings or plantings. In concert with invasive species control, the successful reintroduction of important structural native grasses such as bluebunch wheatgrass and Sandberg's bluegrass will greatly increase the likelihood of recovery of the disturbed area. Suggestions regarding invasive species control and assisted recovery of this area are discussed.

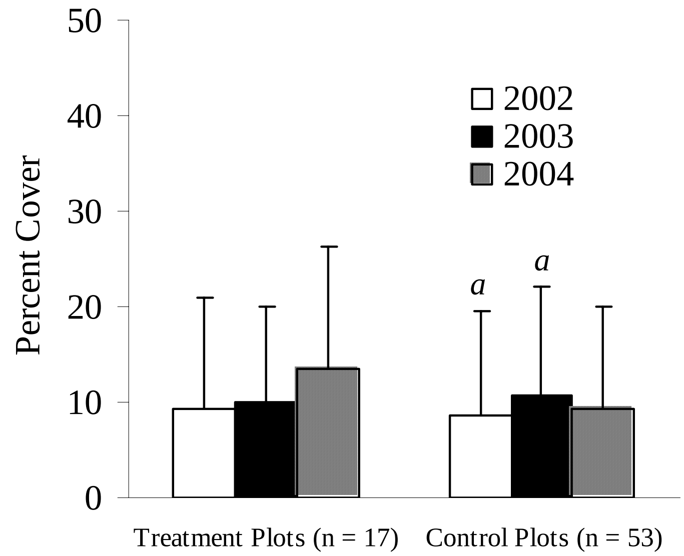
Sources of seeds for rehabilitation or restoration projects should be carefully considered. Existing native vegetation on the Hanford Site represents a poorly understood but potentially irreplaceable genetic resource, a pool of genetic variability uniquely adapted over millennia to the unique range of environmental variability that characterizes this portion of the Columbia Plateau. Seeds collected from distant sites may introduce genotypes that are poorly adapted to local conditions, may fail to establish vigorous populations, or, if successful, may have unpredictable impacts on the genetic integrity of local stocks.

The lack of commercial availability of sufficient quantities of locally derived native seed was a limiting factor in this project as it often is in large-scale rehabilitation and restoration efforts. Wildfire will remain a part of the shrub-steppe landscape and resource management agencies, from the local to the national level, will do well to plan responses to future wildfires before they occur, rather than forcing managers to scramble to respond after wildfires have passed. Restoration opportunities are often greatest within the first one to several years following wildfire, before invasive species populations recover or newly colonize a site (see Section I). For the Monument to take advantage of this window of opportunity it must develop, either on its own or in concert with partners and contractors, the capacity to stockpile native seeds that will be available when needed soon after disturbances occur. The U.S. Fish and Wildlife Service and the U.S. Department of Energy, as co-managers of the Hanford Reach National Monument, are urged take the lead in promoting the creation of a Columbia Basin native seed storage bank. If willing partners are found, a seed storage bank of this kind could serve a consortium of federal and state agencies.

Disturbed sites in the Wyoming big sagebrush steppe of the Columbia Basin present extreme challenges to restoration efforts due to the warm, semi-arid climate, the presence of aggressive invasive species, and the increasingly frequent occurrence of wildfires within the region. Beyond initial seeding and planting success, restoration outcomes depend upon climatic factors that are outside the control of the restorationist, as well as upon effective fire management and control of invasive species.

No project can hope to restore the ecological structure and function of a complex ecosystem such as that which existed on the ALE Reserve within a few years. However, through persistent effort, a landscape may be set on a successional trajectory that will lead to the recovery of ecological processes and habitat quality within a reasonable period of time. Complete recovery of the structure and function of ALE shrublands impacted by the 24 Command Fire is still decades away. The most optimistic scenario for the full recovery of shrub-steppe qualities on the Arid Lands Ecology Reserve involves many years of continued planting and monitoring, persistent efforts at weed and fire management, and years of patience as restored stands slowly develop.

a.



b.

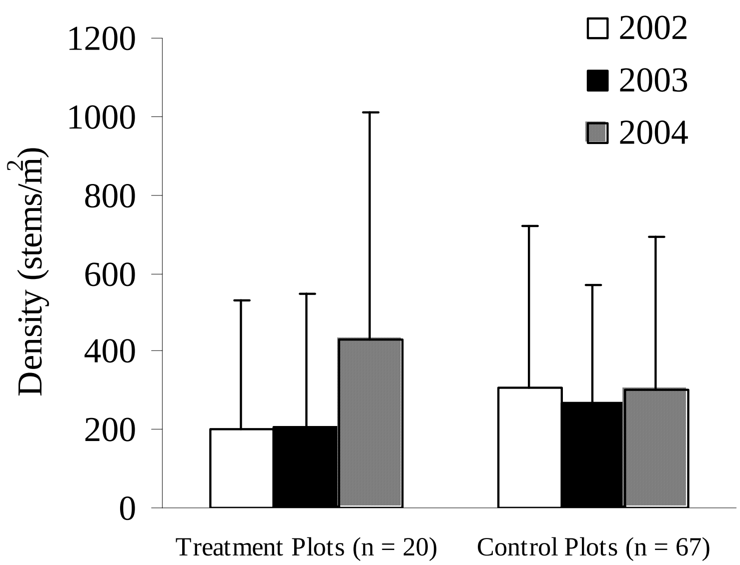
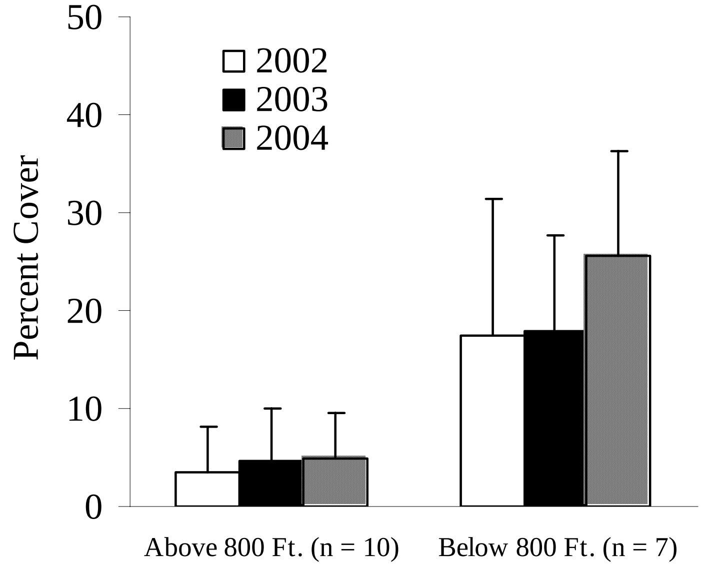


Fig. S7. Abundance measures of cheatgrass (*Bromus tectorum*) in long-term permanent plots in ALE Reserve rehabilitation areas: (a) percent cover; (b) density (stems/m^2). Bars = 1 standard deviation. Statistical comparisons are between 2002 and 2003 values only. Columns exhibiting the same script letter are significantly different ($P < 0.01$).

a.



b.

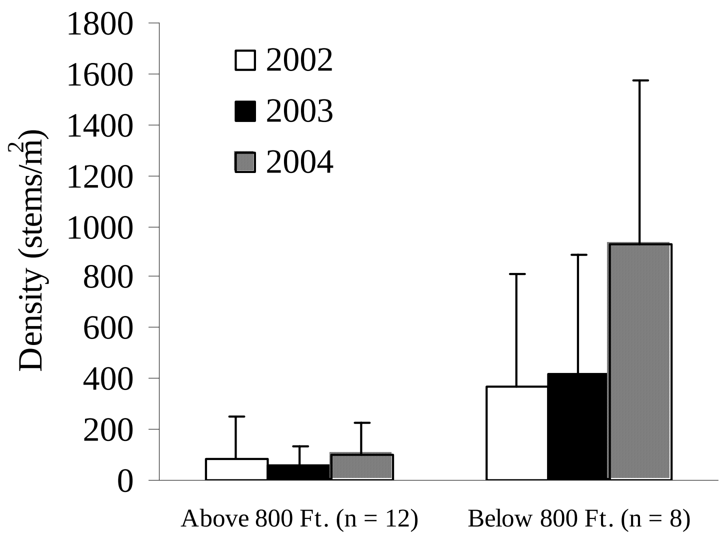
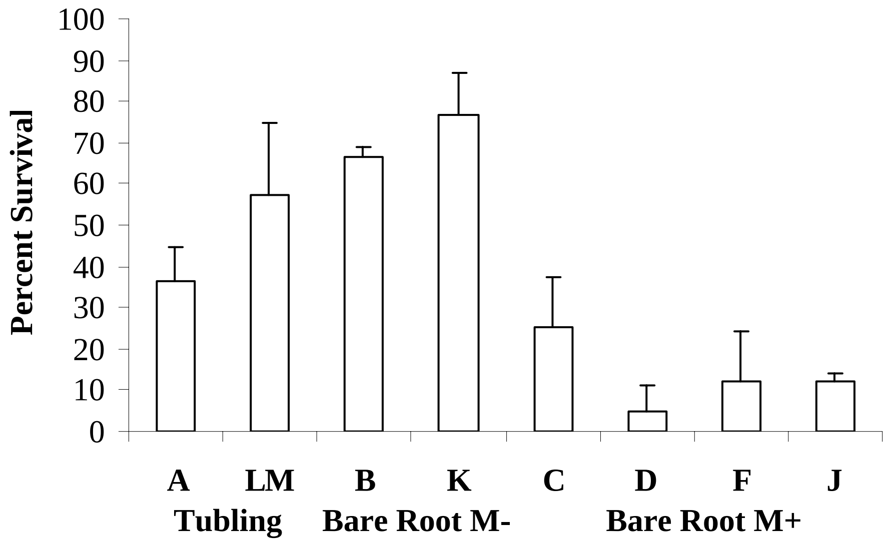


Fig. S8. Abundance measures of cheatgrass (*Bromus tectorum*) in long-term permanent plots in ALE Reserve rehabilitation areas by elevation: (a) percent cover; (b) density (stems/m^2). Bars = 1 standard deviation. Statistical comparisons (between 2002 and 2003 values only) yielded no significant differences.

a.



b.

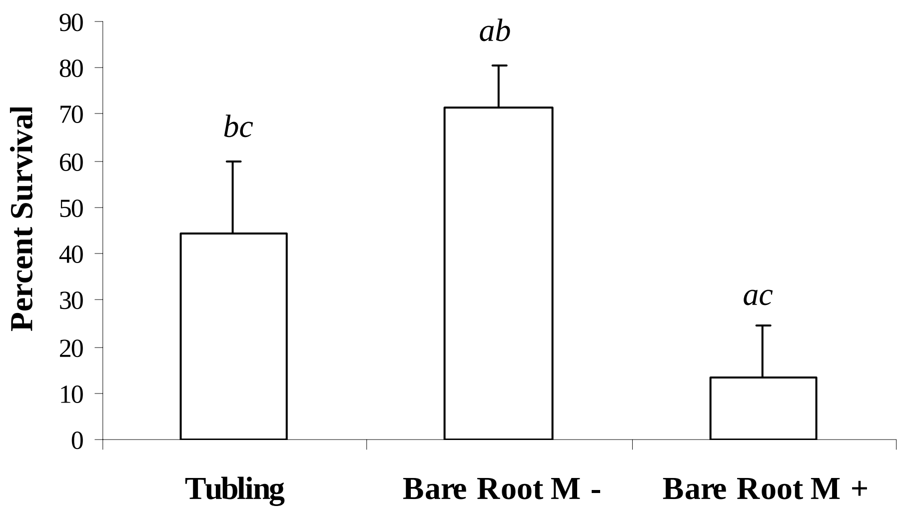


Fig. S9. Two-year survival of Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) outplantings in ALE Reserve rehabilitation areas, fall 2004: (a) survival by polygon; (b) survival by treatment type. Treatment codes are as follows: M + = inoculated with mycorrhizal fungi (Polygons C,D,F,J); M - = not inoculated (Polygons B, K). Bars = one standard deviation.

Treatments with identical superscripts in (b) are significantly different (ANOVA: $P < 0.0001$ followed by Tukey pairwise multiple comparisons, $\alpha = 0.05$).

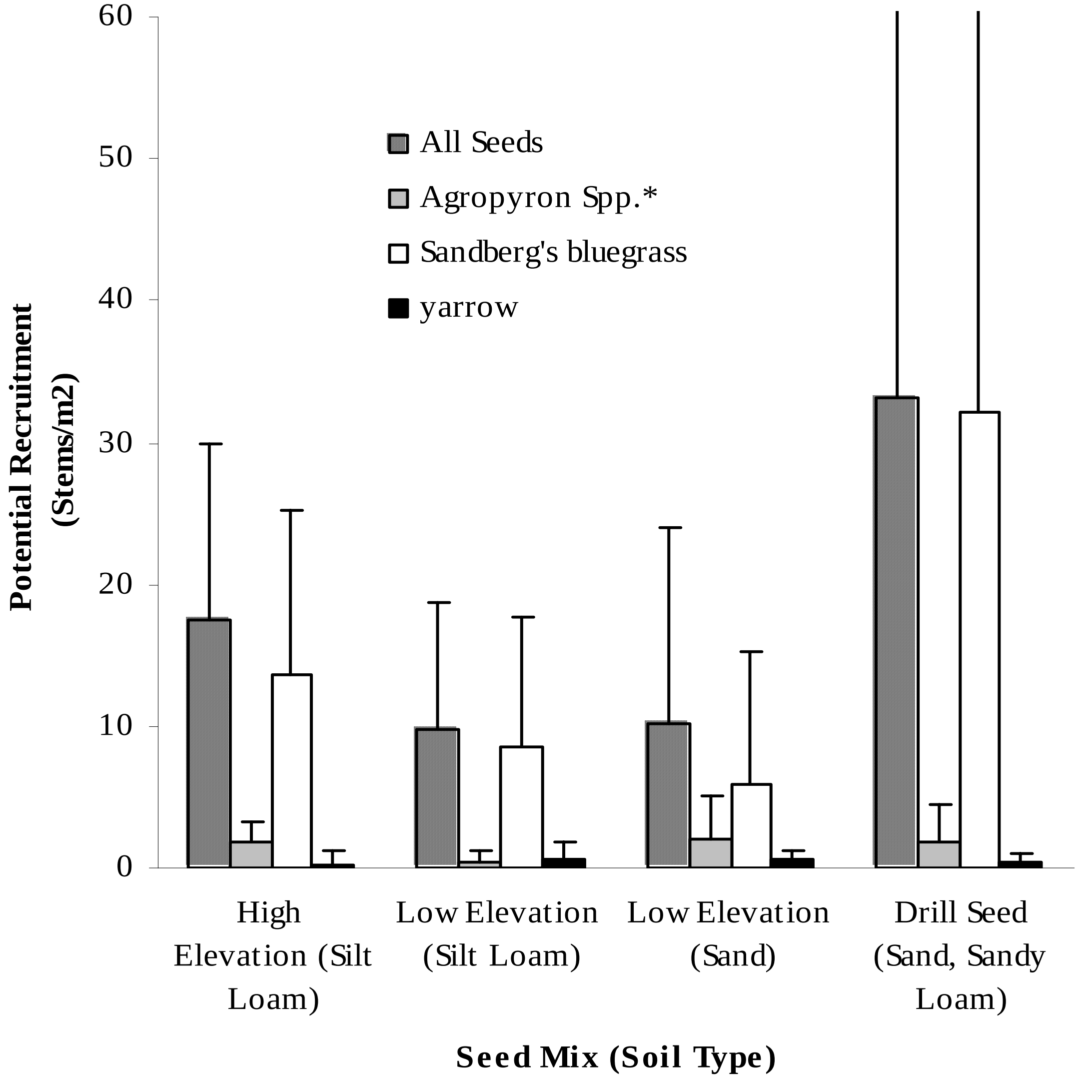


Fig. S10. Potential recruitment densities (seedlings/m^2) of seeded species in rehabilitation project areas, ALE Reserve 2004. Bars = 1 standard deviation (SD). The large SD of Sandberg's bluegrass in the drill seed area drives the large overall SD in that treatment and in overall values.

*Species composition of seed mixes differed by elevation. *Agropyron* species refers to bluebunch wheatgrass (*Agropyron spicatum*) in the high elevation mix (> 800 ft.) only, and thickspike wheatgrass (*Agropyron dasystachyum*) in the low elevation mix (< 800 ft.) only.

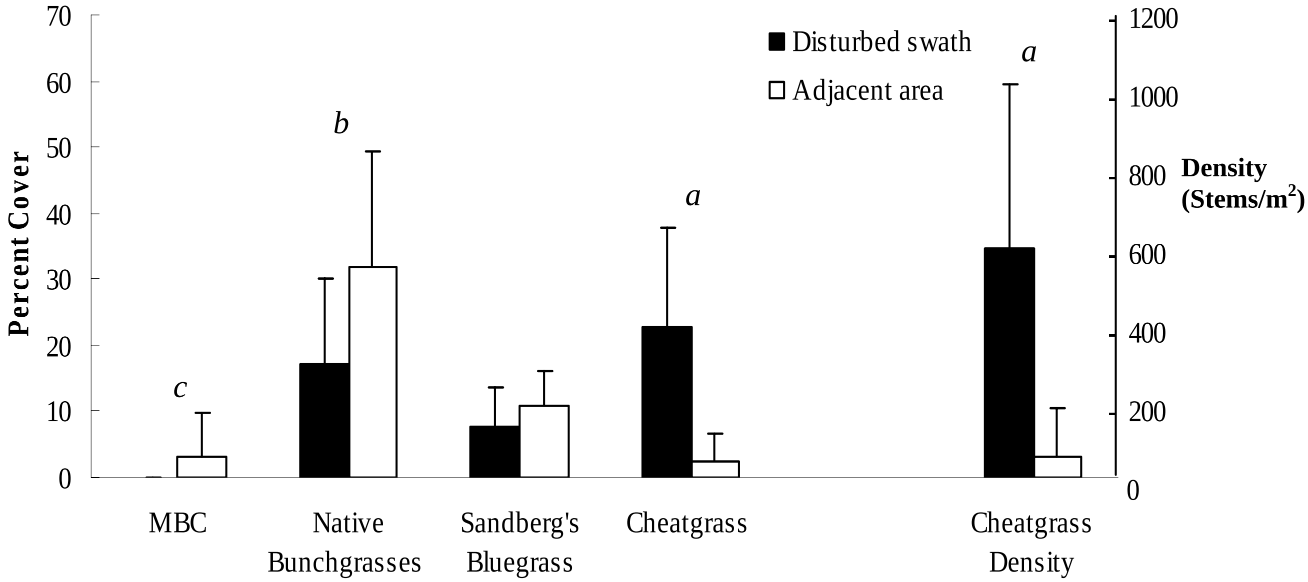


Fig. S11. Abundance of native and invasive species in the disturbed fire suppression swath compared to adjacent, less disturbed grasslands in upper Snively Basin, ALE Reserve, 2004. Note the different units on the density axis along the right-hand side of the diagram. MBC = microbiotic crust. Column pairs are significantly different as follows: $a = P < 0.0001$; $b = P < 0.01$; $c = P < 0.05$. Bars = 1 standard deviation.

ACKNOWLEDGEMENTS

This study was conducted as part of Cooperative Agreement No. 13410 between the United States Fish and Wildlife Service and the Nature Conservancy. Thanks to the entire staff of the Hanford Reach National Monument for their help in all aspects of this project. It is a pleasure to work with such dedicated stewards of our public lands; they are too often underappreciated. Heidi Newsome, Wildlife Biologist, David Smith, Resource Management Specialist, and Jennie Meisel, Biological Technician, all were particularly generous with their time and assistance during the course of multiple studies on the Monument over the past four years. Thanks also to Jennie Barnett for the maps of rehabilitation areas presented in Part II. Dr. Janelle Downs of Pacific Northwest National Laboratory provided vegetation and fire history data for the ALE Reserve. We have benefited greatly from her friendship and from the insights into the ecology of the Columbia Basin that she has shared with us. Special thanks to Dr. Peter Dunwiddie of The Nature Conservancy of Washington for his invaluable editorial reviews of so many manuscripts and for the thoughtful guidance and encouragement he has provided from the inception of this project to its completion.

Other colleagues at The Nature Conservancy made important contributions to this project. Misha Henshaw's thoroughness with contractual and budgetary matters was invaluable. Mark Goering and Jesse Langdon produced the maps used in Part I of this report. Claire Bronson designed the cover and the layout of color illustrations. Without Neil Corcoran, Technology and Information Systems Wizard, no one would be reading this right now.

Volunteers played an extremely important part in data collection for this project, and our ambitious field objectives could not have been achieved without their generous assistance. Warmest thanks to Sue Carver and Bob Fortman, who rivaled project staff in their commitment to our work in the field. Alex Ammonette, Mickie Chamness, Paula Clark, Peter Evans, Jim Hansen, Scott Luchessa, Mike Marsh, Dennis Plank, and Wendy Waichler all contributed multiple days in the field under conditions ranging from chilling fog in midwinter to the wilting heat of summer. Many others shared a day in the field or a few hours in the Seattle office in support of this project. We gratefully acknowledge the help of all these friends and look forward to finding new opportunities to work together.

Special thanks are due to Dr. Michael P. Marsh and the dedicated volunteers who collected data for the Steppe-in-Time project from 1992 to 2001. Their data have provided an important long-term record of vegetation change in the Columbia Basin, and their efforts serve as a model of selfless public service in the interest of our natural resources.

INTRODUCTION

The studies in this volume were performed as part of a cooperative agreement between the U.S. Fish and Wildlife Service (USFWS) and The Nature Conservancy (TNC) of Washington. The purposes of these studies were to assess the effects of the June-July, 2000, 24 Command Fire on invasive species and native plant communities of the 77,000-acre Fitzner-Eberhardt Arid Lands Ecology (ALE) Reserve, and to monitor the progress of a large scale rehabilitation project begun in late fall, 2002, on a portion of the lands impacted by the 2000 wildfire.

The 24 Command Fire began on Tuesday, June 27, 2000, along Washington State Route 24 along the northern boundary of the ALE Reserve. Before being controlled on July 2, the fire burned more than 160,000 acres of public and private land, as well as a number of residences and structures. Included within the burned acreage was approximately 90 percent or more of the ALE Reserve (BAER Team 2000). The Nature Conservancy's effort in cooperation with USFWS to assess the effects of the wildfire on this important natural area began the following spring, 2001, and continued through mid-summer 2004. Results of post-fire vegetation monitoring are presented in Section I of this volume. In late November 2002 a rehabilitation effort targeting approximately 10,000 acres of the ALE landscape most seriously effected by the 24 Command Fire was initiated by USFWS and their contractors. Rehabilitation measures included herbicide treatment of invasive plant species, aerial seeding of native grasses, and planting of nursery-grown seedlings of Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*). Measures to monitor the results of rehabilitation efforts were begun concurrently with the rehabilitation efforts themselves, and continued through fall 2004. Results of this monitoring are presented in Section II of this volume.

The Arid Lands Ecology Reserve. The lands that now comprise the Fitzner-Eberhardt Arid Lands Ecology (ALE) Reserve lie within the southwestern portion of the U.S. Department of Energy site at Hanford, in Benton County, Washington (Fig. 1). While the lands within Reserve boundary have been a part of the Hanford Site since its establishment in 1943, the ALE Reserve was formally established in 1967 by the Atomic Energy Commission in recognition of the rich and relatively undisturbed character of its native shrub-steppe ecosystem (O'Connor and Rickard 2003). The Reserve was subsequently designated a federal Research Natural Area (1971) and National Environmental Research Park (1977). In 2000, the ALE Reserve was incorporated into the newly designated Hanford Reach National Monument.

The ALE Reserve lies within the Columbia Basin, the hottest, driest part of Washington state (Franklin and Dyrness 1973). Environmental characteristics are summarized in Rickard et al. (1988), Soll et al (1999), and elsewhere. Elevations range from as low as 435 ft. (132.5m) a.s.l. in the Cold Creek drainage near the southeastern boundary of ALE to more than 3500 ft. (1067m) at the summit of Rattlesnake Mountain near the western boundary. Annual precipitation varies with elevation, from as little as 6.3 inches (16 cm) at the lowest elevations up to 13.8 inches (35 cm) along the crest of Rattlesnake Mountain (DOE-RL 2001).

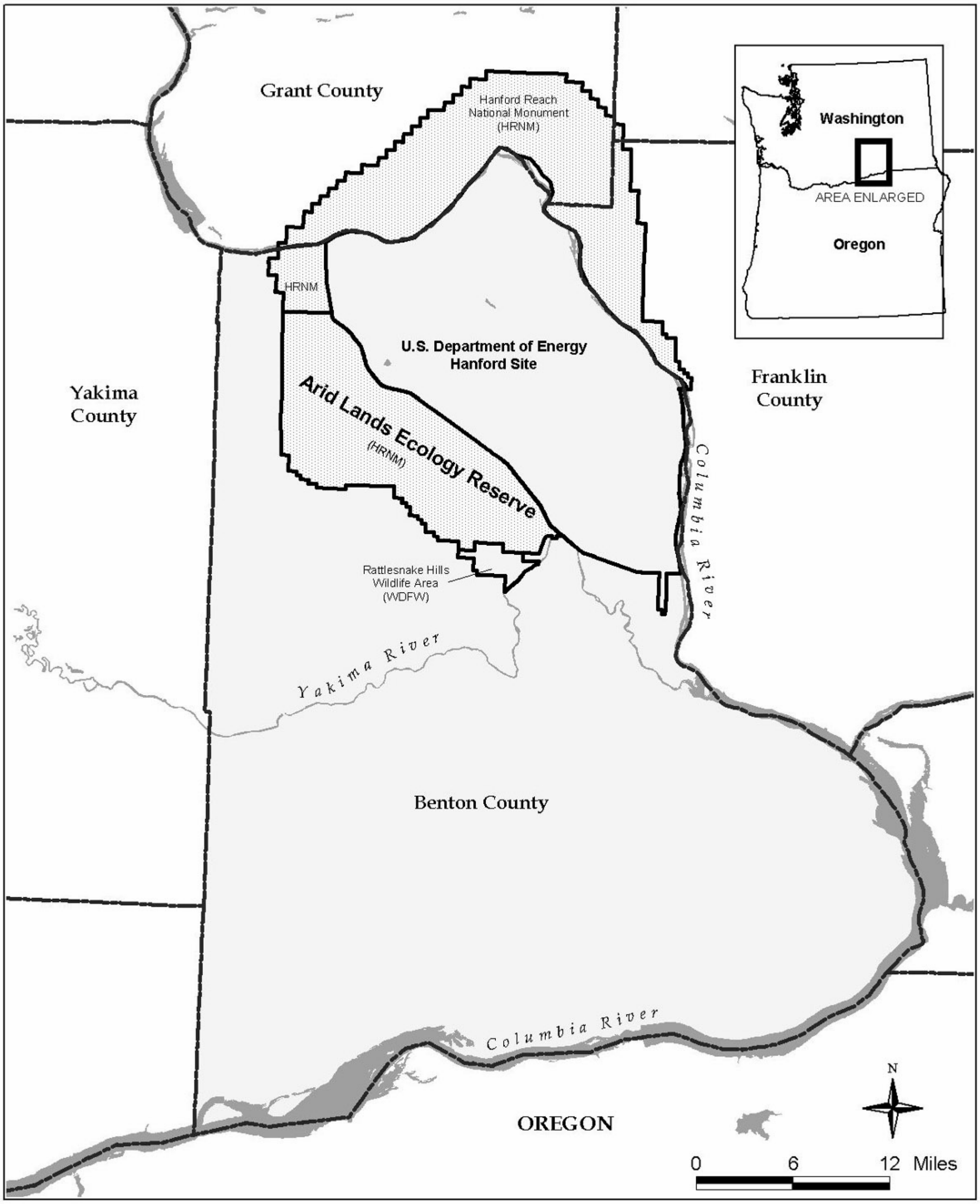


Figure 1. The Fitzner-Eberhardt Arid Lands Ecology (ALE) Reserve in relation to the Hanford Site and the surrounding area, Lower Columbia Basin, Washington.

Plant Communities of the Arid Lands Ecology Reserve. Most of the landscape of the ALE Reserve was dominated by sagebrush (*Artemisia*) species as recently as the establishment of the Hanford Nuclear Reservation in 1943. However, between 1957 and 1998, major wildfires reduced the shrub-dominated portion of the Reserve to less than 20 percent of its extent at the mid-point of the 20th century (Rickard et al, 1988, Soll, et al, 1999, T. Skinner, pers. comm., 4/01). Major plant communities of the ALE Reserve prior to the 24 Command fire included shrublands dominated by Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) at low and middle elevations, primarily in the northern portion of the Reserve. Shrublands dominated by three-tip sagebrush (*A. tripartita*), sometimes with big sagebrush or rabbitbrush (*Chrysothamnus* spp.) as important components, occurred at the middle to higher elevations, primarily on more or less northerly aspects, on the slopes of Rattlesnake Mountain and in the Rattlesnake Hills. Other shrubland types included a small black greasewood (*Sarcobatus vermiculatus*) community surrounding Rattlesnake Springs and a series of areas between 800-1200 ft. (250 – 365 m) a.s.l. characterized by winterfat (*Eurotia lanata*). Shrubland understories were dominated by native bunchgrasses such as bluebunch wheatgrass (*Agropyron spicatum = Pseudoroegneria spicata*) and Sandberg's bluegrass (*Poa sandbergii = P secunda*), and associated hemishrubs and forbs (Wilderman 1994).

Where previous wildfires or human activities had cleared the sagebrush, perennial and annual grasslands dominated the landscape. Bluebunch wheatgrass-Sandberg's bluegrass was the most common perennial grassland association, occurring at elevations as low as 800' – 1000 ft. (250 – 300m) and upwards. Needle-and-thread (*Stipa comata*) dominated another perennial grassland association found from lower to middle elevations, often on sandy soils. Another perennial bunchgrass, Idaho fescue (*Festuca idahoensis*), became important at higher elevations and on more northerly aspects, often in association with threetip sagebrush and bluebunch wheatgrass. Soil surfaces in relatively undisturbed native grasslands and shrublands were characterized by a microbiotic crust composed of mosses, lichens, fungi, algae, and cyanobacteria (Link, et al., 2000, Johansen, et al. 1993), especially on silt-loam soils.

Highly disturbed areas on the ALE Reserve were frequently dominated by alien invasive plant species. Among others, alien perennials such as Russian knapweed (*Acroptilon repens*) and whitetop (*Cardaria draba*), were common in riparian areas and some uplands, while rush skeletonweed was abundant in disturbed uplands in the southeast corner of the site (Evans et al. 2003). Most disturbed uplands, however, were dominated by alien annual species such as tumble mustard (Sisymbrium altissimum) and redstem filaree or storksbill (*Erodium cicutarium*), and, most importantly, the invasive annual grass cheatgrass (*Bromus tectorum*). Annual grasslands dominated by cheatgrass were typical at lower elevations, in the Cold Creek and Dry Creek valleys, and on Yakima Ridge (Wilderman, 1994), but were not limited to these elevations.

Cheatgrass arrived in the Pacific Northwest during the late 19th Century (Mack 1981). The spread of this Mediterranean annual throughout western North America has been accompanied by increases in the frequency, extent, and severity of rangeland wildfires, and is frequently associated with declines in native plant diversity (Whisenant 1990) and wildlife habitat (Dobler 1994, Connelly, et al. 2000). Informative reviews and discussions of cheatgrass ecology and influence on rangeland wildfire regimes are contained in Young and Evans (1985), Pellant (1996), Carpenter and Murray (1999) and other sources.

Anthropogenic factors have promoted the proliferation of cheatgrass throughout western North America (Mack 1981, Stewart and Hull 1949) and land use activities have likely contributed to the present distribution and abundance of cheatgrass, as well as other invasive plant species, on the ALE Reserve. While the establishment of the Hanford Reservation protected what is now the ALE Reserve from the development that has characterized much of the surrounding area in the decades since 1943, the lands that comprise ALE have a history of land use and disturbance dating from the early to mid-19th Century and continuing to the present. Since the onset of Euroamerican influences in the Columbia Basin in the 1800s, major land uses on what is now the ALE Reserve have included homesteading, dryland and irrigated agriculture, sheep and cattle grazing, natural gas exploration and production, military and scientific activities, and construction and maintenance of roads, powerlines, and communication facilities (Hinds and Rogers 1991, O'Conner and Rickard 2003).

Populations of at least nine rare vascular plant taxa have been documented on the ALE Reserve (Table 1). The shrub-steppe communities of ALE, along with the rest of the Hanford Site, provide critical habitat for native biota, including shrub-steppe dependent wildlife and a diverse invertebrate fauna (Rickard and Poole 1989, Soll, et al. 1999, DOE-RL 2001). Listed vertebrate species known to inhabit the area include reptiles, birds, and small mammals (Table 2). At least one endangered shrub-steppe mammal, the pygmy rabbit (*Brachylagus idahoensis*) has been extirpated locally in recent decades (Rickard and Poole 1989, Soll, et al. 1999).

Table 1. Vascular plant species of management concern known to occur on the Arid Lands Ecology Reserve. Sources include Soll, et al. (1999), WNHP (1997, 2000), and Sackschewsky and Downs (2001).

Scientific Name	Common Name	Federal Status	State Status
Local Endemics			
Astragalus columbianus	Columbia milkvetch	Species of Concern	Threatened
Astragalus conjunctus var. rickardii	basalt milkvetch		Review
Erigeron piperianus	Piper's daisy		Sensitive
Regional Endemics			
Camissonia minor	smallflower evening primrose		Review
Camissonia pygmaea	dwarf evening primrose		Threatened
Cryptantha scoparia	desert cryptantha		Review
Cryptantha spiculifera	Snake River cryptantha		Sensitive
Nana densum var. *parviflorum*	small-flowered nana		Review
Oenothera caespitosa ssp. *caespitosa*	desert evening primrose		Sensitive

Table 2. Wildlife species of management concern known to occur in shrub-steppe or grassland habitats on the Arid Lands Ecology Reserve. Sources include Soll, et al. (1999), H. Newsome (pers. comm.), and Appendix C, DOE-RL (2001).

Scientific Name	Common Name	Federal Status	State Status
Reptiles			
Masticophis taeniatus	striped whipsnake		Candidate
Sceloporus graciosus graciosus	northern sagebrush lizard	Species of Concern	
Birds			
Amphispiza belli	sage sparrow		Candidate
Athene cunicularia	burrowing owl	Species of Concern	Candidate
Buteo regalis	ferruginous hawk	Species of Concern	Threatened
Centrocercus urophasianus	greater sage grouse		Threatened
Lanius ludovicianus:	loggerhead shrike	Species of Concern	Candidate
Oreoscoptes montanus	sage thrasher		Candidate
Mammals			
Lagurus curtatus	sagebrush vole		Monitor
Lepus californicus	black-tailed jack-rabbit		Candidate
Sorex merriami	Merriam's shrew		Candidate

I. Short-Term Impacts of the 24 Command Fire on Vegetation of the Fitzner-Eberhardt Arid Lands Ecology Reserve, 2000-2004.

INTRODUCTION

Investigations into the effects of the 24 Command fire on the vegetation of the Arid Lands Ecology (ALE) Reserve began in March 2001. The goals of the study were twofold: 1) to assess the short-term effects of the wildfire on native plant communities and on the abundance and distribution of alien plant species and to make inferences regarding trajectories of recovery for burned-over lands; and, 2) to use existing permanent vegetation plots and methodologies, and to institute new methodologies where necessary, to assemble long-term datasets that may be applied toward management of the Reserve's vegetation resources. To accomplish these goals, three sets of vegetation plots that had been installed for various purposes and sampled from one to several years prior to the 24 Command Fire were relocated and resampled using the original methodologies during April through June, 2001-2004. The results of these samples were then compared to pre-fire observations from the same sites. In addition, a series of new plots was established and new protocols were added to the sampling of reference plots in order to monitor recent and future changes in alien species.

MATERIALS AND METHODS

VEGETATION PLOTS

Vegetation reference plots used in this study originated from three sources: 1) the Biodiversity Inventory and Analysis of the Hanford Site (Soll, et al, 1999, Wilderman, 1994); the Steppe-in-Time project of the Washington Native Plant Society (Marsh, 1999), and the Hanford Site Biological Resources Management Plan (DOE-RL 2001, PNNL 1997). Characteristics and methodologies associated with the different plot types are described below. From 2001 through 2004 vegetation plots were resampled duplicating original methodologies as much as possible and sampling as close to the date of the original surveys as possible. Photographic records of each plot were taken annually from 2001 through 2004. Images were recorded in 2001 with a Nikon FM2 SLR camera using 35mm slide film. Images were later professionally digitized (ProLabWest Visual Imaging Services, Inc., Seattle, WA.). Images from 2002 through 2004 were recorded using a Nikon Coolpix 995 digital camera. A digital archive of plot images was delivered to USFWS along with this report. Coordinates, directions, and details of plot layouts for all the plots used in this study since 2001 are provided in Appendix A.

For a variety of reasons, several plots that were sampled in 2001 and 2002, from which data was included in analyses contained in earlier annual project reports, were not resampled after 2002. Reasons why sampling was discontinued at these plots are discussed at the end of this section. Vegetation analyses presented in this report used data only from 70 long-term vegetation plots for which a complete data record from pre-fire reference years through 2004 was available.

Biodiversity Plots. Biodiversity Plots were established on the ALE Reserve in 1994 during vegetation inventories associated with the Biodiversity Inventory and Analysis of the Hanford Site (Soll, et al, 1999). Plots were established in stands representative of major native plant community types for purposes of ground-truthing vegetation maps, classifying plant communities, and comparing mapped communities to conservation 'element occurrences' (Wilderman 1994). Areas that were highly disturbed or dominated by aliens were generally avoided. Percent cover of vascular plant species and microbiotic crust was visually estimated within 5m x 20m plots and recorded according to the following scale:

> Present - <1% = 1; 1%-5% =3; 6%-15%=10; 16% - 25% =20; 26%-35%=30; 36%-45% = 40; 46%-55% = 50; 56%-65% = 60; 66%-75% = 70; 76%-85% = 80; 86%-95% = 90; 96%-100%=100 (Bourgeron et al. 1992, cited in Wilderman 1994).

Percent cover of litter was estimated to the nearest percent in 20 cm x 50 cm ($0.1m^2$) microplots along cheatgrass density transects (see below) associated with each Biodiversity Plots in 2004 only.

In all, 58 Biodiversity Plots were sited on the ALE Reserve in 1994. In 2001, 40 of these plots were selected for resampling as part of post-fire monitoring. Plots were selected for resampling based on meeting one or more of the following biological criteria:

- Presence of $\geq$ 10% cover of sagebrush species and/or other major native. shrubs prior to the 24 Command Fire.
- Representation of major native plant community types.
- Representation of the range of pre-fire (1994) cheatgrass cover from low (0%) to high (20%-40%).
- Presence of Piper's daisy (*Erigeron piperianus*), a rare plant (WNHP 1997, 2000) recorded within some plots in 1994.
- High diversity of native forbs.

Questions of field time efficiency, such as access and proximity to other plots, were also considered in selecting the target plots. Between 2002 and 2003 sampling was discontinured at a number of Biodiversity Plots for reasons discussed at the end of this section. Data from 33 Biodiversity Plots were included in the analyses for this final report.

Plots were relocated using GPS coordinates and print reproductions from 35 mm slides taken during the original survey. Only plots that could be relocated with reasonable confidence were resampled. Plots were marked permanently in 2001 using ½-inch rebar stakes at all four corners (Appendix A).

In order to estimate the wildfire's impact upon native bunchgrasses in study plots, survival of large bunchgrasses was recorded in all Biodiversity Plots within the fire perimeter in 2001. Survival of large bunchgrasses was estimated within eight randomly-selected 1m-wide belt transects located along the long axis of each Biodiversity Plot and running across the width of the plot. Within each belt transect all bunchgrasses with a pre-fire basal diameter of 5.0 cm or greater were tallied according to species if some part of the tussock was alive. Burned tussocks with no apparent living tissue were tallied as dead. The resulting sample included all warm season bunchgrasses of mature size, as well as larger individuals of Sandberg's bluegrass (*Poa sandbergii*) but excluded smaller individuals of *Poa* which are numerous and frequently difficult to delineate between individual plants. Although 40 Biodiversity Plots were thus sampled in 2001, four plots had no bunchgrasses large enough to tally. Of the 36 remaining plots, those with less than 70% survival in 2001 (n = 15) were resurveyed in 2002. Combining the density of both living and dead bunchgrasses in survival samples yielded a crude estimate of the density of large bunchgrasses in ALE habitats prior to the 24 Command Fire.

Biological Resource Management Plan (BRMaP) Plots. Biological Resource Management Plan (BRMaP) Plots were established across the Hanford Site in 1996 through a cooperative effort involving the U.S. Department of Energy (DOE), Pacific Northwest National Laboratory (PNNL) and the Washington Department of Fish and Wildlife (WDFW). Plots were established to provide baseline data on important terrestrial plant communities and wildlife habitat (DOE-RL 2001, PNNL 1997). Seven BRMaP Plots were established on the ALE Reserve in areas dominated by big sagebrush (3 macroplots), threetip sagebrush (2 macroplots), and bluebunch wheatgrass grassland (2 macroplots).

Each 20 ha BRMaP macroplot hosts an array of 3 – 5 vegetation plots. A 1.0 km transect runs through the center of each macroplot, with five permanently marked points or point count stations located at 200m intervals. Point count locations serve as the origins of permanently marked vegetation transects oriented perpendicular to the main axis of the plot. Vegetation transects were established from each of five point count stations in big sagebrush stands, but only from point counts 1, 3, and 5 in grasslands and threetip sagebrush stands. Macroplot diagrams showing the orientations of vegetation transects for each BRMaP array are included in Appendix A. Along each vegetation transect, visual estimates of percent cover of vascular plant species, microbiotic crust, and plant litter were recorded to the nearest full percent within 20cm x 50cm ($0.1m^2$) microplots (n = 20) located at regular intervals every 5m along each vegetation transect. The minimum score for a trace occurrence within each microplot was 1.0%.

Steppe-in-Time (SIT) Project. Steppe-in-Time (SIT) was a volunteer program initiated to monitor change in the shrub-steppe vegetation of the Columbia Basin. The program was initiated in 1992 by the US Environmental Protection Agency to monitor potential effects of agricultural herbicide drift upon native plant communities. The project continued as a volunteer program from 1993 to 2001 coordinated by Michael P. Marsh, Ph.D., with support from the Washington Native Plant Society and staff from Columbia Basin College and the Pacific Northwest National Laboratory. In 1992, 32 plots (or plot pairs; see below) were established across Benton County. Plot origins were randomly-selected points within the shrub-steppe cover type that met criteria for native plant predominance and relative lack of recent disturbance. Through cooperative agreements with managing agencies and private individuals, sample sites were established on public and private lands throughout the county. A few of the sites on private property were subsequently lost to development or other closure. Replacement sites were established in 1993, 1996 and 2001.

SIT Plots were laid out as 200 m transects oriented to magnetic north from their point of origin. Along each transect, visual estimates of percent cover of vascular plant species, microbiotic crust, and plant litter were recorded to the nearest full percent within

20cm x 50cm (0.1m^2) microplots (n = 20) located at regular intervals every 10m along the transect. The minimum score for a trace occurrence within each microplot was 1.0%.

SIT Plots established in 1992 consisted of paired transects separated by 300m. Six transect pairs were established on the ALE Reserve at this time. This study treats each transect as a separate plot. Three additional sites, consisting of a single 200m transect each, were established in 1996.

Sampling was not conducted every year. Principal data collection years of the Steppe-in-Time Project were 1992, 1993, 1997, 2000, and 2001. A smaller subset of plots were sampled in 1996. Even so, the SIT plots represent the only array of permanent plots on the ALE Reserve with multiple years of pre-fire data, with a data record stretching as far back as 1992, and with a data record from spring 2000, immediately prior to the occurrence of the 24 Command Fire.

Data collections from the year 2000 were used as the reference year for pre- and post-fire comparisons in this study. With the exception of one plot (SIT 188), only even-numbered microplots (n = 10) were sampled in 2001. Only data from those microplots which were sampled in 2001 were used to make between-year vegetation comparisons, even though all microplots were sampled during 2000 and 2002 - 2004.

Steppe-in-time practice through 2001 was, whenever possible, to sample plots in both April and May of each sample year. Cover values used in this study represent the means of cover estimates for the two sample periods each year. Exceptions were made for two species, cheatgrass and Sandberg's bluegrass, whose senesced foliage was often recorded as litter during May visits according to SIT convention. In order to obtain the best estimate of annual live cover, April values were used for these species. From 2002 through 2004 plots were sampled only once per year between late April and early May.

Cheatgrass cover and frequency data from unburned SIT transects outside of the ALE boundaries were used to compare cheatgrass abundance in burned (ALE) vs. unburned (non-ALE) plots and to provide an indication of between-year variations in cheatgrass abundance in the absence of a large-scale disturbance. Unburned plots were either big sagebrush stands or grassland cover types distributed around Benton County south of the ALE Reserve. Elevations of unburned, non-ALE sites ranged from 600 ft. (183m) to as high as 2600 ft. (793m) but 18 of the 23 sites used occurred between 800 ft. (244m) and 1400 ft. (427m).

Twenty-three SIT transects outside of the ALE reserve had full records of even numbered microplots for the years 1992, 1993, 1997, and 2000. Between-year variations in this set of plots were examined to provide insight into longer-term fluctuations in cheatgrass abundance in the absence of a large-scale disturbance. This analysis was extended into 2001, although data in 2001 was available from only 16 of the original 23 plots.

Nearly all SIT data through 2001 were collected by volunteers (the TNC field crew assisted with two plots in 2001). SIT volunteers – typically people with academic study or long-standing interests in biology and natural history -- received a minimum of 16 hours of training in plant identification and field methods from Dr. Marsh, staff ecologists from Pacific Northwest National Laboratory, and faculty from Washington State University and Columbia Basin College. Observers worked in teams of two or three, and plots were sampled twice – by different observer teams – during each observation period as a check on accuracy. In 2002 through 2004, SIT plots on ALE were sampled only by the TNC field crew. Plots outside of the ALE Reserve were not sampled after 2001.

Vegetation plots not resampled after 2002.

Resurveying of selected plots from the original 2001 sample set was discontinued in order to allow time for monitoring of post-fire rehabilitation efforts on ALE (section II, this volume). Reasons for discontinuing sampling of plots included difficulty or errors in relocating plots, the confounding of data interpretation by the occurrence of multiple recent wildfires effecting some plots, difficulty of access, and potential for plot damage through repeated sampling.

The largest group of discontinued plots consisted of five plots (Biodiversity Plots 33 and 34, and BRMaP 23, transects 1, 3, and 5) located in the Cold Creek Valley east of Yakima Ridge, between the Gate 118 and Gate 120 roads. A wildfire in 1998 removed shrub canopies and highly disturbed these sites two years prior to the 24 Command Fire, which also burned through the area. The occurrences of multiple recent disturbances appeared to set these plots apart from the rest of those used in the study. In both 2001 and 2002, these five plots exhibited by far the highest cheatgrass cover and densities recorded in the study and were removed from many analyses because of their outlier characteristics. The vegetation of these plots has been dramatically changed since the original surveys in 1994 and 1996. Because of the close occurrence of the two wildfires, we were unable to seperate the effects of the 24 Command Fire from that of the previous fire or from the cumulative effects of the two fires.

Seven other Biodiversity Plots were also excluded from sampling after 2002. Two of the 42 plots sampled in the 2001 sample set were unburned by the 24 Command Fire and were never included in analyses. One plot sampled in 2001 could not be relocated in 2002. Another plot was determined to have been mislocated in 2001 and was relocated in the correct position in 2002; however, lacking data for 2001 this plot proved of limited utility and had to be left out of most subsequent analyses . Two plots (Biodiversity plots 105 and 106) were discontinued because of difficulty of access when field schedules became more crowded. One plot (Biodiversity plot 59), a high elevation grassland rich in native forbs and in excellent condition in 2001 and 2002 was excluded because the steep (40^o) slope on which it was situated made sampling highly disruptive to the soils and vegetation of the plot and the surrounding area.

While the vegetation reference plots used in this study represent very well the native upland shrub-steppe communities of the Arid Lands Ecology Reserve, some types of vegetation are not represented. Biodiversity plots and BRMaP plots were originally located subjectively with an intentional bias towards high quality native plant communities and/or wildlife habitat (Wilderman 1994, PNNL 1997). Biodiversity plots in particular were chosen to characterize the best observable native plant communities at the time. Steppe-in-Time sites, while located from a pool of random points within the shrub-steppe cover class, were accepted as plots locations only if native species dominated the surrounding plant community. Large areas where alien plant species have dominated since before the original establishment of these plots were thus not sampled quantitatively in the course of this study. Special habitats such as riparian areas and talus are not represented by any plots, and lithosol habitats are barely included in portions of a few transects.

CHEATGRASS DENSITY

Vegetation Plots. Measurements of cheatgrass density were initiated in 2001 in existing permanent vegetation plots and in new plots established specifically for density measurements. Density measurements were begun in Biodiversity and BRMaP plots in 2001, and in SIT Plots in 2002.

Density measurements in Biodiversity Plots were recorded as follows. Paired, parallel 50m transects were positioned along the long (20m) sides of each plot. Transects began at the photo point end of the plot and extended 30m beyond the end of the plot (Appendix A, Fig. 1a). In 2001 and 2002, cheatgrass stems were counted within two randomly located 20cm x 20cm microplots within each 5m segment of the transect (40 microplots/plot). Random points were the same along each parallel transect and microplots were oriented towards the interior of the plots. In 2003 and 2004, sampling intensity was reduced to three microplots within each 10m segment of the 50m transects (30 microplots/ plot).

Measurements in BRMaP and SIT Plots utilized vegetation microplots. Cheatgrass stems were counted within the 20cm x 20cm subsection of each vegetation microplot directly adjacent to the transect (Appendix A, Fig. 1b).

Sagebrush Refugia Plots. Refugia plots were non-permanent plots set up to compare stem densities of cheatgrass between burned areas and adjacent unburned refugia. Plots were located along the borders of fragments of big sagebrush stands that had escaped the 24 Command Fire and met the following criteria: a bordering area that had obviously burned in 2000; a distinct enough boundary between burned and unburned areas so that burned (treatment) and unburned (control) microplots could be placed close together; soils and other environmental factors appeared to be similar between burned and unburned sides of the boundary. A border long enough to accommodate a 100m transect was considered optimal; however, refugia were either too small or distinct boundaries too discontinuous to achieve this in some cases. Following selection of a site, a baseline transect was laid out along the border of the refugia and divided into 10m segments. Within each 10m segment, three secondary transects were run perpendicular to and crossing the baseline at randomly chosen whole-meter points along the baseline. No secondary transect could be immediately adjacent to another secondary transect. Cheatgrass stems were counted within 20cm x 20cm microplots located in random pairs along secondary transects on each side of and equidistant from the baseline. One pair of microplots was located at randomly selected points between 2-7m on either side of the baseline, and the second pair of microplots was located at points between 7-12m. A 2m setback on either side of the baseline was not sampled, to help absorb the variability in the fire boundary.

GPS coordinates were recorded for refugia plots, but plots were not permanently marked. Plots in 2002 through 2004 were placed in approximately the same locations as plots in 2001.

Transition Density Plots. Transition Density Plots were established as permanent plots in 2001 to provide baseline data and to monitor changes in cheatgrass densities from roadways and other invasive corridors to the interiors of. native plant communities that were relatively free of cheatgrass. The majority of Transition Plots were located in high-quality bluebunch wheatgrass – Sandberg's bluegrass grasslands. Polygons of native plant communities with low densities of alien species were identified during field surveys. Plots consisted of a long transect (0.5 – 1.4 km) through the long axis of selected polygons, with one or more additional transects intersecting perpendicular to the longest transect. Because of topography and community boundaries, intersection was not always exactly perpendicular. Transect origins (typically along a road or wash) were marked with steel 'T-bar' fence posts. Transect lines were marked with ½" x 36" rebar at 100m intervals. All posts and rebar were marked with aluminum tags as follows: "T" (Transition Plot) followed by a number (the transect number) above, with a second number, the stake number (identifying distance along the transect) below. For example, a disk marked as follows:

T1
2

indicates Transition Plot T1 at the 200m point from the origin.

Cheatgrass density was sampled at the origin and every 25m along each transect (Appendix A, Fig. 1c) as follows: a 1m plot frame with 20cm divisions was placed at each sample point along the transect to the right of the transect line, facing back towards the origin (the origin itself is an exception to this model, where the plot frame faces away from the origin along the transect). Cheatgrass stems were counted within the 20cm subplot closest to the sample point. Cheatgrass density, including the nested 20cm subplot, was then counted (if $n < 25\text{-}30$) or estimated (if $n > 30$) within the entire 1m square. Observations regarding the presence and abundance of cheatgrass and other invasive species in the general area were recorded at each sample point.

For density analysis in this study, a sampling unit was considered to be a single long transect or clusters of 2 – 4 short transects within polygons.

OTHER METHODS

GPS Technology and Plot Locations. Plot location coordinates were recorded using a Garmin *etrex* Personal Navigator portable GPS unit. Location coordinates for all of the plots used in this study are presented in Appendix A. Navigation to known fixed points such as BRMaP plot markers using the Garmin *etrex* yielded results that were typically within 2-5 m and never more than 10m from the selected point. Operational settings used with the portable GPS unit were as follows:

Time: Zone – Pacific
Format – 12 Hr
Position format: UTM/ UPS
Map Datum: NAD27 CONUS
North Reference: Grid

Environmental Variables. Environmental variables associated with sample plots are presented in Appendix B. Slope (degrees), aspect, and elevation were recorded for every plot visited. Aspect values were recorded as magnetic bearings in the field and later corrected for declination. Prior to correlation analyses, aspect was rescaled symmetrically about the NE-SW axis to a scale of zero (the coolest aspects, 45^o) to one (the warmest aspects, 225^o) according to the following formula:

(Equation 1)
$$\frac{1 - \cos(\text{aspect}^o - 45)}{2}$$

(McCune and Keon, 2002)

The unitless product of this calculation approximates the influence of aspect upon the heat load of a site; however it fails to account for the influence of slope angle on solar incidence. To account for this important variable the following equation was used to calculate a heat load index:

(Equation 2)
$$0.339 + 0.808 * \cos(\text{latitude}) * \cos(\text{slope}) - 0.196 * \sin(\text{latitude}) * \sin(\text{slope}) - 0.482 * \cos(\text{folded aspect}) * \sin(\text{slope})$$

(McCune and Keon 2002, Eqn. 3)

With aspect folded symmetrically about the NE-SW axis similarly to Equation 1, Equation 2 integrates slope, aspect, and latitude into a unitless index of potential heat load ranging again from zero to one.

Following an examination of soil characteristics, each plot was identified as belonging to one of three general soil types, silt-loam, stony silt-loam, or sand, based on Hajek (1966) and Rasmussen (1971). Data on monthly and seasonal precipitation were taken from the Hanford Meteorological Station (HMS 2004) and the National Weather Service (NWS 2004).

During the 2001 field season fire severity was estimated based on the following scale:

0 – Did not burn.
1 – Low severity. Previous year's standing dead herbaceous biomass entirely removed, or a few small patches may remain.
2 – Moderate severity. Previous year's standing dead herbaceous biomass entirely removed. Previous year's herbaceous litter mostly to entirely removed. If shrubs present, branches singed and killed but fine twigs still at least partly present on the snags.
3 -- Moderate-high severity. All or nearly all previous year's standing and down herbaceous biomass removed. A few small patches or remnants of charred cheatgrass mats may remain. If shrubs present, main stems may remain standing but fine twigs are burned away. Some patches of charred ground present.
4 – High severity (shrublands only). Charred earth common. Shrubs burned down to short charred stumps or to ground level.

Percent cover of litter was visually estimated along with cover of vegetation in BRMaP and SIT microplots. Litter was not recorded in Biodiversity Plots until 2004, when it was visually estimated in 20cm x 50cm microplots that were extensions of the 20cm x 20cm microplots used for cheatgrass density counts in those plots. Since litter data was not available for all plots prior to 2004, only 2004 data was used as a variable in correlation tests.

Land use activities and disturbance. Land uses and related disturbance features on the ALE Reserve were identified using USGS 7.5' topographic maps, resource documents (Hinds and Rogers 1991, O'Conner and Rickard 2003) local experts (J. Gaston pers. comm., J. Downs pers. comm.. Valentine 2001) and personal observations. Distance from vegetation plots to the nearest identifiable land use disturbance features (Table 1.1) was plotted from the approximate center of each plot. Information on the spatial extent of wildfires between 1974 and 2000 was compiled from draft materials made available by Pacific Northwest National Laboratory, personal accounts (J.L. Downs pers. comm.), and vegetation maps (Downs et al. 1993). Many potentially important events and practices have not been mapped or documented and could not be captured in this analysis. These include disturbances associated with fire suppression activities, military and scientific activities away from established facilities, and open range livestock grazing. Of seven natural gas production sites described in Hinds and Rogers (1991) only four are located on USGS maps. The locations of the remaining developed sites and of up to nine test wells that were not subsequently developed , are not precisely known. .Valentine (2001) provided information regarding two gas well sites, but it is not clear whether these were developed sites or test wells.

For the purposes of analysis, land use and disturbance features were placed into one of five categories based on shared characteristics (Table 1.1). Pearson's product-moment correlation was used to examine possible relationships between cheatgrass abundance and distance to both specific land uses or disturbance types and the broader categories.

The presence/ absence of pocket gopher mounds and small mammal burrows (openings $\geq$ 2.0 cm) was recorded as indicators of natural disturbance and small mammal activity for each vegetation microplot in the study. Presence/ absence of these features was recorded for each vegetation microplot in the study. Pearson's product-moment correlation was used to examine possible relationships between cheatgrass abundance and mean frequency of these factors both individually and combined.

Data Analyses and Presentation. In the presentation of findings that follows, mean percent cover values for all vegetation plots are combined to give an overall mean value. By design, the 5m x 20m Biodiversity plots do not yield frequency data, so that frequency analyses were limited to SIT and BRMP transect microplots. In addition to overall percent cover, percent cover values from transects only are typically presented alongside frequency values to allow for direct comparison of these two parameters. The 20 cm x 50 cm quadrat size used along BRMP and SIT transects may allow for more precise cover estimates in microplots compared to the larger biodiversity plots.

Between-year differences within species or species groups were tested for

Table 1.1. Land-use activities potentially associated with the proliferation of cheatgrass (*Bromus tectorum*) on the ALE Reserve. See text for complete explanation.

Category	**Activity**	**Period**	**Impacts**
Agriculture & Development	Homesteading	1880-1943	Soil and vegetation disturbance around homestead sites
	Crop agriculture	1880-1943	Soil and vegetation distrubance associated with cultivation
	Livestock grazing	1880-1967	Biomass removal; changes in plant community composition; removal of microbiotic crust; soil disturbance, compaction and erosion;
	Natural gas exploration & development	1913-1940	Soil and vegetation disturbances around well sites and associated homesteads
Water Features	Natural springs and streams	1880-1967 (or later)	Concentrated use by livestock; hydrological changes; soil and vegetation disturbance associated with development projects
	Water developments[1]		
Corridors	Road building and maintenance	1880-present	Perpetual disturbance along linear features that act as corridors and refugia for invasive plant species
	Powerline development & maintenance	ca. 1950 - present	
Facilities	Military facilities	1950-1960	Soil and vegetation disturbance associated with heavy construction and operations
	Scientific and communications facilities	ca. 1950 - present	Development and maintenance of facilities along Rattlesnake Summit, at Nike site post-1960, at Rattlesnake Spring, and at groundwater test wells in Cold Creek Valley
Wildfire & fire suppression		Ca. 1950 - present	Biomass removal & plant death; changes in plant community structure, composition, nutrient regimes, etc.; removal of microbiotic crust; soil disturbance, erosion.

1 – Features include water tanks, cisterns, & wells.

significance using single factor repeated measures analysis of variance (ANOVA). Following significant results from ANOVA, 2004 values were tested against pre-fire values using paired-sample *t* tests. Correlations between vegetation data and community and environmental factors were explored using Pearson's product-moment correlation. The significance of correlations greater than $r = 0.30$ was tested using simple regression (Zar 1996, Elzinga et al. n.d.). Since data on percent cover of litter was not collected for Biodiversity Plots until 2004, only that year's data were used for all correlation analyses with vegetation plots.

Multivariate analyses. Two methods of multivariate ecological analyses were used to compare the most recent (2004) vegetation data of ALE study plots with pre-fire records. Both of these methods provide a measure of plot similarity between the selected sample years and highlight potential differences in fire effects and recovery trajectories between plots and community types.

To enhance comparability between years, only perennial plant species data were used in these analyses. Perennial plant species tend to be the more conspicuous and, as a group, were probably more accurately and consistently identified and documented by the multiple observers and methodologies employed over the course of this study. Analyses of sample plot data indicate that the methodology used in sampling Biodiversity Plots in 1994 tended towards underrepresentation of both native and non-native annual forbs, as well as the native annual grass *Festuca microstachys*, in pre-fire data. Early spring ephemerals and warm season annuals also present problems in terms of accurate representation in data collected during the April to mid-July sampling period used in this study. Restricting the analyses to perennial species data reduced these potential sources of error and focused attention on native climax species as indicators of the status and trajectories of plant communities affected by the 24 Command Fire.

Compositional similarity of vegetation plots between pre-fire years and 2004 was compared using Sorensen's coefficient of similarity (***Sc***; McCune and Grace 2002, Kent and Coker 1992). ***Sc*** was calculated using quantitative data pairing each plot's pre-fire composition against the composition recorded in 2004.

Ordination is a powerful tool for summarizing and revealing patterns in complex data sets. Post-fire vegetation dynamics on the ALE Reserve were examined using the community ordination method detrended correspondence analysis (DCA). DCA is a weighted averaging ordination technique that uses reciprocal averaging to summarize species and sample data in multi-dimensional ordination space (Hill and Gauch 1980). DCA is an indirect method of gradient analysis to which environmental information is applied informally in the interpretation of the community gradients revealed by the ordination. Ordination axes are scaled so that each unit along the principal axes represents one half-change in species diversity.

Both pre-fire and 2004 perennial species data from all 70 ALE vegetation plots were used in DCA, yielding a total data set of 140 samples. Runs of DCA were made using default settings except in the following instances. Species data were log-transformed to normalize species abundance curves. Rare species contribute little information but a great deal of noise to gradient analyses (del Moral, Titus, and Cook 1995); therefore, species occurring in $\leq$ four sites were removed from the analyses.

Uncommon species (those with 5-10 occurrences in the data set) were downweighted to reduce but not entirely remove their influence.

Effect of restoration efforts on vegetation studies. Rehabilitation efforts were begun in a portion of the study area between the 2002 and 2003 field seasons. Although these efforts have the potential to affect vegetation patterns within the project area, analyses of results indicate little effect on cover values or other vegetation parameters during the time period covered by this study (Section II this volume). Sagebrush plantings were kept away from the vicinity of reference plots. Percent cover of newly emerging seedlings was negligible during 2003 and 2004, as is to be expected at this stage. Abundance of invasive species was statistically similar between treated and untreated areas. The one area where rehabilitation effects could have made a significant impact on vegetation data was in terms of percent frequency, where a single emerged seedling, however small, counts as a record. In order to present a picture of natural recovery following the 24 Command Fire that was not complicated by the early impact of rehabilitation efforts, frequency records that were clearly contributions of the restoration effort were not included in the analyses for this section. The authors believe that the data presented here through 2004 reliably represent natural conditions of unaided recovery on the ALE Reserve.

Scientific Nomenclature. A list of scientific and common names for vascular plant species referred to in the text is presented in Table 1.2. Botanical nomenclature follows Hitchcock & Cronquist (1973). Since the publication of this regional flora, taxonomic and nomenclatural revisions have been applied to many plant species, particularly graminoids. A table of synonyms following Kartesz (1999) is presented in Appendix C.

Table 1.2. Scientific and common names of vascular plant species referred to in the text. Updated nomenclature based on Kartesz (1999). * = introduced; A = annual; B = biennial; P = Perennial. Boldface indicates species with nomenclatural changes since Hitchcock and Cronquist (1973).

Common Name	Hitchcock and Cronquist (1973)	Updated nomenclature	
Shrubs			
spiny hopsage	*Atriplex spinosa*	***Grayia spinosa***	P
Wyoming big sagebrush	*Artemisia tridentata* ssp. *wyomingensis*	*Artemisia tridentata* ssp. *wyomingensis*	P
threetip sagebrush	*Artemisia tripartita*	*Artemisia tripartita*	P
gray rabbitbrush	*Chrysothamnus nauseosus*	***Ericameria nauseosa***	P
green rabbitbrush	*Chrysothamnus viscidiflorus*	*Chrysothamnus viscidiflorus*	P
winterfat	*Eurotia lanata*	***Krascheninnikovia lanata***	P
bitterbrush	*Purshia tridentata*	*Purshia tridentata*	P
Hemishrubs			
low pussytoes	*Antennaria dimorpha*	*Antennaria dimorpha*	P
threadleaf daisy	*Erigeron filifolius*	*Erigeron filifolius*	P
Piper's daisy	*Erigeron piperianus*	*Erigeron piperianus*	P
parsnip-flowered buckwheat	*Eriogonum heracleoides*	*Eriogonum heracleoides*	P
rock buckwheat	*Eriogonum sphaerocephalum*	*Eriogonum sphaerocephalum*	P
strict desert buckwheat	*Eriogonum strictum*	*Eriogonum strictum*	P
thymeleaf buckwheat	*Eriogonum thymoides*	*Eriogonum thymoides*	P
narrowleaf goldenweed	*Haplopappus stenophyllus*	***Stenotus stenophyllus***	P
cushion phlox	*Phlox hoodii*	*Phlox hoodii*	P
longleaf phlox	*Phlox longifolia*	*Phlox longifolia*	P
Graminoids			
crested wheatgrass *	*Agropyron cristatum*	***Agropyron desertorum***	P
thickspike wheatgrass	*Agropyron dasystachyum*	***Elymus lanceolatus* var. *lanceolatus***	P
bluebunch wheatgrass	*Agropyron spicatum*	***Pseudoroegneria spicata***	P
cheatgrass, downy brome*	*Bromus tectorum*	*Bromus tectorum*	A
Great Basin wildrye	*Elymus cinereus*	***Leymus cinereus***	P
Idaho fescue	*Festuca idahoensis*	*Festuca idahoensis*	P
Indian ricegrass	*Oryzopsis hymenoides*	***Achnatherum hymenoides***	P
bulbous bluegrass*	*Poa bulbosa*	*Poa bulbosa*	P
Cusick's bluegrass	*Poa cusickii*	*Poa cusickii*	P
Sandberg's bluegrass	*Poa sandbergii*	***Poa secunda***	P
winter rye*	*Secale cereale*	*Secale cereale*	AB
squirreltail	*Sitanion hystrix*	***Elymus elymoides***	P
needle-and-thread	*Stipa comata*	***Hesperostipa comata***	P
Thurber's needlegrass	*Stipa thurberiana*	***Achnatherum thurberiana***	P
wheat*	*Triticum aestivum*	*Triticum aestivum*	A

(Continued)

Table 1.2 (Continued).

Common Name	**Hitchcock and Cronquist (1973)**	**Updated nomenclature**	
Forbs			
yarrow	*Achillea millefolium*	*Achillea millefolium*	P
annual mountain dandelion	*Agoseris heterophylla*	*Agoseris heterophylla*	A
tarweed, fiddleneck	*Amsinckia tessellata*	*Amsinckia tessellata*	A
buckwheat milkvetch	*Astragalus caricinus*	*Astragalus caricinus*	P
Columbia milkvetch	*Astragalus columbianus*	*Astragalus columbianus*	P
basalt milkvetch	*Astragalus conjunctus*	*Astragalus conjunctus* var. *rickardii*	P
Spalding’s milkvetch	*Astragalus spaldingii*	*Astragalus spaldingii*	P
Carey's balsamroot	*Balsamorhiza careyana*	*Balsamorhiza careyana*	P
diffuse knapweed*	*Centaurea diffusa*	*Centaurea diffusa*	A
Russian knapweed*	*Centaurea repens*	***Acroptilon repens***	P
Canada thistle*	*Cirsium arvense*	*Cirsium arvense*	P
goosefoot chenopodium	*Chenopodium leptophyllum*	*Chenopodium leptophyllum*	A
rush skeletonweed*	*no record	***Chondrilla juncea***	P
slender hawksbeard	*Crepis atribarba*	*Crepis atribarba*	P
winged cryptantha	*Cryptantha pterocarya*	*Cryptantha pterocarya*	A
cymopterus	*Cymopterus terebinthinus*	*Cymopterus terebinthinus*	P
tansy mustard	*Descurainia pinnata*	*Descurainia pinnata*	A
flixweed tansy mustard*	*Descurainia sophia*	*Descurainia sophia*	A
spring whitlowgrass*	*Draba verna*	*Draba verna*	A
tall willowherb	*Epilobium paniculatum*	***Epilobium brachycarpum***	A
redstem filaree*	*Erodium cicutarium*	*Erodium cicutarium*	A
rough chickweed*	*Holosteum umbellatum*	*Holosteum umbellatum*	A
prickly lettuce*	*Lactuca serriola*	*Lactuca serriola*	A
Gray's lomatium	*Lomatium grayi*	*Lomatium grayi*	P
biscuit root	*Lomatium macrocarpum*	*Lomatium macrocarpum*	P
spurred lupine	*Lupinus laxiflorus*	*Lupinus arbustus*	P
velvet lupine	*Lupinus leucophyllus*	*Lupinus leucophyllus*	P
sulfur lupine	*Lupinus sulphureus*	*Lupinus bingenensis*	P
hoary aster	*Machaeranthera canescens*	*Machaeranthera canescens*	P
Evening primrose	*Oenothera pallida*	*Oenothera pallida*	P
whiteleaf phacelia	*Phacelia hastata*	*Phacelia hastata*	P
narrowleaf phacelia	*Phacelia linearis*	*Phacelia linearis*	A
Russian thistle*	*Salsola kali*	***Salsola tragus***	A
tumble mustard*	*Sisymbrium altissimum*	*Sisymbrium altissimum*	A
meadow salsify*	*Tragopogon dubius*	*Tragopogon dubius*	AB

RESULTS

Environmental Measurements. Environmental variables associated with ALE Reserve study plots are presented in Appendix B. Elevations of vegetation plots ranged from 560' (171 m) to 3400' (1037 m). Slopes tended to be gentlest (generally $0^o – 5^o$) at lower elevations, becoming steeper with increasing elevation, up to a maximum of 30^o within vegetation plots. Owing to the SE to NW orientation of Rattlesnake Mountain, 65 of 70 study plots were oriented between 312^o (NW) and 110^o (ESE) aspects. Heat Load Index (HLI) varied between 0.510 and 0.922 out of a potential range of zero to 1.000. HLI tended to vary inversely with elevation, with the highest values occurring at lower elevations.

Silt loam soils predominated in ALE study plots, characterizing 54 of 70 plots, mostly at lower and middle elevations. Sandy soils characterized six plots at low elevations (560' – 675'; 170 – 205 m) in the Cold Creek Valley. Stony silt loams occurred primarily on the upper slopes of Rattlesnake Mountain (9 plots). Stony silt loams also occur in the bottoms of some eroded draws at lower elevations; this was represented by just one plot among the samples.

All four states of estimated fire severity, from low (1) to highest severity (4) were represented in the study plots. Category 4 was restricted by convention to shrublands (see Methods). Category 3 was generally associated with sagebrush and winterfat shrublands, although fire severity in several low and mid-elevation grasslands was also estimated to have been in this category.

Elevation was strongly correlated with slope (r = 0.76; P <0.0001), moderately negatively correlated with fire severity (r = -0.49; P <0.0001) and moderately to strongly negatively coordinated with HLI (r = -0.70; P <0.0001; Table 1.3). HLI was also very strongly negatively correlated with slope (r = -0.93; P <0.0001), one of the components of its calculation, which helps to explain the strong negative correlation of HLI with elevation. Slope was moderately negatively correlated with fire severity (r = -0.45; P <0.001), reflecting in part the occurrence of a number of category 4 severity shrublands at lower and middle elevations where gentle slopes prevailed. Other correlations between environmental variables were weak.

Table 1.3. Correlations between environmental variables in vegetation plots on the ALE Reserve, pre-fire-2004. All plots (n = 70). Values accompanied by the following superscripts are significant: a - P < 0.0001; b - P < 0.001.

	Elevation	Fire Severity	Slope	Aspect	Heat Load Index
Elevation	1				
Fire Severity	-0.49^a	1			
Slope	0.76^a	-0.45^b	1		
Aspect	0.04	-0.05	0.02	1	
Heat Load Index	-0.70^a	0.40^b	-0.93^a	0.22	1

Invasive Species

Cheatgrass cover & frequency. Mean percent cover of cheatgrass (*Bromus tectorum*) in the 70 ALE vegetation plots declined by more than 50% in the year following the 24 Command Fire but recovered to pre-fire values by 2002 (Table 1.4a). Cheatgrass cover in 2004 (10.3% ± 11.4 SD) was unchanged from 2003 values and was statistically indistinguishable from pre-fire values ($P = 0.363$). Cheatgrass cover ranged from zero to 44.3% in 2004. Thirty-seven of 70 plots (52.9%) exhibited ≤ 5.0% cover, while 6 plots (8.6%) had ≥ 30.0% cover (Fig. 1.1).

Cheatgrass cover in stands formerly dominated by Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) and winterfat (*Eurotia lanata*) as well as in low and middle elevation perennial grasslands recovered to pre-fire levels no later than 2003; in 2004 these values were statistically indistinguishable from pre-fire values. Cheatgrass cover along transects in former big sagebrush stands in 2004 (11.8% ± 8.6 SD) was still suppressed below pre-fire values (19.2 % ± 19.8 SD; $P = 0.070$). This difference was primarily due to effects observed in sagebrush stands on sandy soils, where cheatgrass cover in 2004 (19.8 ± 5.0 SD) was still only 54.0% of pre-fire values (36.7% ± 4.4 SD; $P = 0.003$). In contrast, percent cover of cheatgrass in big sagebrush and winterfat stands on silt loam soils was more than 60% greater in 2004 (13 3% ± 13.8 SD) compared to pre-fire values (8.2% ± 14.8 SD; $P = 0.163$). Cheatgrass cover in threetip sagebrush (*Artemisia tripartita*)/ Idaho fescue (*Festuca idahoensis*) stands was significantly greater in 2004 (2.8 ± 4.7 SD; $P = 0.064$) compared to pre-fire values (0.8 ± 1.2 SD), but between-year differences were not significant when only transect data were subjected to ANOVA.

Mean percent frequency of cheatgrass in 2004 (68.1% ± 34.2 SD) was significantly greater than pre-fire values (48.5 % ± 42.0 SD; $P < 0.001$; Table 1.4b). Percent frequency in 2004 was similar to 2003 values and was significantly greater than pre-fire values in all major habitat types ($P < 0.05$) except higher elevation stands characterized by threetip sagebrush and Idaho fescue. Cheatgrass was recorded in 68 out of 70 plots (97.1%) in 2004, the same proportion as in 2002 and 2003. Percent frequency ranged from zero to 100% with fully 70% of all transects recording frequencies of 75% or greater (Fig. 1.2). Only one plot (Biodiversity Plot 88, on the high slopes of Rattlesnake Mountain) has not had cheatgrass recorded within it during the course of this study.

Percent cover of cheatgrass from 2002-2004 was strongly correlated ($r = 0.78$ to 0.90; $P < 0.0001$) with percent cover of the preceding year, but percent cover of post-fire years was only weakly to moderately correlated with pre-fire cover (Table 1.5).

Cheatgrass cover was no more than moderately correlated with any of the environmental variables (Table 1.5). Cheatgrass cover was moderately negatively correlated with elevation ($r = -0.44$; $P < 0.001$) and with slope ($r = -0.46$; $P < 0.0001$). Cheatgrass cover was moderately correlated with percent cover of plant litter ($r = 0.49$; $P < 0.0001$) and was somewhat weakly to moderately correlated with heat load ($r = 0.39$; $P < 0.001$). Percent cover of cheatgrass was only weakly or not at all correlated with the remaining environmental variables.

Percent cover of cheatgrass exhibited moderate negative correlations with several plant community variables (Table 1.5). The strongest correlations were between pre-fire cheatgrass cover and pre-fire cover of native perennial grasses (r = -0.66; $P < 0.001$); between cheatgrass cover and percent cover of native perennial plant species, both before the fire and in 2004 (r = -0.62; $P < 0.001$); and between cheatgrass cover and native perennial plant species richness in 2004 (r = -0.55; $P < 0.001$).

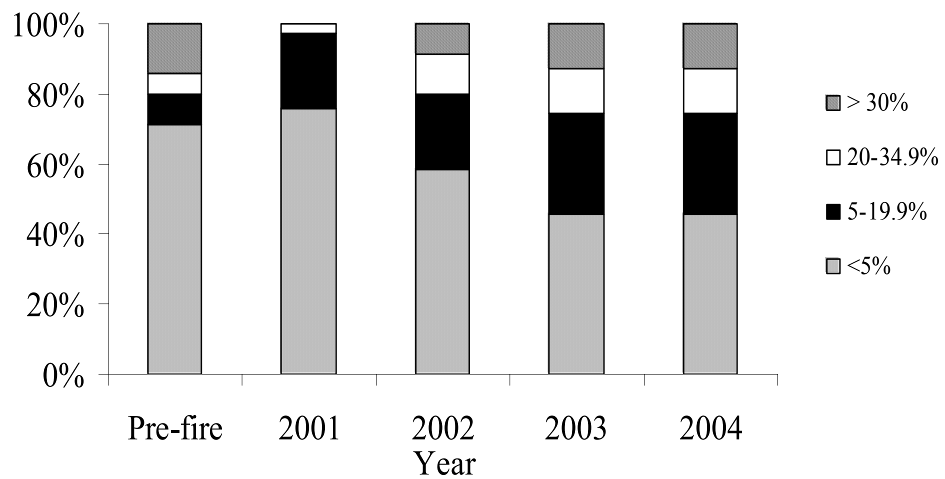


Fig. 1.1. Proportions of vegetation plots (n = 70) in percent cover classes of cheatgrass (*Bromus tectorum*) on the ALE Reserve, pre-fire-2004.

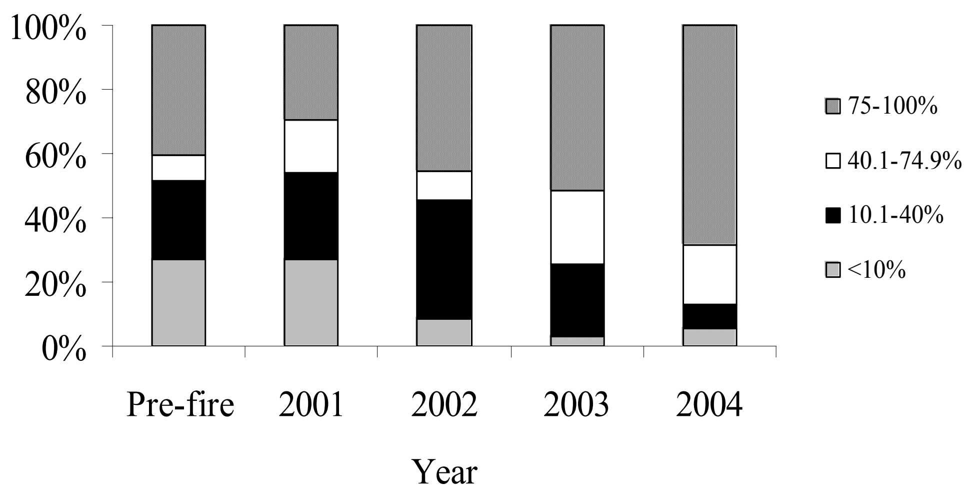


Fig. 1.2. Proportions of vegetation transects (n = 37) in percent frequency classes of cheatgrass (*Bromus tectorum*) on the ALE Reserve, pre-fire-2004.

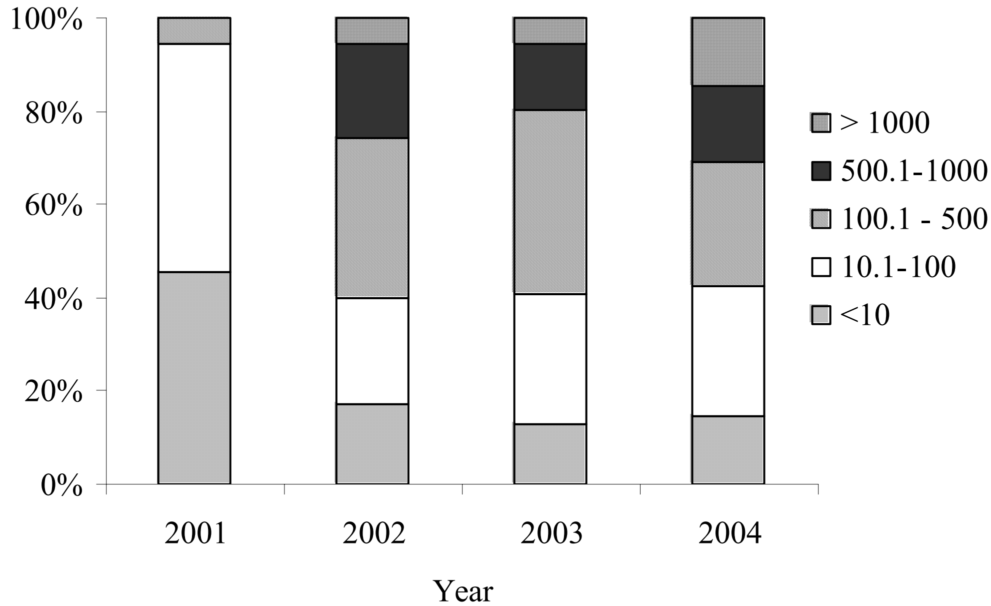


Fig. 1.3. Proportions of vegetation plots in stem density classes of cheatgrass (*Bromus tectorum*) on the ALE Reserve, 2001-2004. Legend units are stems/m^2. n = 70 all years except 2001, when n = 55..

Table 1.4. Mean percent cover (a) and frequency (b) of cheatgrass (*Bromus tectorum*) in ALE Reserve vegetation plots, pre-fire - 2004. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests between pre-fire and 2004 values only.

a.		Percent Cover (± SD)										ANOVA	*t* test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
All Sites	70	8.8	(15.4)	3.7	(4.9)	8.8	(11.0)	10.4	(11.0)	10.3	(11.4)	< 0.0001	0.363
All Transects	37	13.6	(18.7)	3.4	(5.0)	7.7	(10.2)	9.5	(10.8)	10.4	(10.8)	< 0.0001	0.226
Big Sagebrush (and Winterfat) Stands													
All Sites	20	15.3	(18.0)	2.9	(3.5)	11.0	(13.0)	12.6	(11.1)	14.9	(12.4)	< 0.0001	0.928
All Transects	12	19.2	(19.8)	2.1	(3.2)	8.4	(11.5)	9.1	(9.0)	11.8	(8.6)	< 0.0001	0.070
Silt Loam Soils	15	8.2	(14.8)	2.7	(3.5)	9.8	(12.5)	11.7	(11.7)	13.3	(13.8)	0.002	0.163
Sandy Soils	5	36.7	(4.4)	3.3	(3.9)	14.4	(15.4)	15.2	(9.6)	19.8	(5.0)	< 0.0001	0.003
Grasslands													
All Sites	33	8.9	(16.0)	5.2	(5.8)	10.4	(11.2)	12.6	(11.4)	11.3	(11.5)	0.005	0.406
All transects	17	15.6	(20.1)	5.6	(6.2)	10.3	(10.6)	13.6	(12.4)	13.4	(12.8)	0.041	0.665
Threetip Sagebrush/ Idaho Fescue Stands													
All Sites	17	0.8	(1.2)	1.7	(3.2)	3.1	(5.4)	3.6	(7.3)	2.8	(4.7)	0.034	0.064
All Transects	8	0.9	(1.1)	0.4	(0.4)	1.0	(1.5)	1.3	(1.7)	1.3	(1.7)	0.124	--

Continued

Table 1.4 (continued). Mean percent cover (a) and frequency (b) of cheatgrass (*Bromus tectorum*) in ALE Reserve vegetation plots, pre-fire - 2004. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests between pre-fire and 2004 values only.

b.		Percent Frequency (± SD)										ANOVA	*t* test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
All Transects	37	48.5	(42.0)	47.0	(38.4)	58.6	(35.4)	68.4	(32.8)	68.1	(34.2)	< 0.0001	< 0.001
Big Sagebrush Stands													
All Transects	12	55.0	(49.0)	34.6	(36.8)	60.0	(34.6)	72.9	(26.3)	81.3	(24.5)	< 0.0001	0.007
Silt Loam Soil Transects	7	22.9	(39.0)	20.0	(31.0)	52.9	(31.5)	61.4	(26.7)	67.9	(24.5)	< 0.0001	0.001
Sandy Soil Transects	5	100.0	(0.0)	55.0	(37.4)	70.0	(39.8)	89.0	(16.7)	100.0	(0.0)	0.013	--
Grassland Transects	17	60.3	(37.8)	70.6	(32.5)	75.0	(29.2)	83.2	(26.3)	79.4	(28.5)	0.018	0.049
Threetip/Fescue Transects	8	13.8	(15.8)	15.6	(16.1)	21.9	(20.3)	30.0	(24.3)	24.4	(20.3)	0.013	0.175

Table 1.5. Correlation coefficients of environmental and community variables with percent cover of cheatgrass (*Bromus tectorum*) in ALE Reserve vegetation plots (n = 70), pre-fire -2004. Coefficients accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

		Percent Cover				
		Pre-fire	2001	2002	2003	2004
Cheatgrass Cover	Pre-fire	1				
	2001	0.28	1			
	2002	0.40^{b}	0.78^{a}	1		
	2003	0.40^{b}	0.79^{a}	0.90^{a}	1	
	2004	0.50^{a}	0.70^{a}	0.86^{a}	0.90^{a}	1
Pre-Fire Shrub Cover		0.08	-0.23	-0.07	-0.07	0.10
Litter		0.34^{c}	0.45^{b}	0.37^{c}	0.44^{b}	0.49^{a}
Elevation		-0.31^{d}	-0.23	-0.37^{c}	-0.43^{b}	-0.44^{b}
Fire		0.25	-0.08	0.15	0.12	0.28
Slope		-0.31^{e}	-0.24	-0.37^{c}	-0.44^{b}	-0.46
Aspect		-0.08	-0.14	-0.16	-0.06	-0.07
Heat Load Index		0.26	0.20	0.30^{e}	0.39^{b}	0.39^{b}
Native Perennial Cover	Pre-fire	-0.62^{a}	-0.29	-0.40^{b}	-0.47^{a}	-0.46^{a}
	2001	-0.38^{c}	-0.20	-0.38^{c}	-0.40^{b}	-0.48^{a}
	2002	-0.47^{a}	-0.18	-0.41^{b}	-0.44^{b}	-0.56^{a}
	2003	-0.48^{a}	-0.32^{c}	-0.48^{a}	-0.52^{a}	-0.61^{a}
	2004	-0.51^{a}	-0.30^{e}	-0.45^{a}	-0.51^{a}	-0.62^{a}
Native Perennial Grass Cover	Pre-fire	-0.66^{a}	-0.16	-0.30	-0.34^{c}	-0.39^{b}
	2001	-0.37^{c}	-0.23	-0.34^{c}	-0.34^{c}	-0.42^{b}
	2002	-0.42^{b}	-0.10	-0.27	-0.27	-0.40^{b}
	2003	-0.42^{b}	-0.23	-0.36^{c}	-0.41^{b}	-0.50^{a}
	2004	-0.43^{b}	-0.19	-0.32^{c}	-0.38^{c}	-0.47^{a}
Native Perennial Warm Season Grass Cover	Pre-fire	-0.48^{a}	-0.17	-0.30^{e}	-0.34^{c}	-0.41^{b}
	2001	-0.32^{c}	-0.23	-0.35^{c}	-0.38^{c}	-0.42^{b}
	2002	-0.36^{c}	-0.25	-0.38^{c}	-0.44^{b}	-0.50^{a}
	2003	-0.35^{c}	-0.26	-0.40^{b}	-0.43^{b}	-0.51^{a}
	2004	-0.34^{c}	-0.19	-0.34^{c}	-0.41^{b}	-0.48^{a}
Native Perennial Richness	Pre-fire	-0.43^{b}	-0.18	-0.37^{c}	-0.42^{b}	-0.51^{a}
	2001	-0.42^{b}	-0.23	-0.41^{b}	-0.45^{a}	-0.56^{a}
	2002	-0.41^{b}	-0.24	-0.39^{b}	-0.45^{a}	-0.53^{a}
	2003	-0.34^{c}	-0.23	-0.38^{c}	-0.43^{b}	-0.53^{a}
	2004	-0.35^{c}	-0.30^{e}	-0.44^{b}	-0.49^{a}	-0.55^{a}

Table 1.6. Correlation coefficients of environmental and community variables with percent frequency of cheatgrass (*Bromus tectorum*) in ALE Reserve vegetation transects (n = 37), pre-fire -2004. Coefficients accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

		Percent Frequency				
		Pre-fire	2001	2002	2003	2004
Cheatgrass Frequency	Pre-fire	1				
	2001	0.64[a]	1			
	2002	0.60[a]	0.80[a]	1		
	2003	0.66[a]	0.73[a]	0.88[a]	1	
	2004	0.69[a]	0.69[a]	0.86[a]	0.93[a]	1
Cheatgrass Cover	Pre-fire	0.79[a]	0.37[e]	0.46[c]	0.52[b]	0.59[b]
	2001	0.37[e]	0.69[a]	0.50[c]	0.49[c]	0.47[c]
	2002	0.48[c]	0.66[a]	0.63[a]	0.59[b]	0.58[b]
	2003	0.54[b]	0.68[a]	0.66[a]	0.68[a]	0.68[a]
	2004	0.54[b]	0.53[b]	0.58[b]	0.66[a]	0.69[a]
Pre-Fire Shrub Cover		-0.19	-0.49[c]	-0.31	-0.23	-0.08
	Litter	0.32	0.56[b]	0.52[c]	0.48[c]	0.44[d]
	Elevation	-0.49[c]	-0.37[e]	-0.55[b]	-0.66[a]	-0.75[a]
	Fire	0.13	-0.16	0.05	0.12	0.28
	Soil	0.44[d]	0.05	0.02	0.08	0.18
	Slope	-0.39[e]	-0.37[e]	-0.50[c]	-0.60[a]	-0.70[a]
	Aspect	0.13	0.07	-0.03	-0.06	-0.16
	Heat Load Index	0.43[d]	0.39[e]	0.52[c]	0.59[b]	0.66[a]
Native Perennial Cover	Pre-fire	-0.84[a]	-0.61[a]	-0.61[a]	-0.64[a]	-0.67[a]
	2001	-0.18	0.00	-0.32	-0.40[e]	-0.56[b]
	2002	-0.31	-0.07	-0.37[e]	-0.46[c]	-0.61[a]
	2003	-0.38[e]	-0.18	-0.46[c]	-0.52[b]	-0.68[a]
	2004	-0.46[c]	-0.22	-0.46[c]	-0.52[b]	-0.68[a]
Native Perennial Grass Cover	Pre-fire	-0.85[a]	-0.39[e]	-0.49[c]	-0.56[b]	-0.64[a]
	2001	-0.06	0.15	-0.14	-0.22	-0.37[e]
	2002	-0.20	0.14	-0.12	-0.21	-0.38[e]
	2003	-0.24	0.06	-0.22	-0.28	-0.46[c]
	2004	-0.27	0.07	-0.17	-0.27	-0.42[e]
Native Perennial Warm Season Grass Cover	Pre-fire	-0.61[a]	-0.15	-0.34[e]	-0.40[e]	-0.45[c]
	2001	-0.22	-0.03	-0.29	-0.38[e]	-0.48[c]
	2002	-0.30	-0.07	-0.30	-0.40[e]	-0.49[c]
	2003	-0.17	0.10	-0.19	-0.22	-0.36[e]
	2004	-0.18	0.11	-0.12	-0.21	-0.29
Native Perennial Richness	Pre-fire	-0.38[e]	-0.29	-0.45[d]	-0.55[b]	-0.62[a]
	2001	-0.37[e]	-0.21	-0.45[d]	-0.57[b]	-0.69[a]
	2002	-0.49[c]	-0.41[e]	-0.57[b]	-0.69[a]	-0.78[a]
	2003	-0.48[c]	-0.34[e]	-0.53[b]	-0.62[a]	-0.67[a]
	2004	-0.43[d]	-0.42[d]	-0.64[a]	-0.69[a]	-0.75

Percent frequency of cheatgrass was strongly negatively correlated with elevation (r = -0.75; $P < 0.0001$) and slope (r = -0.70; $P < 0.0001$) in 2004 (Table 1.6). Frequency was moderately correlated with heat load (r = 0.66; $P < 0.0001$) and with litter (r = 0.44; $P < 0.01$). Correlations with other environmental variables were weak.

Percent frequency of cheatgrass was strongly correlated with the previous year's frequency from 2002-2004 (r = 0.80 to 0.93; $P < 0.0001$). Frequency was moderately to strongly correlated within years with percent cover of cheatgrass during all years of the study (r = 0.63 to 0.79; $P < 0.0001$), although the pre-fire correlation was the highest. Frequency from 2002-2004 was also moderately to strongly correlated with percent cover of the previous year (r = 0.66 all years; $P < 0.0001$).

Percent frequency of cheatgrass was strongly negatively correlated with native perennial species richness (r = –0.75; $P < 0.0001$) in 2004. This strength of this correlation has increased each year since 2001. Correlations between cheatgrass frequency and variables of total native perennial cover, total perennial grass cover, and cover of large warm season perennial grasses exhibited somewhat similar patterns of negative correlation. The highest within year correlations were between pre-fire values. Following the wildfire, coefficients were greatly diminished. By 2004 the correlation with native perennial cover was moderately to strongly negative (r = -0.68; $P < 0.0001$) and the correlation with total native perennial grass cover was moderately negative (r = -0.42; $P < 0.05$) while the correlation with large warm season perennial grasses remained weak.

Percent cover of cheatgrass in SIT transects with complete long-term records outside of the ALE Reserve ("Benton plots") ranged between 7.3% (± 13.4 SD) and 8.8% (± 12.3 SD) from 1992 to 2001 (Fig. 1.4), but values were not significantly different between years. (ANOVA, $P = 0.996$; Table 1.7). Cheatgrass cover in SIT transects on the ALE Reserve closely paralleled that of the Benton plots from 1992 through 1997. In 2000, just prior to the 24 Command Fire, percent cover increased sharply (16.3% ± 17.9 SD) compared both to the previous year on ALE (8.6% ± 15.1 SD) and compared to Benton plots in 2000 (8.8% ± 7.3 SD), although neither difference was significant ($P > 0.200$). Cheatgrass cover on ALE fell to its lowest values (4.6% ± 4.7 SD) in 2001 following the fire. Again, this figure was not significantly different from other years' values on the ALE Reserve ($P = 0.317$) or from cover in Benton plots during the same year ($P = 0.114$)

ANOVA indicated no significant between-year differences in percent frequency of cheatgrass in Benton plots ($P = 0.343$). Although ANOVA did indicate a significant difference between years for plots on the ALE Reserve ($P = 0.002$), no difference was found by a Scheffé multiple comparisons test ($\alpha = 0.05$).

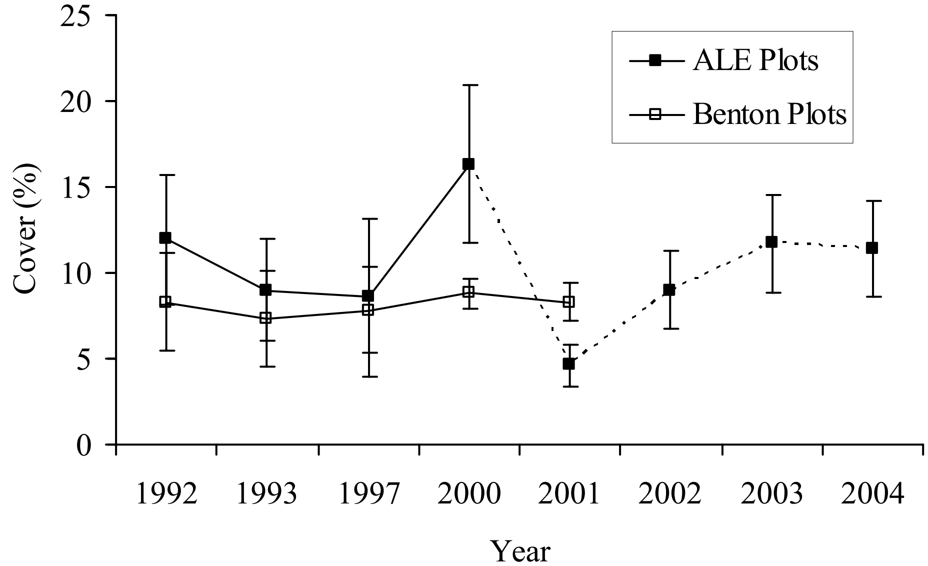


Fig. 1.4. Percent over of cheatgrass (*Bromus tectorum*) in Steppe-in-Time plots on the ALE Reserve ("ALE Plots") and elsewhere in Benton County ("Benton Plots"), 1992-2004. No data was available for Benton plots after 2001. Bars indicate ± one standard error. Dashed line indicates trend of ALE plots following the 24 Command Fire. There were no significant differences within years between the two groups.

Table 1.7. Percent Cover and frequency of cheatgrass (*Bromus tectorum*) in Steppe-in-Time transects on the ALE Reserve ('ALE Transects') and elsewhere in Benton County ('Benton Transects'), 1992-2004. ANOVA *P* values are the results of single factor ANOVA; within-year *P* values are the results of two-tailed independent sample *t* tests. n.d. = no data.

	1992	1993	1997	2000	2001	2002	2003	2004	ANOVA *P*
Cover									
ALE Transects	12.0 (13.0)	9.0 (10.4)	8.6 (15.1)	16.3 (17.9)	4.6 (4.7)	9.0 (9.1)	11.7 (11.1)	11.4 (10.9)	0.317
Benton Transects	8.3 (13.7)	7.3 (13.4)	7.8 (12.0)	8.8 (12.3)	8.3 (12.4)	--n.d.--	--n.d.--	--n.d.--	0.996
P value, within years	0.449	0.687	0.891	0.206	0.114	--n.d.--	--n.d.--	--n.d.--	
Frequency									
ALE Transects	40.0 (31.0)	37.5 (32.8)	38.2 (30.9)	70.0 (31.6)	65.0 (37.9)	72.3 (32.2)	72.3 (32.2)	72.3 (32.2)	0.002
Benton Transects	44.3 (35.1)	34.3 (38.5)	36.5 (37.2)	54.8 (34.6)	43.4 (33.9)	--n.d.--	--n.d.--	--n.d.--	0.343
P value, within years	0.710	0.802	0.892	0.297	0.499	--n.d.--	--n.d.--	--n.d.--	
Number of Plots									
ALE Transects	12	12	11	15	15	15	15	15	
Benton Transects	23	23	23	23	16	--n.d.--	--n.d.--	--n.d.--	

Cheatgrass density. Values for cheatgrass density in burned areas were at their lowest in 2001 following the 24 Command Fire in all types of plots (Table 1.8, Table 1.9). Density in burned plots increased dramatically in 2002 but densities in different plot types exhibited somewhat differing trends in 2003 and 2004.

Mean density of cheatgrass in all vegetation plots was statistically higher in 2004 (376.1 stems/m^2 ± 479.3 SD) compared to 2003 (282.5 stems/m^2 ± 338.4 SD; P = 0.002; Table 1.8). This difference was due to large increases in density in sagebrush and winterfat plots, while density in other habitat types remained similar between years. Density ranged from zero to 1911.7 stems/m^2, with ten or fewer stems/m^2 being recorded in ten out of 70 samples and more than 1,000 stems/m^2 recorded in ten plots at the other extreme (Fig. 1.3). In contrast to vegetation plots, cheatgrass density in transition density plots in 2004 (147.9 stems/m^2 ± 104.1 SD) was statistically similar to 2003 values (142.3 ± 93.6 SD; P =0.653).

In 2001, the first growing season following the fire, cheatgrass density in burned areas adjacent to unburned refugia was lower (164.3 stems/m^2 ± 185.1 SD) than density in the refugia themselves (279.4 stems/m^2 ± 236.9 SD; P = 0.097 Table 1.9), suggesting that the wildfire reduced density by more than 40.0%. From 2002 to 2004 density within burned areas was numerically greater than in refugia, although the difference has not always been statistically significant. The difference between burned areas (487.2 stems/m^2 ± 334.7) and refugia (389.8 stems/m^2 ± 388.6) was not significant in 2004 (P = 0.112).

Cheatgrass density in refugia fluctuated between 262.8 stems/m^2 (± 265.8 SD) and 459.9 stems/m^2 ± 358.1 SD) in the years since the 24 Command Fire, but values have remained statistically similar (P = 0.169). Density in adjacent burned areas was lowest in 2001 (164.3 stems/m^2 ± 185.1 SD) and highest in 2002 (562.2 stems/m^2 ± 410.3; P < 0.0001). Values in 2003 and 2004 were in the intermediate to higher parts of this range but were not significantly different from either extreme (Table 1.9).

Cheatgrass density in vegetation plots was strongly correlated with percent cover of cheatgrass (r = 0.67 to 0.95; P < 0.0001; Table 1.10) both within and between years following the 24 Command Fire. The strongest correlation between these two factors was observed in 2004. Correlation between density and pre-fire cover of cheatgrass tended to be weak to moderate. Cheatgrass density in 2004 was moderately correlated with litter (r = 0.46; P < 0.0001) and moderately negatively correlated with slope (r = -0.41; P < 0.001). Correlations with other environmental variables were weak. Density was also moderately negatively correlated with native perennial species richness (r = -0.54; P < 0.0001), percent cover of native perennials (r = -0.50; P < 0.0001), and percent cover of large warm season perennial grasses (r = -0.46; P < 0.0001).

Cheatgrass density in burned over big sagebrush and winterfat stands was significantly greater in 2004 (547.9 stems/m^2 ± 582.0 SD) than in 2003 (297.7 stems/m^2 ± 399.2 SD; P = 0.002). In contrast, density in all other plots was statistically unchanged between 2003 and 2004 (P = 0.203; Table 1.8)

Table 1.8. Mean densities of cheatgrass (*Bromus tectorum)* in vegetation and density plots on the ALE Reserve, 2001-2004. ANOVA *P* values are the result of standard single factor ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests applied between 2004 and 2003 data only.

	Cheatgrass density (stems / m^2 ± SD)				ANOVA	*t* test
	2001	2002	2003	2004	***P***	***P***
Vegetation Plots						
All	28.2 (38.9)	314.8 (396.2)	282.5 (338.4)	376.1 (479.3)	<0.0001	0.002
n	55	70	70	70		
Sagebrush & winterfat plots	24.4 (39.8)	262.3 (381.5)	297.7 (399.2)	547.9 (582.0)	<0.0001	0.002
n	18	20	20	20		
All Other Plots	30.1 (38.9)	329.3 (410.4)	275.2 (324.5)	307.4 (418.9)	<0.001	0.203
n	37	50	50	50		
Transition Density Plots	10.5 (9.0)	76.7 (52.8)	142.3 (93.6)	147.9 (104.1)	<0.0001	0.653
n	17	17	17	17		

Table 1.9. Mean densities of cheatgrass (*Bromus tectorum*) in unburned refugia and adjacent burned areas on the ALE Reserve, 2001-2004. Between-year *P* values are results of standard single factor ANOVA; values with the same superscript are significantly different (Tukey 0.05). Burn v. Refugia *P* values are the result of paired-sample *t* tests.

		Cheatgrass density (stems / m^2 ± SD)		
	n	**Refugia**	**Burned Areas**	*P* value Burn v. Refugia
2001	19	279.4 (236.9)	164.3[a] (185.1)	0.097
2002	17	475.9 (358.1)	562.2[a] (410.3)	0.371
2003	17	262.8 (265.8)	395.3 (228.6)	0.008
2004	17	389.8 (388.6)	487.2 (334.7)	0.112
P value - Between Years		0.169	0.001	

Table 1.10. Correlation coefficients of environmental and community factors with stem density of cheatgrass (*Bromus tectorum*) in ALE Reserve vegetation plots, 2001 -2004. Coefficients accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

		Cheatgrass Density			
		2001	2002	2003	2004
Cheatgrass Density	2001	1			
	2002	0.88^{a}	1		
	2003	0.76^{a}	0.84^{a}	1	
	2004	0.71^{a}	0.71^{a}	0.88^{a}	1
Cheatgrass Cover	Pre-fire	0.23	0.36	0.31	0.45^{a}
	2001	0.67^{a}	0.82^{a}	0.78^{a}	0.67^{a}
	2002	0.79^{a}	0.83^{a}	0.86^{a}	0.83^{a}
	2003	0.73^{a}	0.85^{a}	0.91^{a}	0.85^{a}
	2004	0.66^{a}	0.73^{a}	0.85^{a}	0.95^{a}
Pre-Fire Shrub Cover		-0.25	-0.23	-0.11	0.07
	Litter	0.22	0.52^{a}	0.44	0.46^{a}
	Elevation	-0.40^{c}	-0.30^{e}	-0.37^{c}	-0.39^{b}
	Fire	0.02	0.00	0.07	0.25
	Slope	-0.33^{e}	-0.31^{d}	-0.37^{c}	-0.41^{b}
	Aspect	-0.04	-0.11	-0.08	-0.09
	Heat Load Index	0.30^{e}	0.27	0.32^{d}	0.35^{c}
Native Perennial Cover	Pre-fire	-0.34^{e}	-0.42^{b}	-0.34^{c}	-0.37^{c}
	2001	-0.23	-0.28	-0.29	-0.39^{b}
	2002	-0.26	-0.29	-0.32^{d}	-0.48^{a}
	2003	-0.31^{e}	-0.37^{c}	-0.41^{b}	-0.54^{a}
	2004	-0.25	-0.39^{b}	-0.37^{c}	-0.50^{a}
Native Perennial Grass Cover	Pre-fire	-0.09	-0.25	-0.21	-0.31^{d}
	2001	-0.12	-0.23	-0.22	-0.32^{d}
	2002	-0.11	-0.13	-0.13	-0.16
	2003	-0.16	-0.26	-0.28	-0.42^{e}
	2004	-0.09	-0.27	-0.22	-0.36^{c}
Native Perennial Warm Season Grass Cover	Pre-fire	-0.15	-0.22	-0.28	-0.37^{c}
	2001	-0.27	-0.30^{e}	-0.36^{c}	-0.39^{b}
	2002	-0.27	-0.35^{c}	-0.40^{b}	-0.46^{a}
	2003	-0.31^{e}	-0.30^{e}	-0.39^{b}	-0.49^{a}
	2004	-0.22	-0.31^{d}	-0.37^{c}	-0.46^{a}
Native Perennial Richness	Pre-fire	-0.37^{d}	-0.30^{e}	-0.35^{c}	-0.50^{a}
	2001	-0.32^{e}	-0.26	-0.36^{c}	-0.51^{a}
	2002	-0.37^{d}	-0.33^{d}	-0.39^{b}	-0.50^{a}
	2003	-0.37^{d}	-0.34^{c}	-0.41^{b}	-0.52^{a}
	2004	-0.45^{b}	-0.37^{c}	-0.45^{b}	-0.54^{a}

Table 1.11. Correlation coefficients of microbiotic crust (MBC) cover with abundance measures of cheatgrass (*Bromus tectorum*) in ALE Reserve vegetation plots, pre-fire - 2004. Coefficients accompanied by the following superscripts are significant: a - $P <$ 0.0001; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

		Percent Cover MBC				
		Pre-fire	2001	2002	2003	2004
Cheatgrass Cover	Pre-fire	-0.40 [b]	-0.22	-0.29	-0.27	-0.21
	2001	-0.24	-0.14	-0.16	-0.20	-0.20
	2002	-0.27	-0.18	-0.23	-0.25	0.28
	2003	-0.34 [c]	-0.20	-0.28	-0.29	-0.29
	2004	-0.28	-0.29	-0.35 [c]	-0.36 [c]	-0.33 [d]
Cheatgrass Frequency	Pre-fire	-0.72 [a]	-0.16	-0.17	-0.08	-0.04
	2001	-0.36 [e]	0.18	0.11	0.05	0.07
	2002	-0.39 [e]	0.03	-0.06	-0.12	-0.07
	2003	-0.43 [d]	-0.06	-0.17	-0.24	-0.14
	2004	-0.41 [e]	-0.17	-0.29	-0.34 [e]	-0.25
Cheatgrass Density	2001	-0.13	0.16	0.09	-0.01	-0.07
	2002	-0.28	-0.11	-0.15	-0.17	-0.16
	2003	-0.29	-0.07	-0.10	-0.18	-0.20
	2004	-0.23	-0.19	-0.24	-0.32 [d]	-0.29

Relationship of cheatgrass abundance to microbiotic crust cover. Pre-fire percent cover of microbiotic crust (MBC) was strongly negatively correlated with pre-fire percent frequency of cheatgrass ($r = -0.72$; $P < 0.0001$). Some relationship between pre-fire MBC and cheatgrass frequency was also apparent from 2001 through 2004, although the correlations were weaker (Table 1.11). Pre-fire cover of MBC was also moderately negatively correlated with pre-fire percent cover of cheatgrass ($r = -0.40$; $P < 0.001$) but the two factors were only weakly correlated in subsequent years. Other correlations between cheatgrass abundance and MBC cover were only weakly or not at all correlated.

Relationship of cheatgrass abundance to disturbance and land use factors. Cheatgrass abundance was generally poorly correlated with land use and disturbance measures recorded by this study (Table 1.12). The only moderate correlations were between percent frequency of cheatgrass from 2001 through 2003 and both number of fires since 1974 and number of years since last wildfire before 2000 ($r = 0.40$ to 0.57; $P <$ 0.05). Correlations between both pre-fire and 2004 frequency were weak, however. All other correlations between cheatgrass abundance and disturbance and land use factors were only weakly or not at all correlated.

Table 1.12. Correlation coefficients of disturbance and land use factors with abundance measures of cheatgrass (*Bromus tectorum*) in ALE Reserve vegetation plots, pre-fire - 2004. Coefficients accompanied by the following superscripts are significant: b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

a. Cheatgrass Cover		Pre-fire	2001	2002	2003	2004
Frequency	Animal Mounds	-0.03	-0.16	-0.16	-0.16	-0.13
	Animal Holes	0.25	-0.04	0.13	0.06	0.00
	Mounds + Holes	0.07	-0.15	-0.07	-0.11	-0.13
Distance	Active Road	-0.20	0.02	0.22	0.10	0.03
	Any Road[1]	-0.20	0.10	0.24	0.14	0.12
	Any Corridor[2]	-0.23	0.11	0.25	0.15	0.11
	Active Points[3]	-0.16	0.22	0.17	0.24	0.09
	Any Point[4]	0.00	0.02	0.04	0.05	0.01
	Any Water Feature[5]	-0.10	0.20	0.26	0.34[c]	0.27
Fire	Number Since 1974	-0.12	0.23	0.16	0.18	0.00
	Years Since Last Fire	-0.12	0.31[e]	0.15	0.22	0.03

b. Cheatgrass Frequency		Pre-fire	2001	2002	2003	2004
Frequency	Animal Mounds	-0.22	-0.22	-0.12	-0.17	-0.21
	Animal Holes	0.00	0.00	0.15	0.10	0.24
	Mounds + Holes	-0.16	-0.20	-0.09	-0.15	-0.11
Distance	Active Road	-0.27	-0.18	-0.13	-0.17	-0.27
	Any Road[1]	-0.20	-0.11	0.06	-0.02	-0.04
	Any Corridor[2]	-0.29	-0.16	-0.05	-0.03	-0.09
	Active Points[3]	-0.17	0.20	0.14	0.24	0.17
	Any Point[4]	-0.08	0.00	-0.02	0.07	0.00
	Any Water Feature[5]	-0.03	0.15	0.12	0.13	0.13
Fire	Number Since 1974	0.16	0.54[b]	0.40[e]	0.46[c]	0.34[e]
	Years Since Last Fire	0.25	0.57[b]	0.41[e]	0.43[d]	0.29

c. Cheatgrass Density		2001	2002	2003	2004
Frequency	Animal Mounds	-0.21	-0.17	-0.13	-0.09
	Animal Holes	0.20	0.05	0.02	0.02
	Mounds + Holes	-0.06	-0.12	-0.11	-0.07
Distance	Active Road	0.31 [e]	0.03	0.19	0.05
	Any Road[1]	0.36 [d]	0.05	0.22	0.13
	Any Corridor[2]	0.36 [d]	0.06	0.23	0.12
	Active Points[3]	0.17	0.24	0.24	0.08
	Any Point[4]	0.03	0.08	0.10	0.01
	Any Water Feature[5]	0.28	0.25	0.33[d]	0.21
Fire	Number Since 1974	0.24	0.22	0.24	0.06
	Years Since Last Fire	0.22	0.28	0.23	0.02

1 – Nearest road whether currently active or abandoned. 2 – Nearest road or powerline corridor.

3 – Points include scientific, military, communication, or maintenance facilities, homesteads, natural gas and groundwater test wells,, gravel pits, and other point disturbances.

4 – Any point whether currently active or abandoned. 5 – Springs, tanks, cisterns, or wells.

Alien Forbs. Alien forbs were recorded in 68 of 70 vegetation plots in 2004. With the exception of a single perennial forb species (*Cirsium arvense*, BRMaP 20/ PC1) in 2002, all alien forbs recorded in vegetation plots during the course of this study were annuals or biennials (Table 1.13).

Mean percent cover and frequency as well as species richness of alien forbs tended to be greatest in 2003 regardless of plot type or habitat type. Values for all three parameters declined in 2004. Both percent cover and percent frequency of alien forbs in 2004 were statistically similar to pre-fire values ($P > 0.250$) except in big sagebrush and winterfat plots (Table 1.14a, b). Overall alien forb richness also declined from 2003 to 2004, but was still greater than pre-fire values in Biodiversity Plots (see below) as well as in big sagebrush and winterfat plots (Table 1.14c). Values for all three parameters tended to be greatest in big sagebrush and winterfat plots.

Abundance of alien forbs was greatest in plots formerly dominated by big sagebrush or winterfat. Mean percent cover of alien forbs in these burned-over shrublands (7.3% ± 5.1 SD) was double that of all vegetation plots in 2004 (Table 1.14a). Mean percent frequency (93.3% ± 7.8 SD) was also substantially higher, and mean species richness was somewhat higher, in former shrublands compared to overall means (Table 1.14b, c). As with the overall means, means for each of these parameters in burned over shrublands were greater in 2003 than in 2004. In contrast with overall values, however, cover and frequency in these former shrublands were statistically greater in 2004 compared to pre-fire values ($P < 0.001$).

Alien forb abundance values were not uniform between plot types. Much lower pre-fire values for alien forb cover and richness were recorded in Biodiversity Plots compared to other plot types. The 5m x 20m sample unit used in Biodiversity Plots is relatively insensitive to low abundances of small-statured, ephemeral species such as spring whitlow-grass (*Draba verna*) and rough chickweed (*Holosteum umbellatum*) compared to the transect-and-microplot based methodologies of the Steppe-in-Time and BRMaP Plots. Moreover, detection of these species in detail was not an objective of the original Biodiversity survey, as it was from 2001 through 2004. Beginning in 2001, alien forb abundance in biodiversity plots was more nearly within the range of values observed in transect-based plots, although values remained consistently lower in Biodiversity plots throughout the study (Table 1.14a, c).

Percent cover of alien forbs was moderately correlated with fire severity ($r = 0.60$; $P < 0.0001$) and pre-fire shrub cover ($r = 0.53$: $P < 0.0001$; Table 1.15). Both cover ($r = -0.48$) and richness of alien forbs ($r = -0.55$) were moderately negatively correlated with native perennial species richness in 2004 ($P < 0.0001$). Percent frequency of alien forbs was strongly negatively correlated with elevation ($r = -0.76$), percent cover of native perennial species ($r = -0.83$), and with percent cover of perennial grasses ($P < 0.0001$). Alien forb frequency was also moderately negatively correlated with native perennial species richness ($r = -0.65$) and moderately correlated with fire severity ($r = 0.65$ in 2004 ($P < 0.0001$). Correlations of alien forb abundance with cheatgrass cover ranged from $r = 0.44$ to $r = 0.46$ ($P < 0.005$).

The abundance of individual alien forbs in 2004 tended to decline or remain similar to 2003 values. Nevertheless, tumble mustard (*Sisymbrium altissimum*), Russian thistle (*Salsola kali*), and redstem filaree (*Erodium cicutarium*) were statistically more abundant in 2004 compared to pre-fire values ($P < 0.005$; Table 1.16). Both percent cover and frequency of tumble mustard decreased in 2004 for the first time in the course of this study. However, the species still accounted for the greatest percent cover among the alien forbs in 2004 (2.0% ± 2.8 SD) as it has each year following the 24 Command Fire.

Both tumble mustard and spring whitlow-grass have been nearly ubiquitous in ALE vegetation plots since 2001. Rough chickweed had been equally ubiquitous in 2003; however, cover and frequency of this species declined considerably in 2004 and were significantly reduced compared to pre-fire values. Both rough chickweed and spring whitlow grass are small-statured spring ephemerals which may be undersampled and occasionally even overlooked, especially in larger sample units such as those in Biodiversity Plots. Percent cover and frequency of Russian thistle remained similar to 2003 values in 2004, but the proportion of plots in which the species was recorded continued to increase annually as it has since 2001. Percent cover at maturity of this warm season annual is greatly underestimated during the April-June field season, when its annual growth is just beginning. This may also be somewhat true for prickly lettuce (*Lactuca serriola*), although this species does not appear to be anywhere dense except along roadsides and similarly disturbed places.

Table 1.13. Alien forbs encountered in ALE Reserve vegetation plots, pre-fire - 2004. A = annual; B = biennial.

Scientific name	**Common name**	**Habit**
Descurainia sophia	flixweed, tansy mustard	A
Draba verna	spring whitlow grass	A
Erodium cicutarium	filaree	A
Holosteum umbellatum	rough chickweed	A
Lactuca serriola	prickly lettuce	A/B
Salsola kali	Russian thistle; tumbleweed	A
Sisymbrium altissimum	tumble mustard	A
Tragopogon dubius	meadow salsify	A/B

Table 1.14. Percent cover (a), frequency (b), and species richness (c) of alien forbs in ALE Reserve vegetation plots, pre-fire - 2004. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests.

a.		Percent Cover (± SD)										ANOVA	*t* test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
All Sites	70	2.5	(2.9)	3.5	(2.9)	5.1	(4.6)	7.0	(5.7)	3.2	(4.0)	<0.0001	0.254
Biodiversity Plots	33	0.6	(1.1)	3.2	(1.6)	3.7	(2.5)	5.8	(5.3)	2.6	(1.3)	<0.0001	<0.0001
BRMaP Transects	22	5.0	(3.0)	2.8	(2.8)	5.4	(5.0)	8.5	(6.3)	3.8	(4.4)	<0.0001	0.332
SIT Transects	15	3.1	(2.8)	5.3	(4.4)	8.0	(6.3)	7.5	(5.1)	3.7	(2.9)	0.002	0.609
All Transects	37	4.2	(3.0)	3.8	(3.7)	6.4	(5.6)	8.1	(5.9)	3.8	(3.8)	<0.0001	0.578
Big Sagebrush & Winterfat Plots													
All Sites	20	1.8	(2.5)	2.1	(1.7)	6.9	(5.0)	10.1	(6.8)	7.3	(5.1)	<0.0001	<0.0001
All Transects	12	2.7	(2.9)	1.2	(1.3)	8.4	(5.2)	12.5	(6.4)	7.5	(4.0)	<0.0001	<0.0001
Silt-Loam Soils	15	2.0	(2.9)	2.6	(1.6)	7.4	(4.9)	11.2	(7.5)	8.4	(5.3)	<0.0001	<0.001
Sandy Soils	5	1.0	(1.0)	0.4	(0.3)	5.4	(5.5)	6.6	(2.2)	3.8	(1.8)	0.008	0.019

b.		Percent Frequency (± SD)										ANOVA	*t* test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
BRMaP Transects	22	70.0	(28.9)	51.4	(26.3)	73.9	(25.4)	91.4	(9.3)	60.9	(35.6)	<0.0001	0.458
SIT Transects	15	65.3	(36.2)	88.0	(16.1)	90.3	(19.1)	84.3	(16.4)	61.0	(31.4)	0.004	0.620
All Transects	37	68.1	(31.7)	66.2	(28.9)	80.5	(24.2)	88.5	(12.9)	60.9	(33.5)	<0.0001	0.368
Big Sagebrush Transects													
All Transects	12	54.2	(29.2)	41.7	(26.5)	77.9	(23.2)	96.7	(3.9)	93.3	(7.8)	<0.0001	<0.001
Silt-Loam Soils	7	72.9	(20.2)	45.5	(24.3)	72.3	(29.8)	77.1	(41.5)	75.6	(38.8)	<0.0001	0.022
Sandy Soils	5	28.0	(16.4)	24.0	(21.9)	59.0	(25.3)	94.0	(4.2)	87.0	(7.6)	<0.0001	0.001

Continued

Table 1.14 (continued). Percent cover (a), frequency (b), and species richness (c) of alien forbs in ALE Reserve vegetation plots, pre-fire - 2004. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests.

c.	n	Species Richness (± SD)										ANOVA *P*	*t* test *P*
		Pre-fire		2001		2002		2003		2004			
All Sites	70	2.0	(1.8)	3.2	(1.3)	3.1	(1.0)	3.4	(1.0)	3.0	(1.3)	<0.0001	<0.0001
Biodiversity Plots	33	0.7	(1.1)	3.2	(1.2)	2.8	(1.0)	3.1	(1.0)	2.6	(1.3)	<0.0001	<0.0001
BRMaP Transects	22	3.0	(1.2)	2.9	(1.3)	3.3	(1.0)	3.6	(0.8)	3.3	(1.1)	0.080	0.296
SIT Transects	15	3.5	(1.8)	3.7	(1.1)	3.6	(0.9)	3.7	(1.1)	3.5	(1.1)	0.966	--
All Transects	37	3.2	(1.5)	3.2	(1.3)	3.4	(1.0)	3.7	(0.9)	3.4	(1.1)	0.250	--
Big Sagebrush & Winterfat Plots													
All Sites	20	1.9	(1.5)	2.8	(1.3)	3.2	(1.3)	3.9	(0.9)	3.7	(0.7)	<0.0001	<0.0001
All Transects	12	2.8	(1.0)	2.2	(0.9)	3.3	(1.3)	4.0	(0.9)	3.7	(0.8)	<0.0001	0.011
Silt-Loam Soils	15	1.7	(1.6)	3.1	(1.1)	3.6	(1.2)	4.1	(1.0)	3.8	(0.8)	<0.0001	<0.001
Sandy Soils	5	2.4	(1.1)	1.6	(1.1)	2.0	(0.7)	3.2	(0.4)	3.2	(0.4)	0.016	0.099

Table 1.15. Correlations coefficients of community and environmental variables with a) percent cover and frequency, and b) richness of alien forbs in vegetation plots on the ALE Reserve, Pre-fire-2004. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

a.		Alien forbs									
		Percent Cover					Percent Frequency				
		Pre-fire	2001	2002	2003	2004	Pre-fire	2001	2002	2003	2004
Pre-Fire Shrub Cover		-0.04	-0.24	0.25	0.33^{c}	0.53^{a}	-0.33^{e}	-0.68^{a}	-0.12	0.36^{e}	0.39^{e}
Litter		0.33^{d}	0.46^{a}	0.45^{b}	0.27	0.27	0.13	0.48^{c}	0.31	-0.09	0.33^{e}
Elevation		0.11	0.15	-0.26	-0.23	-0.42^{b}	0.01	0.09	-0.45^{d}	-0.46^{c}	-0.76^{a}
Fire Severity		-0.15	-0.25	0.29	0.39^{b}	0.60^{a}	-0.37^{e}	-0.57^{b}	-0.12	0.45^{c}	0.65^{a}
Slope		-0.07	-0.03	-0.21	-0.22	-0.33^{d}	-0.07	0.14	-0.15	-0.21	-0.39^{e}
Aspect		0.05	-0.06	-0.08	-0.26	-0.10	-0.14	0.20	-0.21	-0.36^{e}	-0.13
Heat Load Index		0.15	0.03	0.18	0.18	0.28	0.07	-0.07	0.07	0.10	0.38^{e}
Cheatgrass cover	Pre-fire	0.04	0.14	0.32^{d}	0.17	0.24	-0.29	-0.08	0.01	0.14	0.36^{e}
	2001	0.03	0.47^{a}	0.10	0.07	0.11	0.13	0.43^{d}	0.17	0.05	0.20
	2002	-0.10	0.34^{c}	0.06	0.16	0.34^{c}	-0.03	0.24	-0.03	-0.03	0.33^{e}
	2003	-0.07	0.41^{b}	0.18	0.15	0.26	0.00	0.29	0.14	-0.06	0.40^{e}
	2004	-0.08	0.37^{c}	0.29	0.21	0.44^{b}	-0.12	0.13	0.12	0.02	0.45^{c}
Native Perennial Cover	Pre-fire	0.02	-0.09	-0.29	-0.13	-0.21	0.08	-0.28	-0.31	-0.14	-0.43^{d}
	2001	-0.26	0.04	-0.34^{c}	-0.30^{e}	-0.48^{a}	0.03	0.45^{c}	-0.01	-0.38^{e}	-0.68^{a}
	2002	-0.07	0.13	-0.48^{a}	-0.45^{a}	-0.62^{a}	0.22	0.40^{e}	-0.11	-0.37^{e}	-0.84^{a}
	2003	-0.07	0.06	-0.49^{a}	-0.42^{b}	-0.64^{a}	0.13	0.32	-0.21	-0.40^{e}	-0.87^{a}
	2004	-0.09	0.00	-0.52^{a}	-0.35^{c}	-0.59^{a}	0.20	0.28	-0.20	-0.31	-0.83^{a}
Perennial Grass Cover	Pre-fire	-0.11	0.00	-0.30	-0.21	-0.34^{c}	0.39^{e}	0.08	-0.17	-0.21	-0.57^{b}
	2001	-0.32^{e}	-0.01	-0.32^{d}	-0.30^{e}	-0.45^{b}	0.12	0.50^{c}	0.12	-0.29	-0.53^{b}
	2002	-0.12	-0.05	-0.15	-0.20	-0.18	0.34^{e}	0.52^{b}	0.09	-0.29	-0.70^{a}
	2003	-0.07	0.06	-0.44^{b}	-0.40^{b}	-0.58^{a}	0.35^{e}	0.47^{c}	0.00	-0.36^{e}	-0.77^{a}
	2004	-0.10	0.02	-0.46^{a}	-0.39^{b}	-0.57^{a}	0.45^{d}	0.44^{d}	-0.01	-0.37^{e}	-0.77^{a}
Native Perennial Richness	Pre-fire	-0.04	0.02	-0.21	-0.17	-0.38	0.01	0.15	-0.29	-0.52^{c}	-0.48^{c}
	2001	-0.08	0.11	-0.31^{d}	-0.26	-0.55^{a}	0.00	0.29	-0.30	-0.48^{c}	-0.71^{a}
	2002	-0.10	0.02	-0.34^{c}	-0.18	-0.47^{a}	-0.01	0.13	-0.32	-0.42^{d}	-0.59^{b}
	2003	-0.04	-0.05	-0.36^{c}	-0.23	-0.46^{a}	0.11	0.02	-0.41^{e}	-0.38^{e}	-0.61^{a}
	2004	-0.10	-0.01	-0.34^{c}	-0.20	-0.48^{a}	-0.16	-0.04	-0.47^{c}	-0.40^{e}	-0.65^{a}

Continued

Table 1.15 (continued), Correlations coefficients of community and environmental variables with a) percent cover and frequency, and b) richness of alien forbs in vegetation plots on the ALE Reserve, Pre-fire-2004. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

b.		Alien Forb Richness				
		Pre-fire	2001	2002	2003	2004
Pre-Fire Shrub Cover		-0.10	-0.20	0.16	0.21	0.21
Litter		0.52[a]	0.24	0.41[b]	0.18	0.4 [b]
Elevation		-0.13	0.09	-0.10	-0.21	-0.46[a]
Fire Severity		-0.03	-0.15	0.21	0.23	0.36[c]
Slope		-0.26	-0.02	-0.16	-0.24	-0.46[a]
Aspect		0.03	0.09	0.06	-0.23	-0.25
Heat Load Index		0.27	0.08	0.18	0.20	0.45[a]
Cheatgrass cover	Pre-fire	0.38[c]	-0.03	0.06	0.24	0.32[d]
	2001	0.20	0.29	0.01	0.18	0.27
	2002	0.13	0.18	-0.07	0.07	0.36[c]
	2003	0.19	0.17	-0.05	0.17	0.42[b]
	2004	0.17	0.09	0.01	0.17	0.46[a]
Native Perennial Cover	Pre-fire	-0.40[b]	-0.01	-0.05	-0.24	-0.32[d]
	2001	-0.39[b]	0.15	-0.23	-0.32[d]	-0.47[a]
	2002	-0.29	0.22	-0.16	-0.37[c]	-0.48[a]
	2003	-0.33[c]	0.19	-0.17	-0.39[b]	-0.51[a]
	2004	-0.36[c]	0.20	-0.22	-0.24	-0.47[a]
Perennial Grass Cover	Pre-fire	-0.42[b]	0.03	-0.05	-0.37[c]	-0.36[c]
	2001	-0.42[b]	0.16	-0.17	-0.37[c]	-0.40[b]
	2002	-0.18	-0.05	-0.04	-0.01	-0.07
	2003	-0.27	0.24	-0.08	-0.39[b]	-0.38[c]
	2004	-0.28	0.23	-0.14	-0.33[c]	-0.39[b]
Native Perennial Richness	Pre-fire	-0.19	0.07	-0.19	-0.21	-0.46[a]
	2001	-0.28	0.19	-0.19	-0.28	-0.49[a]
	2002	-0.36[c]	0.07	-0.17	-0.33[d]	-0.54[a]
	2003	-0.25	0.04	-0.21	-0.23	-0.53[a]
	2004	-0.32[d]	0.07	-0.17	-0.31[d]	-0.55[a]

Table 1.16. Percent cover (a) and frequency (b) of selected common alien forb species in ALE Reserve vegetation plots, pre-fire - 2004. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests.

a.	# Plots (n = 70)					Cover (± SD; n = 70)											
Species	Pre-fire	2001	2002	2003	2004	Pre-fire		2001		2002		2003		2004		ANOVA P	*t* test P
Sisymbrium altissimum	19	39	54	58	57	0.2	(0.6)	1.1	(2.1)	1.8	(3.1)	3.6	(5.2)	2.0	(2.8)	< 0.0001	< 0.0001
Erodium cicutarium	4	10	11	14	18	0.1	(0.3)	0.1	(0.4)	0.3	(0.8)	0.3	(0.8)	0.5	(0.8)	< 0.001	0.004
Salsola kali	4	3	9	17	24	0.1	(0.2)	0.1	(0.2)	0.6	(2.2)	0.5	(1.4)	0.6	(0.9)	0.005	< 0.0001
Draba verna	34	59	59	63	59	0.7	(1.2)	0.8	(0.7)	1.1	(1.2)	1.6	(1.5)	1.4	(2.8)	0.025	0.174
Holosteum umbellatum	33	45	47	58	34	0.7	(1.2)	0.7	(1.0)	0.9	(1.1)	2.0	(3.3)	0.3	(0.4)	< 0.0001	0.002
Lactuca seriola	17	13	10	19	10	0.1	(0.3)	0.1	(0.3)	0.1	(0.3)	0.1	(0.3)	0.1	(0.2)	0.472	--
Tragopogon dubius	18	41	11	4	11	0.3	(0.8)	0.5	(0.5)	0.1	(0.3)	0.1	(0.2)	0.2	(0.4)	0.002	0.345

Continued

Table 1.16 (continued). Percent cover (a) and frequency (b) of selected common alien forb species in ALE Reserve vegetation plots, pre-fire - 2004. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests.

b.	Frequency ((± SD; n = 37)										ANOVA	*t* test
Species	Pre-fire		2001		2002		2003		2004		P	P
Sisymbrium altissimum	8.4	(14.2)	16.4	(24.7)	33.2	(33.6)	36.1	(35.4)	31.1	(32.7)	< 0.0001	< 0.0001
Erodium cicutarium	5.4	(17.9)	3.0	(9.4)	3.0	(8.1)	4.3	(10.2)	6.2	(12.8)	0.348	--
Salsola kali	0.0	(0.0)	0.0	(0.0)	8.1	(21.6)	15.5	(31.0)	15.2	(28.5)	< 0.0001	0.002
Draba verna	45.2	(36.0)	46.2	(29.1)	58.4	(34.2)	60.4	(21.4)	37.4	(31.7)	< 0.001	0.295
Holosteum umbellatum	39.0	(30.8)	28.3	(29.1)	50.8	(37.6)	49.6	(28.8)	13.3	(15.3)	< 0.0001	< 0.0001
Lactuca seriola	13.8	(22.6)	2.6	(7.3)	2.7	(6.8)	4.7	(9.6)	2.7	(6.4)	< 0.0001	0.008
Tragopogon dubius	6.2	(12.1)	8.8	(14.9)	0.3	(1.1)	0.0	(0.0)	1.0	(2.8)	< 0.0001	0.018

Native Species and Communities

Shrubs. The 24 Command Fire dramatically reduced the cover and frequency of large dominant shrubs in ALE Reserve study plots (Table 1.17). Dominant shrubs were entirely removed from all big sagebrush stands in the path of the wildfire. Percent cover of Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) in 17 shrub-dominated plots decreased from a pre-fire mean of more than 20.0% to zero immediately following the fire, while big sagebrush frequency in microplots declined from 53.8% (± 30.1 SD) to zero. Big sagebrush does not resprout following fire. Four years after the 24 Command Fire percent cover and frequency of big sagebrush remain far below pre-fire levels (P < 0.0001) with cover at only 0.3% (± 0.4 SD) and frequency at just 2.1 % (± 5.8 SD). While both cover and frequency of big sagebrush have increased slightly within vegetation plots since 2001, in 2004 sagebrush seedlings occurred in only 5 of 17 plots where the species had been dominant prior to the 24 Command Fire. In general, chance encounters by the TNC field crew with big sagebrush seedlings remote from surviving adult plants have increased every year since 2001, when such observations were extremely rare. While no quantitative data were collected regarding incidental observations, such encounters were not uncommon in 2004. In one instance a small stand of eight or more well-established seedlings was observed in a draw near an existing vegetation plot (SIT 14E).

Spiny hopsage (*Atriplex* [= *Grayia*] *spinosa*), a common associate of big sagebrush in lower elevation stands, was apparently eliminated from burned areas. Percent cover of spiny hopsage in study plots remained at 0.0% in 2004, down from a pre-fire mean of 5.8% (± 3.4 SD: P = 0.004), while frequency remained at 0.0%, down from 9.0% prior to the fire (± 4.2 SD; P = 0.009).

Percent cover of both threetip sagebrush (*Artemisia tripartita*) and winterfat (*Eurotia lanata*) was reduced but not eliminated from burned stands (Table 1.17a). These species, along with gray and green rabbitbrush (*Chrysothamnus* spp.), are capable of resprouting from their crowns following fire. Observations suggest that where dense stands (>10% cover in study plots) of these species occurred, many individuals were killed outright by the more intense fires generated by woody fuels. Where these species were more scattered, resprouting was common. Cover of threetip sagebrush in 2004 (2.8% ± 2.8 SD) was still far below pre-fire values (8.2 % ± 5.8 SD; P = 0.011). Frequency of threetip sagebrush also remained depressed compared to pre-fire values (P = 0.013; Table 1.17b). Percent cover of winterfat was reduced from 13.3% (±5.8 SD) to 0.7% (±0.6 SD) in 2001 and had recovered only to 1.7 % (± 1.2 SD; P = 0.088) by 2004.

Fire severity was strongly correlated with pre-fire shrub cover (r = 0.78; P < 0.0001). Other correlations of environmental variables with pre-fire shrub-cover were only weakly or not at all correlated.

Table 1.17. Mean percent cover (a) and frequency (b) of dominant large shrubs (Wyoming big sagebrush [*Artemisia tridentata* ssp. *wyomingensis*], spiny hopsage [*Atriplex* (=*Grayia*) *spinosa*], threetip sagebrush [*Artemisia tripartita*], and winterfat [*Eurotia lanata*]) in ALE Reserve vegetation plots, pre-fire – 2004. Analysis of spiny hopsage included all stands where the species was present. Analyses of other species included only stands where percent cover > 5.0 %. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests.

a.		Percent Cover (± SD)										ANOVA	*t* test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
Big Sagebrush													
All Plots	17	20.5	(7.3)	0.0	(0.0)	0.1	(0.3)	0.2	(0.3)	0.3	(0.4)	<0.0001	<0.0001
All Transects	12	20.1	(9.2)	0.0	(0.0)	0.03	(0.1)	0.06	(0.1)	0.1	(0.3)	<0.0001	<0.0001
Spiny Hopsage													
All Plots	7	5.8	(3.4)	0.0	(0.0)	0.0	(0.0)	0.0	(0.0)	0.0	(0.0)	<0.0001	0.004
All Transects	5	4.2	(2.4)	0.0	(0.0)	0.0	(0.0)	0.0	(0.0)	0.0	(0.0)	<0.0001	0.017
Threetip Sagebrush													
All Plots	11	8.2	(5.8)	1.4	(1.2)	1.5	(1.2)	2.7	(2.7)	2.8	(2.8)	<0.0001	0.011
All Transects	7	8.1	(7.1)	0.8	(0.8)	0.9	(0.9)	1.5	(1.3)	1.6	(1.7)	<0.001	0.041
Winterfat -- All Plots	3	13.3	(5.8)	0.7	(0.6)	1.0	(0.0)	1.7	(1.2)	1.7	(1.2)	0.002	0.088

Continued

Table 1.17 (continued). Mean percent cover (a) and frequency (b) of dominant large shrubs (Wyoming big sagebrush [*Artemisia tridentata* ssp. *wyomingensis*], spiny hopsage [*Atriplex* (=*Grayia*) *spinosa*], threetip sagebrush [*Artemisia tripartita*], and winterfat [*Eurotia lanata*]) in ALE Reserve vegetation plots, pre-fire – 2004. Analysis of spiny hopsage included all stands where the species was present. Analyses of other species included only stands where percent cover > 5.0 %. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests.

b.		Percent Frequency (± SD)										ANOVA	*t* test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
Big Sagebrush	12	53.8	(30.1)	0.0	(0.0)	1.3	(2.3)	0.8	(1.9)	2.1	(5.8)	<0.0001	<0.0001
Spiny Hopsage	5	9.0	(4.2)	0.0	(0.0)	0.0	(0.0)	0.0	(0.0)	0.0	(0.0)	<0.0001	0.009
Threetip Sagebrush	7	28.6	(18.4)	12.9	(12.2)	9.3	(8.9)	10.0	(10.4)	15.0	(12.6)	<0.0001	0.013

Hemishrubs. Longleaf phlox (*Phlox longifolia*), threadleaf fleabane, (*Erigeron filifolius*) and many other hemishrubs commonly encountered in ALE Reserve study plots (Table 1.18) are able to resprout following wildfire. Mean percent cover of hemishrubs decreased by more than 50 % following the 24 Command Fire. However, overall cover has increased annually since 2002, and in 2004 exceeded pre-fire values (3.2 % ± 4.0 SD; $P = 0.035$; Table 1.19a). Mean percent frequency of hemishrubs also returned to pre-fire values following a small decrease after the fire, and by 2004 had increased marginally over pre-fire values ($P = 0.078$). Mean richness of hemishrubs changed only slightly following the fire, and values in 2004 were statistically similar to pre-fire values ($P = 0.531$).

Percent cover of hemishrubs was only weakly or not at all correlated with community and environmental variables (Table 1.20a). Percent frequency in 2004 was moderately correlated with slope (r = 0.45; $P < 0.01$) as well as with percent cover (r = 0.55; $P < 0.001$) and richness (r = 0.47; $P < 0.005$) of native perennials. Frequency in 2004 was moderately negatively correlated with percent cover of cheatgrass in 2004 (r = - 0.63; $P < 0.0001$), as well as with heat load (r = - 0.52; $P < 0.005$), and litter (r = - 0.47; $P < 0.005$).

Hemishrub richness in 2004 was moderately correlated with slope (r = 0.66; $P < 0.0001$) as well as with elevation (r = 0.50; $P < 0.0001$; Table 1.20b). Richness was also moderately correlated with native perennial richness (r = 0.67; $P < 0.0001$) and less strongly with native perennial cover (r = 0.44; $P < 0.001$). Hemishrub richness was moderately negatively correlated with heat load (r = - 0.59; $P < 0.0001$) and with percent cover of cheatgrass in 2004 (r = - 0.50; $P < 0.0001$).

Longleaf phlox, the most abundant and widespread of the hemishrubs on ALE, exhibited a pattern of change following the 24 Command Fire that was very similar to that of overall hemishrub abundance. By 2004 percent cover and frequency of longleaf phlox were statistically similar to, but did not exceed, pre-fire values (Table 1.19b).

Table 1.18. Occurrence of selected common hemishrubs in ALE Reserve vegetation plots, pre-fire - 2004. Habitats characterized by *Eriogonum thymoides*, *Phlox hoodii*, and other hemishrubs associated with lithosolic substrates were sampled only peripherally.

Scientific Name	**Common Name**	**Plots pre-fire**	**Plots 2004**
Antennaria dimorpha	low pussytoes	13	18
Erigeron filifolius	threadleaf daisy	25	22
Erigeron piperianus	Piper's daisy	5	7
Eriogonum heracloides	parsnip-flowered buckwheat	7	6
Eriogonum strictum	strict buckwheat	8	3
Phlox hoodii	cushion phlox	2	2
Haplopappus stenophyllus	narrowleaf goldenweed	3	3
Phlox longifolia	longleaf phlox	52	58

Table 1.19. Percent cover, frequency, and richness of hemishrubs in vegetation plots on the ALE Reserve, Pre-fire -2004. a) all hemishrubs; b) *Phlox longifolia* (longleaf phlox). ANOVA *P* values are results of single factor repeated measures ANOVA. *t* test *P* values are the results of comparisons of pre-fire and 2004 values using paired-sample *t* tests.

a. All Hemishrubs	n	Values (+ SD)					ANOVA	*t* test
		Pre-fire	2001	2002	2003	2004	*P*	*P*
Percent Cover	70	2.1 (2.7)	0.9 (0.8)	2.4 (3.0)	3.1 (3.6)	3.2 (4.0)	< 0.0001	0.035
Percent Frequency	37	31.5 (25.8)	25.5 (25.3)	31.6 (24.3)	35.5 (27.8)	39.3 (30.3)	0.002	0.078
Species richness	70	1.7 (1.3)	1.5 (1.0)	1.6 (1.1)	1.6 (1.1)	1.8 (1.2)	0.068	0.531

b. *Phlox longifolia*	n	Pre-fire	2001	2002	2003	2004	ANOVA *P*	*t* test *P*
Percent Cover	70	1.5 (2.2)	0.7 (0.7)	2.3 (3.0)	2.3 (3.2)	2.4 (3.2)	< 0.0001	0.277
Percent Frequency	37	0.3 (0.3)	0.3 (0.1)	0.3 (0.3)	0.3 (0.3)	0.4 (0.3)	0.053	0.120

Table 1.20. Correlation coefficients of community and environmental variables with a) percent cover and frequency, and b) richness of hemishrubs in vegetation plots on the ALE Reserve, Pre-fire-2004. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

a.	Hemishrubs									
	Percent Cover					Percent Frequency				
	Pre-fire	2001	2002	2003	2004	Pre-fire	2001	2002	2003	2004
Pre-Fire Shrub Cover	0.04	-0.19	-0.01	0.06	0.11	-0.01	-0.12	-0.03	0.03	0.09
Litter	-0.07	-0.36^{c}	-0.33^{d}	-0.37^{c}	-0.35^{c}	-0.14	-0.33^{e}	-0.40^{e}	-0.51^{c}	-0.47^{c}
Elevation	0.07	0.19	-0.06	-0.05	-0.09	0.05	0.36^{e}	0.39^{e}	0.34^{e}	0.31
Fire Severity	-0.06	-0.22	0.04	-0.02	0.08	-0.06	-0.25	-0.14	-0.14	-0.06
Slope	0.09	0.33^{d}	0.11	0.10	0.01	0.34^{e}	0.53^{b}	0.49^{c}	0.30	0.45^{d}
Aspect	-0.13	-0.04	0.04	0.07	-0.04	-0.21	-0.04	-0.07	-0.10	-0.12
Heat Load Index	-0.08	-0.25	-0.07	-0.05	0.01	-0.43^{d}	-0.60	-0.51^{c}	-0.31	-0.52^{c}
Cheatgrass cover Pre-fire	-0.33^{e}	-0.23	-0.27	-0.27	-0.26	-0.63^{a}	-0.41^{e}	-0.48^{c}	-0.35^{e}	-0.39
2001	-0.01	-0.20	-0.24	-0.19	-0.15	-0.07	-0.38^{e}	-0.45^{c}	-0.36^{e}	-0.46^{c}
2002	-0.13	-0.22	-0.24	-0.24	-0.13	-0.21	-0.46^{c}	-0.47^{c}	-0.47^{c}	-0.54^{b}
2003	-0.13	-0.25	-0.22	-0.17	-0.03	-0.26	-0.52^{c}	-0.56^{b}	-0.54^{b}	-0.61^{a}
2004	-0.21	-0.36^{c}	-0.33^{c}	-0.30^{e}	-0.21	-0.30	-0.56	-0.60^{a}	-0.57^{b}	-0.63^{a}
Native Perennial Pre-fire	0.26	0.19	0.12	0.10	0.05	0.41^{e}	0.36^{e}	0.41^{e}	0.45^{d}	0.44^{d}
Cover 2001	0.03	0.27	0.11	0.18	0.07	0.12	0.46^{c}	0.32	0.23	0.32
2002	-0.01	0.31	0.20	0.17	0.09	0.15	0.51^{c}	0.45^{d}	0.40^{e}	0.40^{e}
2003	0.01	0.37^{c}	0.18	0.19	0.09	0.13	0.57^{b}	0.49^{c}	0.48^{c}	0.46^{c}
2004	0.11	0.39^{b}	0.20	0.27	0.19	0.24	0.63^{a}	0.58^{b}	0.59^{b}	0.55^{b}
Perennial Grass Pre-fire	0.09	0.21	0.15	0.05	-0.04	0.52^{b}	0.43^{d}	0.43^{d}	0.40^{e}	0.39^{e}
Cover 2001	-0.13	0.19	0.11	0.12	0.02	0.14	0.44^{d}	0.27	0.16	0.27
2002	-0.07	0.06	-0.04	-0.09	-0.07	0.21	0.52^{c}	0.41^{e}	0.35^{e}	0.36^{e}
2003	-0.08	0.27	0.04	0.00	-0.09	0.13	0.50^{c}	0.36^{e}	0.36^{e}	0.34^{e}
2004	-0.02	0.27	0.05	0.05	-0.06	0.18	0.47^{c}	0.40^{e}	0.41^{e}	0.35^{e}
Native Perennial Pre-fire	0.20	0.30	0.18	0.23	0.16	0.17	0.30	0.42^{d}	0.28	0.29
Richness 2001	0.08	0.34^{e}	0.17	0.20	0.13	0.13	0.42^{d}	0.43^{d}	0.36^{e}	0.36^{e}
2002	0.11	0.33^{c}	0.16	0.19	0.11	0.18	0.47^{c}	0.52^{c}	0.43^{d}	0.45^{d}
2003	0.19	0.36^{d}	0.18	0.23	0.11	0.22	0.44^{d}	0.52^{c}	0.50^{c}	0.45^{d}
2004	0.04	0.27^{c}	0.11	0.14	0.05	0.11	0.47^{c}	0.50^{c}	0.45^{d}	0.47^{c}

Continued

Table 1.20 (continued). Correlation coefficients of community and environmental variables with a) percent cover and frequency, and b) richness of hemishrubs in vegetation plots on the ALE Reserve, Pre-fire-2004. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

b.	Hemishrub Richness				
	Pre-fire	2001	2002	2003	2004
Pre-Fire Shrub Cover	-0.23	-0.32^{d}	-0.22	-0.18	-0.32^{d}
Litter	-0.36^{c}	-0.46^{a}	-0.43^{b}	-0.37^{c}	-0.43^{b}
Elevation	0.47^{a}	0.49^{a}	0.49^{a}	0.33^{d}	0.50^{a}
Fire Severity	-0.31^{d}	-0.43^{b}	-0.38^{c}	-0.24	-0.36^{c}
Slope	0.53	0.68	0.66^{a}	0.55^{a}	0.66^{a}
Aspect	-0.03	-0.12	-0.13	0.00	-0.02
Heat Load Index	-0.48	-0.65	-0.59^{a}	-0.44^{b}	-0.59^{a}
Cheatgrass cover Pre-fire	-0.43^{b}	-0.33^{d}	-0.34^{c}	-0.18	-0.23
2001	-0.20	-0.33^{c}	-0.29	-0.27	-0.29
2002	-0.31^{d}	-0.39^{b}	-0.33^{d}	-0.30^{e}	-0.33^{c}
2003	-0.42^{b}	-0.45^{a}	-0.44^{b}	-0.37^{c}	-0.43^{b}
2004	-0.50^{a}	-0.58^{a}	-0.53^{a}	-0.45^{a}	-0.50^{a}
Native Perennial Cover Pre-fire	0.41^{b}	0.40^{b}	0.44^{b}	0.28	0.37^{c}
2001	0.43^{b}	0.70^{a}	0.53^{a}	0.31^{d}	0.42^{b}
2002	0.43^{b}	0.57^{a}	0.47^{a}	0.24	0.43^{b}
2003	0.44^{b}	0.61^{a}	0.51^{a}	0.31^{d}	0.48^{a}
2004	0.43^{b}	0.67^{a}	0.53^{a}	0.30	0.44^{b}
Perennial Grass Cover Pre-fire	0.40^{b}	0.47^{a}	0.39^{b}	0.23	0.38^{c}
2001	0.25	0.62^{a}	0.41^{b}	0.23	0.34^{c}
2002	0.17	0.16	0.01	-0.14	0.09
2003	0.27	0.51^{a}	0.37	0.20	0.33^{d}
2004	0.30^{d}	0.55^{a}	0.41	0.20	0.35^{c}
Native Perennial Richness Pre-fire	0.65^{a}	0.57^{a}	0.63^{a}	0.46^{a}	0.61^{a}
2001	0.53^{a}	0.66^{a}	0.61^{a}	0.45^{a}	0.59^{a}
2002	0.52^{a}	0.63^{a}	0.67^{a}	0.53^{a}	0.61^{a}
2003	0.54^{a}	0.61^{a}	0.65^{a}	0.60^{a}	0.65^{a}
2004	0.48^{a}	0.60^{a}	0.64^{a}	0.56^{a}	0.67^{a}

Table 1.21. Record of observations of Piper's daisy (*Erigeron piperianus*) in and around vegetation plots on the ALE Reserve, pre-fire – 2004. Observations in vicinity of vegetation plots are reported only when the species is not recorded within the sample area.

b.	Number of Observations				
	Pre-fire	2001	2002	2003	2004
BRMaP 8 --in microplots	0	0	0	0	0
(3 transects) --in vicinity	2	0	0	0	0
BRMaP 23 --in microplots	0	0	0	Not	Not
(3 transects) --in vicinity	1	1	0	Sampled	Sampled
BRMaP 24 --in microplots	1	0	1	1	1
(5 transects) --in vicinity	4	2	1	1	4
SIT 14E --in microplots	1	0	0	0	1
--in vicinity	0	1	1	1	0
SIT 14W --in microplots	1	0	1	1	0
--in vicinity	0	1	0	0	1
SIT 67E --in microplots	0	1	0	0	0
--in vicinity	0	0	0	0	0
SIT 326 --in microplots	0	0	0	1	0
--in vicinity	0	0	0	0	0
Biodiversity 42 --in plot	0	1	1	1	1
--in vicinity	0	0	0	0	0
Biodiversity 43 --in plot	0	1	0	0	1
--in vicinity	0	0	0	0	0
Biodiversity 44 --in plot	1	0	1	1	1
--in vicinity	0	1	0	0	0
Biodiversity 47 --in plot	1	1	1	1	1
--in vicinity	0	0	0	0	0
Biodiversity 50 --in plot	1	1	1	1	1
--in vicinity	0	0	0	0	0
Biodiversity 51 --in plot	0	1	0	0	0
--in vicinity	0	0	0	0	0
Biodiversity 46 – in vicinity	1	0	0	0	0
Biodiversity 55 – in vicinity	1	0	0	0	0
Biodiversity 56 – in vicinity	0	0	0	0	1
Biodiversity 96 – in vicinity	0	0	0	0	1
Total within plots	6	6	6	7	7
Total in vicinity	9	6	2	2	7
Total observations	15	12	8	9	14

The hemishrub Piper's daisy (*Erigeron piperianus*) is a regional endemic found only in the Columbia Basin of Washington state, and is categorized as sensitive by the Washington Natural Heritage Program (WNHP 2000, 1997). Prior to the 24 Command Fire, small numbers of Piper's daisy were recorded in each of the reference plot types used in this study. The number of overall observations of Piper's daisy decreased during the first years following the 24 Command Fire, although the number of records within plots remained relatively constant (Table 1.21). By 2004 the number of observations within and nearby to vegetation plots was similar to the number of observations made before the fire .

Percent cover of Piper's daisy declined slightly following the 24 Command Fire, but increased gradually in subsequent years (Table 1.22). Cover in 2004 (3.4 % ± 6.6 SD) was numerically greater than pre-fire values (0.7 % ± 1.2 SD); however, this difference was at best only marginally significant (P = 0.125). Percent frequency, based on data from only four transects, did not vary significantly between years (P = 0.827). In general, Piper's daisy was observed scattered across the lower slopes of Rattlesnake Mountain between approximately 700' and 1400' a.s.l., generally within the area of occurrence suggested by Caplow and Beck (1995). In 2001 this species appeared to be widely scattered, occurring singly or in clusters of a few plants. In a few instances larger clusters of 10-30 plants were observed. By 2004 incidental observations were very common. Plants appeared to be considerably more numerous (or at least more conspicuous) as well as more robust. Circumstantial evidence in support of this observation comes from the fact that the species was recorded for the first time in the vicinity of Biodiversity Plots 56 and 96 in 2004. Areas of dense concentrations (e.g., greater than 15 % cover of this species was recorded in Biodiversity Plot 44 in 2004) were more commonly observed throughout the area, and these areas appeared to be more extensive than in 2001.

Table 1.22. Abundance of Piper's daisy (*Erigeron piperianus*) in vegetation plots on the ALE Reserve, pre-fire – 2004. ANOVA P values are results of single factor repeated measures ANOVA. t test P values are the results of comparisons of pre-fire and 2004 values using paired-sample t tests.

		Abundance (± SD) of *Erigeron piperianus*					ANOVA	*t* test
	n	Pre-fire	2001	2002	2003	2004	*P*	*P*
Percent Cover	10	0.7 (1.2)	0.6 (0.5)	0.8 (1.2)	2.5 (4.1)	3.4 (6.6)	0.155	0.125
Percent Frequency	4	3.8 (4.8)	2.5 (5.0)	3.8 (4.8)	6.3 (4.8)	3.8 (4.8)	0.827	-----

Native Grasses. Mean percent cover of native perennial bunchgrasses declined by more than 50 % in 2001 (19.3% ± 15.1 SD) relative to pre-fire observations (38.9% ± 22.2 SD; Table 1.23a). Large warm season bunchgrasses such as bluebunch wheatgrass (*Agropyron spicatum*), Idaho fescue (*Festuca idahoensis*), needle-and-thread (*Stipa comata*, *S. thurberiana*), and others were affected more than the cool season perennial Sandberg's bluegrass (*Poa sandbergii*). Mean percent cover of large warm season bunchgrasses declined by more than 70% over all vegetation plots where they occurred. Bluebunch wheatgrass, the dominant large bunchgrass over most of the ALE Reserve at elevations above 800 – 900' (240-275 m), declined in cover from 20.1% (± 16.4 SD) prior to the fire to 5.8% (± 7.1 SD) in 2001. While percent cover of bluebunch wheatgrass has more than doubled since 2001, percent cover in 2004 was still only two-thirds of pre-fire values (13.4% ± 11.3 SD; $P < 0.0001$). Similar or greater reductions in percent cover were observed for both Idaho fescue and needle-and-thread following the fire; by 2004 neither of these two species had recovered as much as 40 % of their pre-fire cover (Table 1.23a).

In contrast, Sandberg's bluegrass exhibited cover losses of only 20% to 40% the first year following the 24 Command Fire. Of the major dominant bunchgrasses, only Sandberg's bluegrass appears to have completely recovered in terms of abundance following the fire. Cover values for Sandberg's bluegrass numerically exceeded pre-fire values as early as 2002, and values in 2004 (14.3 % ± 7.9 SD) were statistically similar to pre-fire values (13.0 % ± 9.4 SD; $P = 0.288$).

Changes in percent frequency of perennial bunchgrasses following the 24 command fire were not as great as the declines recorded for percent cover (Table 1.23b). Mean percent frequency declined only slightly in 2001 (78.2 % ± 28.9 SD) compared to pre-fire values (82.7 % ± 29.1 SD), and was statistically similar to pre-fire values in 2004 (81.1 % ± 26.9 SD; $P = 0.568$). Even where post-fire declines were greatest, as in big sagebrush transects and in *Festuca* and *Stipa* stands, frequency values had generally recovered substantially to pre-fire levels by 2004 or earlier. Again, Sandberg's bluegrass contrasted with other native bunchgrasses; although frequency increased numerically from pre-fire values to 2004, differences were not significant ($P = 0.330$).

Although overall species richness of perennial bunchgrasses declined somewhat just after the 24 Command fire, by 2004 richness was statistically similar to pre-fire values ($P = 0.417$). The most dramatic effect was observed in big sagebrush stands, where the decline in richness was still marginally significant in 2004 ($P = 0.110$; Table 1.23c).

The strongest correlations between environmental variables and native perennial bunchgrass cover in 2004 were moderate negative correlations with fire severity (r = - 0.62; $P < 0.0001$) and pre-fire shrub cover (r = - 0.57; $P < 0.0001$; Table 1.24a). Bunchgrass cover in 2004 was also moderately negatively correlated with percent cover of cheatgrass (r = - 0.47; $P < 0.0001$) and with heat load (r = - 0.43; $P < 0.001$), and was moderately correlated with slope and elevation (r = 0.46 and 0.45, respectively; $P < 0.0001$). Percent frequency of native perennial bunchgrasses was similarly correlated with the same suite of environmental and community variables. The correlations between frequency and fire severity were particularly strong from 2001 through 2004 (r = - 0.69 to – 0.84; $P < 0.0001$). The strongest correlation with perennial grass richness in 2004 was a moderate negative correlation with percent cover of cheatgrass (r = - 0.51; *P*

< 0.0001). Richness in 2004 was also moderately negatively correlated with heat load (r = - 0.43; $P < 0.001$) and moderately correlated with elevation and slope (r = 0.50 and 0.49, respectively; $P < 0.0001$; Table 1.24b).

Survival of bunchgrasses $\geq$ 5.0 cm in diameter ranged from zero to 100% following the 24 Command Fire. Mean survival for 38 Biodiversity Plots where survival of large bunchgrasses was sampled in 2001 was 65.6% (± 38.3 SD) but the response was bimodal, with survival in more than 70% of plots (n = 27) equal to or greater than 59.8% (mean 88.1% ± 14.2 SD) while in 11 high-mortality plots survival was equal to or considerably less than 40.8% (mean 10.4% ± 13.4 SD; $P < 0.0001$). There was no difference in survival between plots with < 75.0% survival in 2001 (30.3% ± 29.2 SD) and values for the same plots when resampled in 2002 (32.9% ± 27.6; $P = 0.799$), supporting the field assessments made in 2001.

High mortality plots all occurred below 900' (275 m) elevation. The five plots with the lowest survival (0.0% - 0.7%) were stands with relatively dense (10% - 30%) pre-fire cover of big sagebrush or winterfat. Four of the remaining six stands were grasslands with substantial (> 35.0%) pre-fire cover of needle-and-thread grass.

Bunchgrass survival was most strongly correlated with both total pre-fire cover of native perennial bunchgrasses and with pre-fire cover of large warm season bunchgrasses (r = 0.65 and r = 0.64, respectively; $P < 0.0001$). Survival was also moderately correlated with elevation (r = 0.61; $P < 0.0001$) and slope (r = 0.46; $P < 0.005$), and was moderately negatively correlated with fire severity (r = -0.63; $P < 0.0001$), heat load index (r = 0.46; $P < 0.005$), and pre-fire shrub cover (r = -0.45; $P < 0.005$). Correlation with pre-fire cover of cheatgrass was relatively weak (r = -0.33; $P < 0.05$).

Field observations suggest that, where fire intensities appeared to be low to moderate, Sandberg's bluegrass was little affected. Percent cover reductions over all plots were much smaller for this species compared to the larger bunchgrasses (Table 1.23a), and mortality for this species appeared to be quite low where shrubs were absent. Cusick's bluegrass (*Poa cusickii*), a common associate on more relatively mesic sites at middle and higher elevations, appeared to respond similarly. Where dense stands of shrubs generated more severe fire behavior, Sandberg's bluegrass suffered considerable loss of cover, and mortality appeared to be high.

The density of large native bunchgrasses varied by habitat type (Table 1.25). Some low elevation sites had no bunchgrasses large enough to be included in the sample. Where larger bunchgrasses were present, low and mid-elevation big sagebrush stands and winterfat stands had an overall total density of 2.5 tussocks/ m^2 (± 1.0 SD). Total density in low and mid-elevation bluebunch wheatgrass or needle-and-thread grasslands was 3.9 tussocks/ m^2 (± 2.2 SD). High elevation habitats characterized by threetip sagebrush (where shrubs were present) and Idaho fescue had by far the highest density of all samples, 12.6 tussocks/ m^2 (± 6.3 SD).

Table 1.23. Mean percent cover (a), frequency (b), and richness (c) of native perennial bunchgrasses in vegetation plots on the ALE Reserve, pre-fire -2004. ANOVA *P* values are results of single factor repeated measures ANOVA. *t* test *P* values are the results of comparisons of pre-fire and 2004 values using paired-sample *t* tests.

a.		Percent Cover (± SD)										ANOVA	*t*- test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
All native perennial grasses													
All plots	70	38.9	(22.2)	19.3	(15.1)	26.3	(13.8)	29.8	(16.2)	29.7	(16.3)	< 0.0001	< 0.0001
Big sagebrush plots	17	22.0	(15.4)	5.6	(6.4)	10.0	(10.1)	10.9	(12.6)	11.6	(13.1)	< 0.001	0.005
Transects only	37	28.1	(18.0)	12.1	(9.3)	20.4	(13.0)	23.4	(14.9)	22.6	(14.2)	< 0.0001	0.040
Warm season bunchgrasses[1]													
All plots	65	24.4	(23.2)	7.0	(9.2)	10.3	(10.3)	13.5	(11.3)	13.4	(11.3)	< 0.0001	< 0.0001
Transects only	33	16.5	(14.2)	3.4	(2.7)	7.4	(5.8)	10.4	(6.6)	10.1	(7.2)	< 0.0001	0.005
Agropyron spicatum													
All plots	55	20.1	(16.4)	5.8	(7.1)	10.6	(9.2)	13.3	(9.8)	13.5	(10.1)	< 0.0001	< 0.0001
Transects only	28	15.2	(10.8)	2.4	(1.6)	7.2	(5.3)	10.7	(5.8)	10.3	(6.6)	< 0.0001	0.017
Festuca idahoensis													
All plots	16	23.3	(22.4)	6.4	(8.4)	7.5	(8.2)	9.2	(8.9)	8.2	(9.8)	< 0.0001	0.002
Transects only	8	12.4	(11.4)	1.1	(1.1)	3.7	(3.9)	3.9	(4.1)	3.6	(3.0)	0.004	0.078
***Stipa* spp.**[2]													
All plots	25	9.5	(16.6)	1.4	(3.3)	1.3	(2.6)	3.3	(5.7)	2.2	(4.6)	< 0.001	0.014
Transects only	11	1.6	(2.6)	0.1	(0.1)	0.4	(0.4)	0.5	(0.5)	0.7	(0.5)	0.045	0.269
Poa sandbergii													
All plots	70	13.0	(9.4)	10.3	(7.5)	15.5	(8.9)	14.4	(7.8)	14.3	(7.9)	< 0.0001	0.288
Transects only	37	11.2	(7.3)	6.5	(4.6)	12.4	(7.5)	11.7	(7.1)	11.2	(6.2)	< 0.0001	0.995

Table 1.23 (continued). Mean percent cover (a), frequency (b), and richness (c) of native perennial bunchgrasses in vegetation plots on the ALE Reserve, pre-fire -2004. ANOVA *P* values are results of single factor repeated measures ANOVA. *t* test *P* values are the results of comparisons of pre-fire and 2004 values using paired-sample *t* tests.

b.		Percent Frequency (± SD)										ANOVA	*t* test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
All native perennial grasses													
All Transects	37	82.7	(29.1)	78.2	(28.9)	76.2	(29.8)	80.1	(28.2)	81.1	(26.9)	0.079	0.568
Big sagebrush Transects	12	63.8	(41.3)	44.2	(26.0)	41.3	(25.9)	50.8	(31.1)	55.0	(31.4)	< 0.001	0.147
Warm season bunchgrasses[1]	33	65.5	(33.9)	53.9	(35.5)	50.1	(34.8)	57.9	(31.5)	57.9	(33.1)	< 0.001	0.022
Agropyron spicatum	27	68.9	(30.3)	63.7	(29.3)	57.2	(30.4)	67.2	(19.2)	62.4	(28.3)	0.083	0.225
Festuca idahoensis	8	40.6	(30.8)	23.1	(20.5)	28.8	(21.8)	28.1	(28.4)	30.3	(25.6)	0.295	----
***Stipa* spp.**[2]	11	11.4	(13.2)	2.3	(4.1)	8.2	(3.6)	10.0	(14.0)	9.1	(5.4)	0.074	0.624
Poa sandbergii	37	74.3	(28.0)	75.1	(29.1)	72.0	(28.5)	80.1	(20.8)	89.2	(73.5)	0.330	----

c.		Perennial Species Richness (± SD)										ANOVA	*t* test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
All Plots	70	3.0	(1.1)	2.8	(1.2)	2.9	(1.2)	2.8	(1.2)	3.1	(1.2)	0.026	0.417
Big sagebrush Plots	17	2.6	(1.3)	1.6	(0.9)	2.0	(1.3)	1.9	(1.4)	2.3	(1.1)	< 0.001	0.111

1 -- Warm season bunchgrasses include bluebunch wheatgrass (*Agropyron spicatum*), Idaho fescue (*Festuca idahoensis*), needle-and-thread (*Stipa comata*) and Thurber's needlegrass (*S. thurberiana*), ricegrass (*Oryzopsis hymenoides*), squirreltail (*Sitanion hystrix*), and Great Basin wildrye (*Elymus cinereus*).

2 – Stipa species include needle-and-thread (*Stipa comata*) and Thurber's needlegrass (*S. thurberiana*).

Table 1.24. Correlation coefficients of community and environmental variables with (a) percent cover and frequency, and (b) richness of native perennial bunchgrasses in vegetation plots on the ALE Reserve, Pre-fire-2004. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

a.	Native Perennial Bunchgrasses									
	Percent Cover					Percent Frequency				
	Pre-fire	2001	2002	2003	2004	Pre-fire	2001	2002	2003	2004
Pre-Fire Shrub Cover	-0.32[d]	-0.48[a]	-0.64[a]	-0.59[a]	-0.57[a]	-0.29	-0.76[a]	-0.71[a]	-0.62[a]	-0.56[b]
Litter	-0.35[c]	-0.29	-0.28	-0.26	-0.34[c]	0.00	0.04	0.00	-0.04	-0.11
Elevation	0.55[a]	0.42[b]	0.37[c]	0.43[b]	0.45[a]	0.44[d]	0.51[c]	0.55[b]	0.53[b]	0.54[b]
Fire Severity	-0.42[b]	-0.51[a]	-0.65[a]	-0.60[a]	-0.62[a]	-0.48[c]	-0.84[a]	-0.83[a]	-0.74[a]	-0.69[a]
Slope	0.50[a]	0.58[a]	0.45[a]	0.44[b]	0.46[a]	0.42[d]	0.43[d]	0.46[c]	0.43[d]	0.47[c]
Aspect	-0.11	0.09	-0.03	0.07	-0.01	0.07	0.22	0.25	0.19	0.19
Heat Load Index	-0.51[a]	-0.56[a]	-0.45[a]	-0.38[c]	-0.43[b]	-0.41[e]	-0.38[e]	-0.39[e]	-0.38[d]	-0.43[d]
Cheatgrass Cover Pre-fire	-0.66[a]	-0.38[c]	-0.42[b]	-0.42[b]	-0.43[b]	-0.76[a]	-0.27	-0.41[e]	-0.44[d]	-0.49[c]
2001	-0.16	-0.22	-0.11	-0.23	-0.19	-0.08	-0.02	-0.05	-0.09	-0.15
2002	-0.30[e]	-0.34[c]	-0.28	-0.36[c]	-0.32	-0.36[e]	-0.24	-0.29	-0.30	-0.38[e]
2003	-0.34[c]	-0.34[c]	-0.28	-0.41[b]	-0.38[c]	-0.35[e]	-0.18	-0.25	-0.30	-0.37[e]
2004	-0.39[b]	-0.43[b]	-0.41[b]	-0.50[a]	-0.47[a]	-0.48[c]	-0.35[e]	-0.43[d]	-0.49[c]	-0.54[b]

b.	Native Perennial Bunchgrasses				
	Species Richness				
	Pre-fire	2001	2002	2003	2004
Pre-Fire Shrub Cover	-0.09	-0.32[d]	-0.24	-0.38[c]	-0.22
Litter	-0.16	-0.32[d]	-0.24	-0.34[c]	-0.27
Elevation	0.40[b]	0.68[a]	0.40[b]	0.51[a]	0.50[a]
Fire Severity	-0.27	-0.55[a]	-0.33[c]	-0.42[b]	-0.38[b]
Slope	0.36[c]	0.56[a]	0.41[b]	0.55[a]	0.49[a]
Aspect	0.08	0.02	0.07	0.02	-0.02
Heat Load Index	-0.30[e]	-0.52[a]	-0.35	-0.48[a]	-0.43[b]
Cheatgrass cover Pre-fire	-0.51[a]	-0.45[b]	-0.42[b]	-0.38[c]	-0.37[c]
2001	-0.05	-0.22	-0.22	-0.12	-0.23
2002	-0.28	-0.37[c]	-0.31[d]	-0.28	-0.36[c]
2003	-0.29	-0.42[b]	-0.33[d]	-0.34[c]	-0.38[c]
2004	-0.40[b]	-0.53[a]	-0.45[b]	-0.48[a]	-0.51[a]

Table 1.25. Mortality and survival densities (tussocks/m2) of large bunchgrasses ($\geq$ 5.0 cm diam.) in survival plots on the ALE Reserve, 2001.

Habitat Type	n	Density (tussocks/m2) ± SD									
		Dead		Bluebunch wheatgrass		Idaho fescue		needle-&-thread		Total Density	
Sagebrush and Winterfat Shrublands	6	2.1	(0.8)	0.4	(1.0)	0.0	(0.0)	0.0	(0.0)	2.5	(1.0)
Agropyron and *Stipa* Grasslands	20	0.9	(1.0)	2.1	(1.8)	0.0	(0.0)	0.5	(1.2)	3.9	(2.2)
Threetip Sagebrush/ Fescue Habitats	10	0.6	(1.1)	3.8	(2.5)	5.2	(4.7)	0.5	(1.6)	12.6	(6.3)
All habitats	36	1.0	(1.1)	2.3	(2.2)	1.5	(3.4)	0.3	(1.1)	6.1	(5.2)

Native Forbs. Native perennial forbs were recorded in all 70 vegetation plots surveyed in 2004. Percent cover of native perennial forbs ranged from 0.5% to over 30.0% in vegetation plots, with greater than 10.0% cover recorded in 30 plots and less than 5.0% cover recorded in 28 plots.

Percent cover, percent frequency, and species richness of native forbs in ALE vegetation plots all matched or exceeded pre-fire values in 2004. Mean percent cover of perennial forbs declined following the 24 Command Fire; however by 2004 mean percent cover (8.8% ± 7.1 SD) was statistically similar to pre-fire values (8.4% ± 9.5 SD; P = 0.619; Table 1.26a). Mean percent frequency of native perennial forbs in 2004 (75.7% ± 21.7 SD) was greater than pre-fire values (66.5% ± 25.4 SD; P = 0.095; Table 1.26b). Mean species richness of native perennial forbs also increased, from 5.6 species/ plot (± 3.8 SD) prior to the fire to 7.3 species/ plot (± 3.8 SD; P < 0.0001) in 2004 (Table 1.26c). Richness tended to be higher in Biodiversity Plots than in transect-based plot types, although the differences between pre-fire and 2004 values were significant for both types of plots.

Mean percent cover (9.1% ± 5.7 SD), frequency (78.3% ± 18.5 SD), and richness (5.0 species/ plot ± 1.5 SD) of native perennial forbs in plots formerly dominated by big sagebrush or winterfat were greater in 2004 than at any other time during the course of this study, and differences between pre-fire and 2004 values were significant for each of these parameters (P < 0.001; Table 1.26). The highest values for all three parameters occurred in sandy habitats.

Percent cover of native perennial forbs was moderately correlated with elevation (r = 0.55; P = 0.0001), species richness of all native perennials (r = 0.53; P = 0.0001), and percent cover of all native perennials (r = 0.40; P = 0.0001; Table 1.27a). Percent cover of native perennial forbs has been moderately to strongly correlated with each of these variables within years during all the years of this study. Correlations within 2004 data represented the lowest correlations in these series in each case. Correlations between percent cover of native perennial grasses and percent cover of cheatgrass tended to be weak.

Species richness of native perennial forbs in 2004 was strongly correlated with elevation (r = 0.83; P = 0.0001) and slope (r = 0.73; P = 0.0001)) and was moderately correlated with heat load index (r = 0.67; P = 0.0001). The correlation between perennial forb richness and each of these variables has been relatively strong during each year of this study. Perennial forb richness was also strongly correlated within years with overall native perennial richness (r = 0.91 to 0.96; P = 0.0001) indicating the important role this component plays in overall vascular plant species richness.

Native perennial forb richness was moderately correlated with percent cover of all native perennials (r = 0.58; P = 0.0001) and with percent cover of large, warm season perennial bunchgrasses (r = 0.44; P = 0.001) in 2004. Forb richness has been consistently correlated within years with these variables throughout the years of this study, although correlations with both variables were strongest in 2001. Perennial forb richness was also moderately negatively correlated with percent cover of cheatgrass in 2004 (r = - 0.43; P = 0.001).

Percent cover of individual perennial forb species was typically no more than one or two percent and was very rarely as much as 10.0% in any vegetation plot at any time

Table. 1.26. Mean percent cover (a), frequency (b), and species richness (c) of native forbs on ALE Reserve vegetation plots, pre-fire-2004. All sites n = 70; Biodiversity Plots n = 33; All Transects n = 37). ANOVA *P* values are results of single factor repeated measures ANOVA. *t* test *P* values are the results of comparisons of pre-fire and 2004 values using paired-sample *t* tests.

a.	Percent Cover (± SD)										ANOVA	*t* test
	Pre-fire		2001		2002		2003		2004		*P*	*P*
All Native Forbs - All Sites	9.9	(9.8)	6.7	(10.2)	8.9	(11.1)	10.8	(7.4)	11.1	(7.2)	< 0.0001	0.275
Perennials												
All sites	8.4	(9.5)	5.5	(7.4)	8.1	(8.8)	7.7	(7.3)	8.8	(7.1)	< 0.0001	0.619
Biodiversity Plots	7.1	(9.2)	6.3	(9.4)	8.7	(9.5)	7.0	(7.4)	7.8	(7.7)	0.186	0.527
All Transects	9.5	(9.8)	4.8	(5.0)	7.5	(8.2)	9.3	(7.4)	9.8	(6.4)	< 0.0001	0.862
Big Sagebrush & Winterfat Plots												
All Sites	3.3	(4.0)	1.2	(1.1)	3.4	(3.5)	5.2	(3.5)	9.1	(5.7)	< 0.0001	< 0.001
All Transects	4.1	(5.0)	1.1	(1.1)	2.5	(1.6)	6.0	(3.6)	11.1	(4.6)	< 0.0001	< 0.001
Silt-Loam Soils	3.6	(4.6)	1.3	(1.0)	3.7	(3.9)	4.6	(2.6)	8.8	(6.2)	< 0.0001	0.007
Sandy soils	2.4	(1.0)	0.9	(1.4)	2.6	(2.1)	7.0	(5.3)	10.0	(4.4)	< 0.0001	0.012
Annuals												
All sites	1.6	(2.4)	1.3	(1.5)	2.1	(2.2)	3.5	(4.1)	2.3	(2.5)	< 0.0001	0.060
Biodiversity Plots	0.5	(0.8)	0.9	(0.7)	1.4	(1.0)	2.7	(3.9)	2.1	(2.6)	< 0.0001	0.002
All Transects	2.5	(2.9)	1.6	(1.9)	2.8	(2.7)	4.2	(4.2)	2.4	(2.3)	< 0.001	0.851
Big Sagebrush & Winterfat Plots												
All Sites	1.1	(1.7)	0.8	(0.6)	3.5	(3.0)	6.8	(4.3)	3.3	(2.8)	< 0.0001	< 0.001
All Transects	1.7	(1.9)	0.9	(0.7)	4.7	(3.3)	8.0	(4.6)	3.2	(1.8)	< 0.0001	0.021
Silt-Loam Soils	1.1	(1.9)	0.7	(0.6)	3.5	(3.2)	7.3	(4.6)	3.7	(3.1)	< 0.0001	0.011
Sandy Soils	1.1	(0.5)	1.0	(0.6)	3.5	(2.8)	5.1	(3.1)	2.3	(1.2)	0.004	0.057

Continued

Table 1.26 (continued). Mean percent cover (a), frequency (b), and species richness (c) of native forbs on ALE Reserve vegetation plots, pre-fire-2004. All sites n = 70; Biodiversity Plots n = 33; All Transects n = 37). ANOVA *P* values are results of single factor repeated measures ANOVA. *t* test *P* values are the results of comparisons of pre-fire and 2004 values using paired-sample *t* tests.

b.	Percent Frequency (± SD)										ANOVA	*t* test
	Pre-fire		2001		2002		2003		2004		*P*	*P*
Perennials												
All Transects	66.5	(25.4)	64.9	(31.9)	54.6	(30.8)	70.5	(21.5)	75.7	(21.7)	< 0.001	0.095
Big Sagebrush Transects												
All Transects	42.1	(17.4)	24.6	(15.0)	25.9	(19.4)	66.7	(20.5)	78.3	(18.5)	< 0.0001	< 0.001
Silt-Loam Soils	34.3	(18.8)	25.6	(11.9)	23.2	(8.4)	54.6	(23.8)	62.0	(26.9)	< 0.0001	0.006
Sandy soils	53.0	(6.7)	24.0	(17.1)	23.0	(25.9)	69.0	(25.1)	84.0	(22.5)	< 0.0001	0.030
Annuals												
All Transects	42.4	(30.6)	34.3	(28.9)	57.6	(27.9)	62.0	(30.1)	67.0	(26.1)	< 0.0001	< 0.001
Big Sagebrush Transects												
All Transects	39.6	(22.8)	15.8	(14.0)	63.3	(24.3)	87.1	(14.5)	82.9	(15.7)	< 0.001	< 0.001
Silt-Loam Soils	36.4	(30.0)	16.7	(9.6)	60.2	(25.0)	73.3	(32.8)	69.0	(29.8)	< 0.0001	0.004
Sandy Soils	44.0	(6.5)	19.0	(16.4)	44.0	(25.1)	79.0	(19.2)	73.0	(19.2)	< 0.001	0.043

Continued

Table 1.26 (continued). Mean percent cover (a), frequency (b), and species richness (c) of native forbs on ALE Reserve vegetation plots, pre-fire-2004. All sites n = 70; Biodiversity Plots n = 33; All Transects n = 37). ANOVA *P* values are results of single factor repeated measures ANOVA. *t* test *P* values are the results of comparisons of pre-fire and 2004 values using paired-sample *t* tests.

c.	Species Richness (± SD)										ANOVA	*t* test
	Pre-fire		2001		2002		2003		2004		*P*	*P*
All Native Forbs -- All Sites	7.3	(3.9)	9.2	(5.3)	9.2	(4.8)	9.7	(4.5)	11.6	(6.2)	< 0.0001	< 0.0001
Perennials												
All sites	5.6	(3.8)	7.2	(5.0)	6.7	(4.8)	6.3	(3.7)	7.3	(3.8)	< 0.0001	< 0.0001
Biodiversity Plots	6.4	(4.0)	8.1	(5.2)	8.1	(5.2)	6.8	(4.3)	7.8	(4.5)	< 0.0001	0.004
All Transects	5.0	(3.5)	6.4	(4.7)	5.4	(4.1)	5.9	(3.0)	6.7	(3.0)	< 0.001	< 0.0001
Big Sagebrush & Winterfat Plots												
All Sites	2.8	(1.7)	2.7	(1.9)	3.2	(1.9)	4.3	(1.4)	5.0	(1.5)	< 0.0001	< 0.001
All Transects	2.6	(1.7)	1.8	(1.1)	2.5	(1.7)	4.6	(1.0)	5.2	(1.7)	< 0.0001	0.002
Silt-Loam Soils	2.9	(1.9)	3.1	(2.0)	3.6	(1.8)	4.1	(1.4)	4.7	(1.3)	0.002	0.006
Sandy soils	2.2	(1.3)	1.4	(1.1)	1.8	(1.8)	4.8	(1.3)	5.8	(1.9)	< 0.001	0.009
Annuals												
All sites	1.6	(1.7)	2.0	(1.4)	2.6	(1.2)	3.4	(1.2)	4.4	(5.1)	< 0.0001	< 0.0001
Biodiversity Plots	0.7	(1.2)	1.8	(1.3)	2.2	(1.1)	2.7	(1.7)	4.3	(7.3)	< 0.001	0.006
All Transects	2.5	(1.6)	2.2	(1.5)	3.0	(1.2)	4.0	(1.9)	4.4	(1.4)	< 0.0001	< 0.0001
Big Sagebrush & Winterfat Plots												
All Sites	1.7	(1.7)	1.4	(1.1)	2.9	(1.3)	4.4	(1.8)	4.1	(1.5)	< 0.0001	< 0.0001
All Transects	2.5	(1.7)	1.1	(1.1)	3.2	(1.5)	5.3	(1.6)	4.8	(1.4)	< 0.0001	< 0.0001
Silt-Loam Soils	1.5	(1.9)	1.5	(1.2)	2.9	(1.1)	4.0	(2.0)	4.0	(1.5)	< 0.0001	< 0.0001
Sandy Soils	2.2	(0.4)	1.0	(0.7)	3.0	(1.9)	5.4	(0.9)	4.8	(1.6)	< 0.0001	0.014

Table 1.27. Correlation coefficients of community and environmental variables with percent cover and richness of native forbs in vegetation plots on the ALE Reserve, pre-fire-2004: (a) perennials; (b) annuals. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

a.		Native Perennial Forbs									
		Percent Cover					Species Richness				
		Pre-fire	2001	2002	2003	2004	Pre-fire	2001	2002	2003	2004
Pre-Fire Shrub Cover		-0.13	-0.17	-0.06	0.03	0.24	-0.31[d]	-0.38[c]	-0.27	-0.18	-0.18
Litter		-0.16	-0.16	-0.17	-0.23	-0.21	-0.28	-0.25	-0.36[c]	-0.40[b]	-0.30[e]
Elevation		0.67[a]	0.69[a]	0.78[a]	0.77[a]	0.55[a]	0.82[a]	0.89[a]	0.85[a]	0.84[a]	0.83[a]
Fire Severity		-0.39[b]	-0.41[b]	-0.34	-0.26	-0.07	-0.57[a]	-0.59[a]	-0.50[a]	-0.42[b]	-0.39
Slope		0.38[c]	0.53[a]	0.51[a]	0.51[a]	0.35[c]	0.76[a]	0.78[a]	0.80[a]	0.70[a]	0.73[a]
Aspect		0.05	0.13	0.13	0.17	0.09	0.12	0.12	0.09	0.00	0.11
Heat Load Index		-0.30[e]	-0.44[b]	-0.44[b]	-0.46[a]	-0.30[e]	-0.67[a]	-0.71[a]	-0.74[a]	-0.67[a]	-0.67[a]
Cheatgrass cover	Pre-fire	-0.12	-0.20	-0.23	-0.18	-0.15	-0.38[c]	-0.37[c]	-0.38[c]	-0.27	-0.27
	2001	-0.11	-0.03	-0.19	-0.24	-0.22	-0.15	-0.22	-0.21	-0.25	-0.24
	2002	-0.22	-0.22	-0.34[c]	-0.33[c]	-0.32	-0.36[c]	-0.41[b]	-0.37[c]	-0.38[c]	-0.38[c]
	2003	-0.28	-0.27	-0.41[b]	-0.39[b]	-0.38[c]	-0.39[b]	-0.44[b]	-0.44[b]	-0.43[b]	-0.43[b]
	2004	-0.32[d]	-0.31	-0.40[b]	-0.36[c]	-0.36[c]	-0.47[a]	-0.51[a]	-0.47[a]	-0.46[a]	-0.43[b]
Perennial Grass Cover	Pre-fire	0.16	0.24	0.35[c]	0.23	0.04	0.47[a]	0.57[a]	0.60[a]	0.51[a]	0.48[a]
	2001	0.04	0.25	0.25	0.09	0.04	0.43[b]	0.56[a]	0.52[a]	0.34[c]	0.48[a]
	2002	0.09	0.18	0.15	0.00	-0.14	0.40[b]	0.49[a]	0.37[c]	0.19	0.29
	2003	0.17	0.26	0.35[c]	0.10	-0.08	0.39[b]	0.53[a]	0.45[a]	0.27	0.35[c]
	2004	0.20	0.27	0.27	0.08	-0.05	0.43[b]	0.54[a]	0.49[a]	0.36[c]	0.39[b]
Large Perennial Grass Cover	Pre-fire	0.11	0.26	0.38	0.24	0.04	0.50[a]	0.62[a]	0.63[a]	0.57[a]	0.56[a]
	2001	0.12	0.32[d]	0.34[c]	0.23	0.10	0.54[a]	0.60[a]	0.62[a]	0.56[a]	0.62[a]
	2002	0.21	0.32[d]	0.39[b]	0.26	0.06	0.55[a]	0.59[a]	0.56[a]	0.48[a]	0.53[a]
	2003	0.23	0.33[d]	0.47[a]	0.27	0.01	0.52[a]	0.61[a]	0.58[a]	0.49[a]	0.52[a]
	2004	0.24	0.30[e]	0.33[c]	0.15	-0.03	0.48[a]	0.53[a]	0.51[a]	0.46[a]	0.44[b]
Native Perennial Cover	Pre-fire	0.51[a]	0.46[a]	0.56[a]	0.51[a]	0.39[b]	0.56[a]	0.59	0.65	0.64[a]	0.57[a]
	2001	0.37[c]	0.64[a]	0.55[a]	0.41[b]	0.34[c]	0.67[a]	0.77[a]	0.71[a]	0.56[a]	0.68[a]
	2002	0.43[b]	0.56[a]	0.62[a]	0.44[b]	0.26	0.66[a]	0.77[a]	0.66[a]	0.49[a]	0.55[a]
	2003	0.45[a]	0.56[a]	0.64[a]	0.52[a]	0.30[e]	0.66[a]	0.77[a]	0.68[a]	0.54[a]	0.60[a]
	2004	0.45[a]	0.55[a]	0.52[a]	0.42[b]	0.40[b]	0.63[a]	0.72[a]	0.65[a]	0.54[a]	0.58[a]
Native Perennial Richness	Pre-fire	0.65[a]	0.76[a]	0.65[a]	0.68[a]	0.51[a]	0.94[a]	0.83[a]	0.79[a]	0.71[a]	0.73[a]
	2001	0.59[a]	0.75[a]	0.71[a]	0.66[a]	0.48[a]	0.89[a]	0.96[a]	0.89[a]	0.77[a]	0.83[a]
	2002	0.56[a]	0.68[a]	0.68[a]	0.64[a]	0.44[b]	0.85[a]	0.90[a]	0.96[a]	0.81[a]	0.84[a]
	2003	0.59[a]	0.68[a]	0.63[a]	0.64[a]	0.48[a]	0.81[a]	0.85[a]	0.89[a]	0.91[a]	0.85[a]
	2004	0.57[a]	0.66[a]	0.65[a]	0.69[a]	0.53[a]	0.84[a]	0.89[a]	0.91[a]	0.87[a]	0.94[a]

Continued

Table 1.27 (continued). Correlation coefficients of community and environmental variables with percent cover and richness of native forbs in vegetation plots on the ALE Reserve, pre-fire-2004: (a) perennials; (b) annuals. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

b.		Native Annual Forbs									
		Percent Cover					Species Richness				
		Pre-fire	2001	2002	2003	2004	Pre-fire	2001	2002	2003	2004
Pre-Fire Shrub Cover		-0.25	-0.17	0.52^{a}	0.49^{a}	0.20	-0.09	-0.27	0.19	0.38^{c}	0.28
Litter		0.43^{b}	0.38^{c}	0.22	0.19	0.35^{c}	0.30^{e}	0.25	0.03	-0.03	0.21
Elevation		-0.14	0.11	-0.11	-0.37^{c}	-0.25	-0.11	-0.01	-0.03	-0.03	-0.01
Fire Severity		-0.05	-0.08	0.44^{b}	0.49^{a}	0.33^{d}	-0.02	-0.19	0.14	0.24	0.16
Slope		-0.08	0.17	-0.13	-0.31^{d}	-0.24	-0.02	-0.08	-0.02	-0.12	-0.08
Aspect		0.14	0.05	-0.17	-0.17	-0.14	0.16	0.26	-0.07	-0.06	-0.02
Heat Load Index		0.14	-0.09	0.10	0.25	0.20	0.12	0.13	0.02	0.13	0.08
Cheatgrass cover	Pre-fire	0.15	0.16	0.02	0.13	0.18	0.15	-0.01	0.12	0.11	0.16
	2001	0.25	0.31^{d}	0.08	0.27	0.47^{a}	0.11	0.22	-0.04	-0.14	-0.17
	2002	0.09	0.26	0.06	0.13	0.44^{b}	-0.04	0.18	0.01	-0.09	-0.12
	2003	0.17	0.28	0.04	0.14	0.40^{b}	-0.01	0.20	-0.02	-0.10	-0.12
	2004	0.14	0.30^{d}	0.11	0.24	0.55^{a}	-0.01	0.17	-0.01	-0.01	0.03
Perennial Grass	Pre-fire	-0.25	-0.17	-0.21	-0.26	-0.15	-0.40^{b}	-0.25	-0.22	-0.22	-0.21
Cover	2001	-0.20	-0.16	-0.43^{b}	-0.42^{b}	-0.34^{c}	-0.30^{e}	-0.12	-0.32^{d}	-0.37^{c}	-0.23
	2002	-0.10	-0.13	-0.53^{a}	-0.43^{b}	-0.32^{d}	-0.18	0.09	-0.30^{e}	-0.51^{a}	-0.32^{d}
	2003	-0.13	-0.20	-0.54^{a}	-0.54^{a}	-0.40^{b}	-0.25	0.00	-0.28	-0.48^{a}	-0.26
	2004	-0.15	-0.23	-0.51^{a}	-0.47^{a}	-0.37^{c}	-0.23	-0.09	-0.29	-0.44^{b}	-0.31^{d}
Large Perennial	Pre-fire	-0.22	-0.19	-0.28	-0.37^{c}	-0.26	-0.37^{c}	-0.25	-0.24	-0.19	-0.17
Grass	2001	-0.24	-0.18	-0.30	-0.38^{c}	-0.33^{d}	-0.34^{c}	-0.22	-0.35^{c}	-0.27	-0.22
Cover	2002	-0.17	-0.14	-0.37^{c}	-0.43^{b}	-0.38^{c}	-0.23	-0.12	-0.34^{c}	-0.31^{d}	-0.24
	2003	-0.14	-0.20	-0.40^{b}	-0.49^{a}	-0.40^{b}	-0.27	-0.08	-0.25	-0.34^{c}	-0.25
	2004	-0.13	-0.19	-0.37^{d}	-0.39^{b}	-0.32^{d}	-0.21	-0.10	-0.25	-0.29	-0.25
Native Perennial	Pre-fire	-0.32^{d}	-0.17	0.05	-0.14	-0.13	-0.32^{d}	-0.31^{d}	-0.01	0.04	-0.09
Cover	2001	-0.16	-0.03	-0.36^{c}	-0.47^{a}	-0.33^{c}	-0.16	-0.02	-0.21	-0.22	-0.14
	2002	-0.18	-0.08	-0.43^{b}	-0.48^{a}	-0.37^{c}	-0.15	0.10	-0.16	-0.34^{c}	-0.18
	2003	-0.21	-0.14	-0.44^{b}	-0.58^{a}	-0.47^{a}	-0.23	0.00	-0.12	-0.33^{d}	-0.16
	2004	-0.24	-0.18	-0.35^{c}	-0.46^{a}	-0.43^{b}	-0.21	-0.11	-0.14	-0.26	-0.21
Native Perennial	Pre-fire	0.06	0.16	-0.14	-0.32	-0.25	0.11	0.12	0.07	0.00	-0.04
Richness	2001	-0.10	0.12	-0.24	-0.45^{b}	-0.32^{d}	-0.10	0.15	0.01	-0.08	-0.10
	2002	-0.17	0.05	-0.19	-0.35^{c}	-0.29	-0.11	0.03	0.07	-0.01	-0.07
	2003	-0.08	0.02	-0.16	-0.35^{c}	-0.31^{d}	0.02	-0.02	0.09	0.14	-0.03
	2004	-0.17	0.04	-0.17	-0.37^{c}	-0.36^{c}	-0.09	0.00	0.03	0.04	-0.04

Table 1.28. Percent cover (a) and frequency (b) of selected common native perennial forb species in vegetation plots on the ALE Reserve, pre-fire - 2004. ANOVA *P* values are results of single factor repeated measures ANOVA. *t* test *P* values are the results of comparisons of pre-fire and 2004 values using paired-sample *t* tests.

a.	# Plots (n = 70)					Percent Cover (± SD; n = 70)					ANOVA	*t* test
Species	Pre-fire	2001	2002	2003	2004	Pre-fire	2001	2002	2003	2004	*P*	*P*
Achillea millefolium	27	34	32	47	40	0.5 (0.9)	0.6 (0.9)	0.8 (1.5)	0.7 (0.8)	0.5 (0.7)	0.219	----
Astragalus spp.	23	30	32	36	33	1.3 (2.2)	0.4 (0.8)	0.8 (1.6)	1.1 (2.1)	0.9 (2.1)	0.001	0.248
Balsamorhiza careyana	36	35	35	31	36	1.5 (2.5)	0.7 (1.1)	1.3 (2.2)	1.7 (2.8)	1.5 (2.4)	0.005	0.944
Crepis atrabarba	44	55	50	54	59	1.0 (1.5)	1.2 (1.4)	2.2 (3.6)	1.6 (2.1)	1.4 (1.7)	< 0.001	0.045
Machaeranthera canescens	11	5	18	34	35	0.1 (0.3)	< 0.1 (0.2)	0.2 (0.4)	0.6 (0.1)	1.4 (2.5)	< 0.0001	< 0.0001
Lupinus spp.	33	33	33	37	39	2.0 (4.0)	0.8 (1.3)	1.8 (3.3)	1.7 (3.1)	1.9 (2.8)	0.005	0.934

b.	Percent Frequency (± SD; n = 37)					ANOVA	*t* test
Species	Pre-fire	2001	2002	2003	2004	*P*	*P*
Achillea millefolium	4.7 (8.7)	5.3 (7.6)	5.9 (9.9)	28.1 (25.5)	10.7 (15.1)	< 0.0001	0.003
Astragalus spp.	5.5 (11.8)	5.1 (9.1)	5.1 (9.5)	8.5 (10.8)	6.6 (10.3)	0.226	----
Balsamorhiza careyana	9.3 (12.0)	7.0 (10.8)	6.2 (10.0)	6.5 (10.5)	9.7 (13.9)	0.103	0.810
Crepis atrabarba	21.5 (23.2)	31.6 (26.6)	27.0 (26.7)	26.8 (26.0)	36.5 (29.9)	< 0.0001	< 0.001
Machaeranthera canescens	0.7 (2.4)	0.4 (1.8)	3.1 (7.9)	14.5 (21.6)	20.3 (31.9)	< 0.0001	< 0.001
Lupinus spp.	20.3 (27.1)	20.4 (26.2)	14.3 (18.4)	13.8 (16.0)	19.6 (21.9)	0.029	0.855

during the course of this study. Between-year fluctuations in abundance of common native perennial forb species appeared to be individualistic. While some species declined in abundance the first year following the 24 Command Fire, most common native perennial forb species matched or exceeded their pre-fire abundance by 2004 (Table 1.28).

Percent cover of yarrow (*Achillea millefolium*) was statistically unchanged through the course of this study ($P = 0.219$). Percent frequency of yarrow spiked in 2003 and though lower in 2004 was still significantly greater than pre-fire values ($P = 0.003$). This spike in frequency in 2003 resulted from an abundance of seedlings which failed to contribute discernibly to aerial cover. Although yarrow was included in aerial seed mixes applied in December 2002 – January 2003 over a portion of the study area, yarrow seedling densities were also high outside of the treatment areas (see Section II, this volume).

Slender hawksbeard (*Crepis atrabarba*) was by far the most common perennial forb species, occurring in 59 of 70 vegetation plots and in more than 30.0% of transect microplots in 2004. Both cover and frequency of slender hawksbeard increased in the aftermath of the 24 Command Fire. Percent cover was highest in 2002, but cover in 2004 (1.4% ± 1.7 SD) was still significantly greater than pre-fire values ($P = 0.045$). Percent frequency of slender hawksbeard in 2004 (36.5% ± 29.9 SD) was the highest for this species in the course of the study ($P < 0.001$).

Both Carey's balsamroot (*Balsamorhiza careyana*) and perennial lupine species (*Lupinus laxiflorus*, *L. leucophyllus*, and *L. sulphureus*) exhibited patterns of decline in percent cover the first year following the 24 Command Fire. By 2004, however, both percent cover ($P > 0.900$) and frequency ($P > 0.800$) were statistically similar to pre-fire values for both of these taxa.

Percent cover of selected milkvetch species (*Astragalus caricinus* and *A. spaldingii*) also declined considerably in 2001, but again this species group recovered it's diminished abundance as early as 2003; in 2004 percent cover was similar to pre-fire values ($P = 0.248$). Percent frequency did not change ($P = 0.226$).

Hoary aster (*Machaeranthera canescens*) exhibited the most dramatic increase among the native forbs. This biennial to short-lived perennial became very common and relatively abundant in low elevation habitats in 2003 and 2004.

Two rare native perennial forbs were encountered during fieldwork in 2001-2004. Columbia milkvetch (*Astragalus columbianus*; USFWS Species of Concern; State Threatened. WNHP 1997) did not occur within any study plots, and areas where this species was encountered incidentally in 2002 (primarily along the gate 120 Road) were not revisited in 2003 or 2004, so no further observations were made. The other rare forb, Basalt milkvetch (*Astragalus conjunctus* var. *rickardii*; State Review; WNHP 1997), continued to be fairly common in vegetation plots at higher elevations on Rattlesnake Mt. and in the Rattlesnake Hills. Basalt milkvetch was recorded in 13 plots in in 2004 compared to 12 observations prior to the 24 Command Fire. The species has been recorded continuously since before the 24 Command Fire in 10 of these plots. All recorded occurrences were from sites above 1400 ft. in forb-rich associations classified as threetip sagebrush/ bluebunch wheatgrass or threetip sagebrush/ Idaho fescue communities by Wilderman (1994).

Percent cover of basalt milkvetch decreased from 1.7% (± 1.4 SD) prior to the 24 Command Fire to 0.7% (± 0.5 SD) in 2001 (Table 1.29). Cover in 2004 (1.5 % ± 1.3 SD) was statistically similar to pre-fire values ($P = 0.650$). Frequency along transects in 2004 (22.9% ± 23.4 SD) was greater than pre-fire frequency (7.9% ± 10.7 SD; $P = 0.086$).

Native annual forbs were recorded in 68 of 70 vegetation plots in 2004. Percent cover of native annuals ranged from 0.1% to more than 10.0%. Tarweed (*Amsinckia tessellata*) was by far the most common and abundant species. Tall willowherb (*Epilobium paniculatum*), tansy mustard (*Descurainia pinnata*), mountain dandelion (*Agoseris heterophylla*) and winged cryptantha (*Cryptantha pterocarya*) were also common. Combined percent cover of native annual forbs was less than 5.0% in 63 of the 68 plots where native annual forbs were recorded, with annual forbs accounting for 3.0% cover or less in 55 of those plots.

Mean percent cover of native annual forbs was greater in 2004 (2.3% ± 2.5 SD than pre-fire (1.6% ± 2.4 SD; $P = 0.60$; Table 1.26a). This difference is at least partly artificial, however. The methodology used in Biodiversity Plots is likely to have been relatively insensitive to the detection of annuals, particularly regarding pre-fire (1994) values, when objectives of the original survey were merely to characterize areas for the purposes of mapping vegetation. By comparison, percent cover in transects in 2004 was indistinguishable from pre-fire values ($P = 0.851$). Percent cover values recorded along transects were higher than in Biodiversity plots all years of the study. The difference was most evident in pre-fire values, and was much reduced in subsequent years. Interestingly, from 2001 to 2004 patterns of increasing and decreasing cover were similar between the two types of plots, although values for Biodiversity plots were always numerically lower.

Mean percent frequency of native annual forbs was greater in 2004 (67.0% ± 26.1 SD) compared to pre-fire values (42.4% ± 30.6 SD; $P < 0.001$; Table 1.26b).

Mean species richness of native annual forbs increased steadily over the course of this study, from 1.6 species/ plot (± 1.7 SD) prior to the 24 Command Fire to 4.4 species/ plot (± 5.1 SD) in 2004 (P < 0.0001; Table 1.26c). Although within-year differences between plot types were similar to those discussed above regarding percent cover, the trend of increasing richness was quite similar (and significant) regardless of methodology.

Mean percent cover (3.3% ± 2.8 SD), frequency (82.9% ± 15.7 SD), and richness (4.1 species/ plot ± 1.5 SD) of native annual forbs in plots formerly dominated by big sagebrush or winterfat were all at least somewhat reduced in 2004 compared to 2003 values (Table 1.26). Percent cover declined by more than 50% between years; however, overall values for each of these three parameters in 2004 were still significantly greater than pre-fire values ($P < 0.001$; Table 1.26).

Percent cover and species richness of native annual forbs in 2004 were generally poorly or not at all correlated with community and environmental variables (Table 1.27b). Only percent cover of cheatgrass (r = 0.55; $P < 0.0001$) and percent cover of native perennials (r = - 0.43) exhibited moderate correlations with either of these variables in 2004.

Table 1.29. Mean percent cover (n = 16) and frequency (n = 7) of basalt milkvetch (*Astragalus conjunctus* var. *rickardii*) in vegetation plots on the ALE Reserve, pre-fire – 2004. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests between pre-fire and 2004 values only.

	Percent Cover (± SD)	Percent Frequency (± SD)
Pre-fire	1.7 (1.4)	7.9 (10.7)
2001	0.7 (0.5)	12.1 (15.0)
2002	1.7 (1.3)	23.6 (18.2)
2003	1.6 (1.6)	20.0 (21.0)
2004	1.5 (1.3)	22.9 (23.4)
ANOVA *P*	.014	.052
t test *P*	.650	.086

Microbiotic Crust. Mean percent cover and frequency of microbiotic crust (MBC) declined following the 24 Command Fire. Large changes in overall MBC abundance were not detected immediately after the fire, possibly due to a combination of factors relating to sampling technique (see Discussion). In 2004, however, percent cover of MBC was estimated at 4.6% (± 6.9 SD), a decline of nearly 90 % compared to pre-fire values (37.8 % ± 19.3 SD; $P < 0.0001$; Table 1.30a). Estimated MBC cover in shrublands did exhibit dramatic declines (45.5 % or more) immediately following the fire. By 2004 MBC cover had decreased by 80 % in all plots where pre-fire cover of large shrubs was less than 5.0 %, and had decreased by more than 98 % in stands dominated by Wyoming big sagebrush over the same period.

As was the case with percent cover, large changes in mean percent frequency of MBC were not detected in 2001. By 2004, however, frequency had declined by more than 50 %, from 87.4 % (± 18.4 SD) to 40.1 % (± 31.5 SD; $P < 0.0001$; Table 1.30b). Frequency in plots with less than 5.0 % pre-fire shrub cover declined by approximately 30 % compared to pre-fire values, while frequency in shrub-dominated plots declined by more than 80 %.

Correlations between percent cover of MBC and abundance of cheatgrass have been presented above (Table 1.11). Post-fire cover of MBC was moderately to strongly correlated between years ($r = 0.50$ to 0.83; $P < 0.0001$) but correlations with pre-fire cover were weak (Table 1.31). Post-fire cover of MBC was somewhat weakly to moderately negatively correlated with pre-fire shrub cover ($r = -0.39$ to – 0.55;

Table 1.30. Mean percent cover (a) and frequency (b) of microbiotic crust (MBC) in ALE Reserve vegetation plots, pre-fire - 2004. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests between pre-fire and 2004 values only.

a.		Percent Cover (± SD)										ANOVA	*t* test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
All Plots	70	37.8	(19.3)	33.8	(20.4)	16.2	(12.1)	7.6	(9.9)	4.6	(6.9)	< 0.0001	< 0.0001
All Transects	37	43.3	(19.2)	31.1	(21.8)	13.8	(10.3)	5.3	(6.7)	3.9	(5.9)	< 0.0001	< 0.0001
Shrublands													
Big Sagebrush Plots	17	46.1	(15.8)	19.1	(15.8)	6.8	(6.9)	0.8	(1.0)	0.6	(0.8)	< 0.0001	< 0.0001
Big Sagebrush, Sandy Soils	5	34.2	(13.5)	9.2	(10.8)	0.8	(0.8)	0.5	(1.0)	0.3	(0.6)	< 0.0001	0.005
Plots > 5% shrubs	28	40.7	(16.8)	22.2	(15.1)	6.5	(5.9)	1.0	(1.3)	0.6	(0.8)	< 0.0001	< 0.0001
Plots < 5% shrubs	42	35.9	(20.7)	41.5	(19.9)	22.7	(10.9)	12.1	(10.6)	7.3	(7.9)	< 0.0001	< 0.0001

Continued

Table 1.30 (continued). Mean percent cover (a) and frequency (b) of microbiotic crust (MBC) in ALE Reserve vegetation plots, pre-fire - 2004. ANOVA *P* values are results of one-way repeat measures ANOVA; *t* test *P* values are results of two-tailed paired-sample *t* tests between pre-fire and 2004 values only.

b.		Percent Frequency (± SD)										ANOVA	*t* test
	n	Pre-fire		2001		2002		2003		2004		*P*	*P*
All Transects	37	87.4	(18.4)	86.1	(21.2)	63.1	(33.2)	41.2	(35.0)	40.1	(31.5)	< 0.0001	< 0.0001
Shrublands													
Big Sagebrush Plots - All	12	87.5	(14.4)	74.2	(28.8)	29.6	(23.1)	8.7	(13.2)	12.9	(11.4)	< 0.0001	< 0.0001
Big Sagebrush, Sandy Soils	5	74.0	(12.5)	62.0	(36.8)	10.0	(9.4)	6.0	(13.4)	4.0	(8.9)	< 0.0001	< 0.001
Plots > 5% shrubs	17	87.9	(13.5)	78.8	(25.1)	40.6	(27.3)	15.0	(16.7)	16.5	(13.0)	< 0.0001	< 0.0001
Plots < 5% shrubs	20	87.0	(22.1)	92.3	(15.3)	82.3	(25.0)	63.5	(30.8)	60.3	(28.4)	< 0.0001	< 0.001

Table 1.31. Correlation coefficients of community and environmental variables with percent cover of microbiotic crust (MBC) in vegetation plots on the ALE Reserve, pre-fire - 2004. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

	Percent Cover MBC				
	Pre-fire	2001	2002	2003	2004
MBC Cover Pre-fire	1.00				
2001	0.30[e]	1.00			
2002	0.19	0.74[a]	1.00		
2003	0.13	0.56[a]	0.78[a]	1.00	
2004	0.14	0.50[a]	0.66[a]	0.83[a]	1.00
Pre-Fire Shrub Cover	0.19	-0.46[a]	-0.55[a]	-0.47[a]	-0.39[b]
Litter	0.00	-0.20	-0.18	-0.27	-0.13
Elevation	-0.13	-0.08	-0.03	0.11	0.10
Fire Severity	0.24	-0.38[c]	-0.49[a]	-0.40[b]	-0.38[c]
Slope	-0.14	0.02	0.17	0.37[c]	0.37[c]
Aspect	-0.10	0.07	0.04	0.13	0.19
Heat Load Index	0.18	0.04	-0.14	-0.31[d]	-0.31[d]

$P < 0.0001$). and with fire severity (r = -0.38 to -0.49; $P < 0.005$). Correlations with other environmental and community variables were only weakly or not at all correlated.

Community development. Native perennial plant species were recorded in all 70 vegetation plots during each sample period, pre-fire - 2004. Percent cover of native perennial plants in 2004 ranged from just under 8.0% to over 70.0% in individual plots. Mean percent cover of native perennials declined by just over 55% following the 24 Command Fire (Table 1.32a). Although cover has increased each year since 2001, percent cover of native perennials in 2004 (42.8% ± 18.2 SD) was still significantly lower than pre-fire values (57.4% ± 24.5 SD; $P < 0.0001$). Species richness of native perennial plants ranged between four and 29 species/ plot. Mean species richness of native perennials has been higher since the 24 Command Fire. Richness in 2004 (12.9% ± 5.7) was the highest recorded in the study, and was significantly greater than pre-fire values (11.3% ± 5.7; $P < 0.001$).

Correlations between percent cover of cheatgrass and percent cover and richness of native perennials have been presented above (Table 1.5). Of the environmental variables, elevation exhibited the strongest correlation with percent cover of native perennial plants in 2004 (r = 0.64; $P < 0.0001$; Fig. 1.5a) and was consistently correlated near that value during the course of this study. Slope was also moderately correlated with native perennial cover in 2004 (r = 0.56; $P < 0.0001$) and previous years. Perennial cover was also moderately negatively correlated with fire severity (r = -0.56), heat load (r = -0.50) and percent litter (r = - 0.46; $P < 0.0001$) in 2004 (Table 1.33).

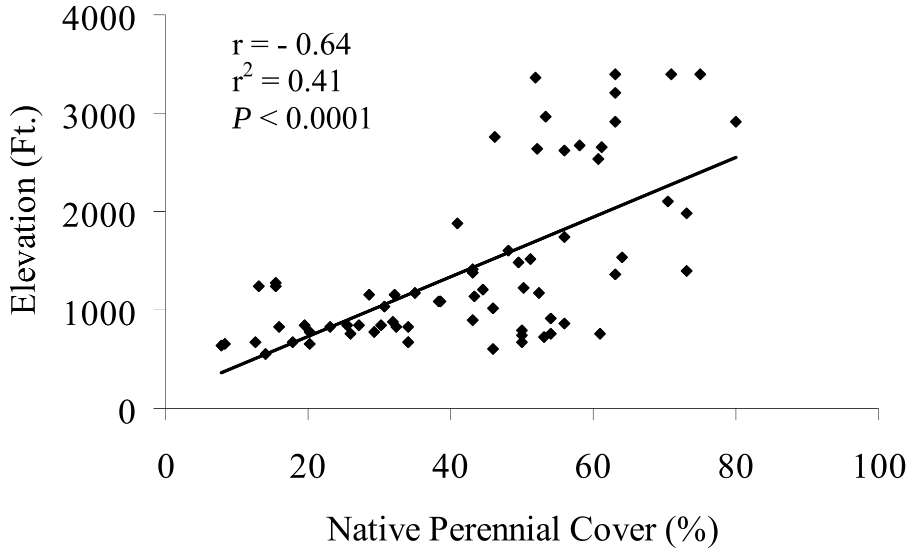
Table 1.32. Mean percent cover and species richness of all native perennial vascular plants in vegetation plots on the ALE Reserve, pre-fire – 2004: (a) all plots (n = 70); (b) big sagebrush and winterfat plots (n = 20); (c) all other plots (n = 50). ANOVA *P* values are results of one-way repeated measures ANOVA; *t*- test *P* values are results of two-tailed paired-sample *t* tests between pre-fire and 2004 values only.

a.	All Plots (± SD)										ANOVA	*t* test
	Pre-fire		2001		2002		2003		2004		*P*	*P*
Percent cover	57.4	(24.5)	25.8	(19.6)	37.1	(18.2)	41.7	(19.8)	42.8	(18.2)	< 0.0001	< 0.0001
Species richness	11.3	(5.7)	11.9	(7.1)	11.9	(6.6)	11.6	(5.3)	12.9	(5.7)	< 0.001	< 0.001

b.	Big Sagebrush & Winterfat Plots (± SD)										ANOVA	*t* test
	Pre-fire		2001		2002		2003		2004		*P*	*P*
Percent cover	47.7	(20.2)	8.9	(7.5)	18.0	(12.5)	21.7	(13.1)	27.4	(13.4)	< 0.0001	< 0.001
Species richness	7.5	(3.0)	5.5	(3.2)	7.0	(3.1)	8.1	(2.6)	9.1	(2.3)	< 0.0001	0.026

c.	All Other Plots (± SD)										ANOVA	*t* test
	Pre-fire		2001		2002		2003		2004		*P*	*P*
Percent cover	61.1	(25.0)	32.6	(18.8)	44.8	(14.1)	49.7	(16.0)	49.0	(16.1)	< 0.0001	< 0.0001
Species richness	12.8	(5.9)	14.5	(6.5)	13.8	(6.6)	13.0	(5.5)	14.5	(5.9)	< 0.0001	0.002

a.



b.

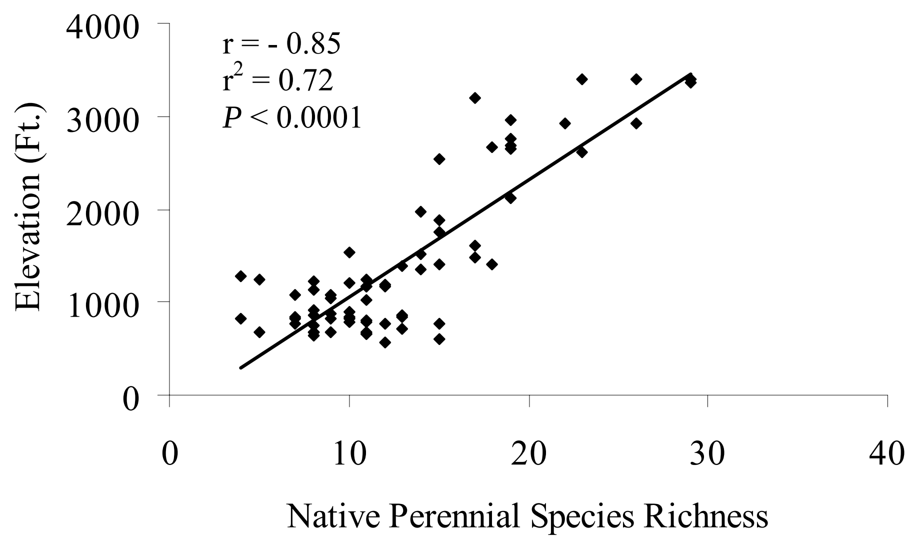


Fig. 1.5. Correlations between elevation and (a) percent cover and (b) richness of native perennial plant species in ALE Reserve vegetation plots, 2004.

Table 1.33. Correlation coefficients of community and environmental variables with percent cover and species richness of native perennial plants in vegetation plots on the ALE Reserve, pre-fire-2004. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$; e - $P < 0.05$.

		Native Perennial Plants									
		Total Percent Cover					Total Species Richness				
		Pre-fire	2001	2002	2003	2004	Pre-fire	2001	2002	2003	2004
Native Perennial Cover	Pre-fire	1									
	2001	0.52[a]	1								
	2002	0.55[a]	0.79[a]	1							
	2003	0.60[a]	0.80[a]	0.94[a]	1						
	2004	0.63[a]	0.85[a]	0.87[a]	0.91[a]	1					
Native Perennial Richness	Pre-fire	0.55[a]	0.65[a]	0.64[a]	0.63[a]	0.61[a]	1				
	2001	0.58[a]	0.82[a]	0.79[a]	0.80[a]	0.76[a]	0.88[a]	1			
	2002	0.60[a]	0.74[a]	0.70[a]	0.72[a]	0.68[a]	0.84[a]	0.92[a]	1		
	2003	0.57[a]	0.68[a]	0.60[a]	0.63[a]	0.63[a]	0.82[a]	0.87[a]	0.91[a]	1	
	2004	0.60[a]	0.72[a]	0.64[a]	0.69[a]	0.67[a]	0.82[a]	0.90[a]	0.92[a]	0.92[a]	1
Pre-Fire Shrub Cover		0.07	-0.43[b]	-0.51[a]	-0.46[a]	-0.37[c]	-0.28[a]	-0.39[b]	-0.31[d]	-0.30[e]	-0.24
Litter		-0.38[c]	-0.29	-0.32[d]	-0.39[b]	-0.46[a]	-0.29	-0.33[c]	-0.38[c]	-0.42[b]	-0.37[c]
Elevation		0.66[a]	0.63[a]	0.66[a]	0.70[a]	0.64[a]	0.76[a]	0.84[a]	0.81[a]	0.79[a]	0.85[a]
Fire Severity		-0.23	-0.55[a]	-0.66[a]	-0.62[a]	-0.56[a]	-0.47[a]	-0.57[a]	-0.50[a]	-0.47[a]	-0.44[b]
Slope		0.46[a]	0.69[a]	0.59[a]	0.59[a]	0.56[a]	0.73[a]	0.79[a]	0.82[a]	0.78[a]	0.79[a]
Aspect		-0.12	0.13	0.08	0.14	0.02	0.14	0.13	0.11	0.09	0.10
Heat Load Index		-0.45[a]	-0.62[a]	-0.53[a]	-0.52[a]	-0.50[a]	-0.63[a]	-0.70[a]	-0.73[a]	-0.71[a]	-0.71[a]

Both elevation (Fig. 1.5b) and slope were strongly correlated with native perennial species richness throughout the course of this study (r $\geq$ 0.73; P < 0.0001). In 2004, plots with > 19 native perennial species all occurred above 2100 ft. (640 m) elevation, while plots with fewer than 10 native perennial species all occurred below 1300 ft. (396 m). Heat load was strongly negatively correlated with richness during the four post-fire years (r $\geq$ - 0.70; P < 0.0001). Fire severity was moderately negatively correlated with perennial richness through the course of the study, and was the only other environmental variable moderately correlated with richness in 2004 (r = - 0.44; P < 0.001).

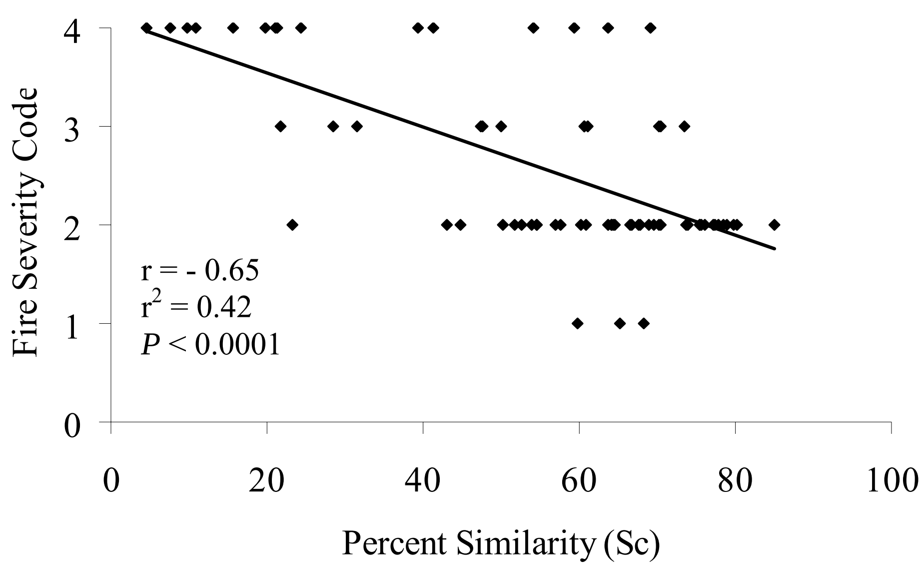
Percent cover of native perennials in big sagebrush and winterfat stands declined by more than 80% following the 24 Command Fire, compared to a reduction in cover of less than 50% in all other plots (Table 1.32b, c). Although percent cover in sagebrush and winterfat shrublands more than tripled between 2001 and 2004, the most recent value for native perennial cover (27.4% ± 13.4 SD) was still less than 60% of pre-fire values (P < 0.0001). By comparison, 2004 percent cover in all other plots (49.0% ± 16.1 SD) was more than 80% of pre-fire values, although this figure was still significantly lower than pre-fire values (P < 0.0001; Table 1.32c). Overall perennial species richness in sagebrush and winterfat shrublands declined by two species/ plot in 2001; However, richness has recovered progressively since 2001, and richness in 2004 (9.1 species/ plot ± 2.3 SD) was greater than pre-fire values (P = 0.026; Table 1.32b).

Sorensen's coefficient of similarity (***Sc***) for all perennial plant species ranged from 4.6% to 84.9%. Twenty of 70 plots had coefficients of 50.0% or lower. Plots where ***Sc*** $\leq$ 50.0% included 13 of 17 Wyoming big sagebrush stands. The eight lowest scores (***Sc*** $\leq$ 21.4%) and 12 of the 15 plots with the lowest *S*c (***Sc*** $\leq$ 41.2%) were located in Wyoming big sagebrush stands. Three of five plots dominated by needle and thread (Wildermand 1994), three bluebunch wheatgrass stands, and a single threetip sagebrush plot completed this group. Plots with greater than 50.0% similarity were mostly middle and higher elevation grasslands. Winterfat plots (n = 3; ***Sc*** 60.6 -70.2%) and the remaining threetip sagebrush stands were also scattered throughout this group.

Sc with large shrub cover removed from the calculation was somewhat higher than when calculated using all perennial plant species, ranging from 12.0% to 86.2%. Changes in ***Sc*** for individual plots ranged from 0.0% (21 plots) to 18.5% (absolute values) between the two methods. ***Sc*** changed by no more than 3.0% for 46 of 70 plots. These changes were in the direction of increasing similarity and resulted in meaningful changes in rank placement for 10-12 plots. Predictably, plots that had been dominated by big sagebrush and other large shrubs prior to the 24 command Fire exhibited the largest changes (1.8% - 18.5%) in ***Sc***. Still, big sagebrush plots accounted for 10 of the 13 lowest scores (***Sc*** = 12.0% - 41.6%), and 13 of 17 big sagebrush plots were in the lower 50% of scores.

The two methods of calculating ***Sc*** were very strongly correlated (r = 0.96; P < 0.001; Table 1.34). Correlations with environmental and community variables were generally stronger for ***Sc*** with shrub cover included in the calculation. Stand similarity with shrubs was most strongly negatively correlated with fire severity (r = - 0.65; P < 0.0001; Fig. 1.6a) and with pre-fire shrub cover (r = -0.61; P < 0.0001; Fig. 1.6b). ***Sc*** with shrub cover was also moderately correlated with pre-fire cover of perennial bunchgrasses (r = 0.57; P < 0.0001).

a.



b.

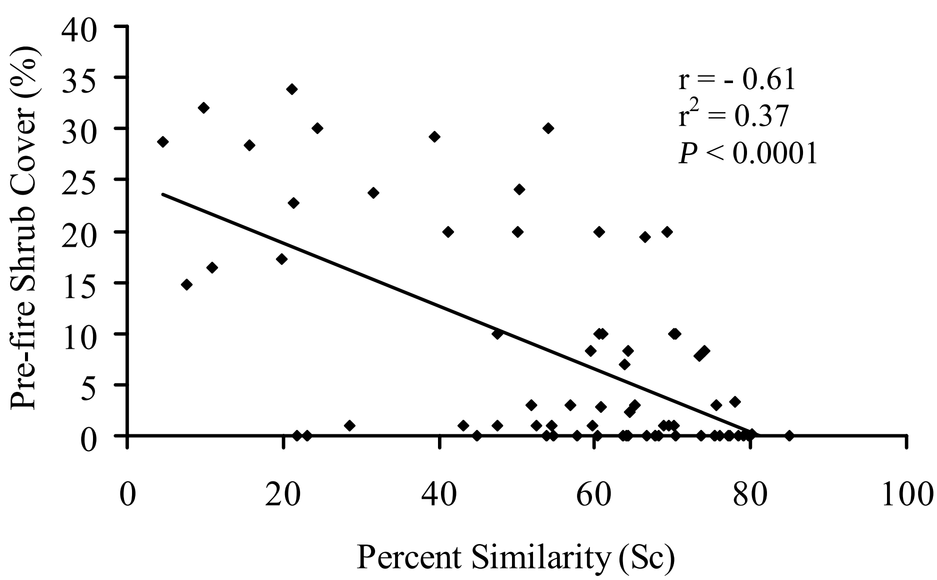


Fig. 1.6. Sorensen's coefficient of similarity (***Sc***) for native perennial vegetation in vegetation plots on the ALE Reserve, pre-fire vs. 2004, in relation to (a) observed fire severity , and (b) pre-fire cover of large shrubs. Estimated fire severity scale is from 1 (least severe) through 4 (most severe).

Table 1.34. Correlations of Sorensen's coefficient of similarity (***S*** c) for ALE Reserve vegetation plots with community and environmental variables. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.001$; c - $P < 0.005$; d - $P < 0.01$.

	S c	***S*** c w/o shrubs
S c	1.00	
S C w/o shrubs	0.96[a]	1.00
Cheatgrass cover, pre-fire	-0.31[d]	-0.28
Cheatgrass cover, 2004	-0.13	-0.07
Cheatgrass density, 2004	-0.13	-0.07
Pre-fire shrub cover	-0.61[a]	-0.44[b]
Litter 2004	-0.04	-0.03
Elevation	0.24	0.15
Fire Severity	-0.65[a]	-0.49[a]
Slope	0.22	0.14
Aspect	0.21	0.23
Heat Load Index	-0.16	-0.08
MBC, pre-fire	-0.21	-0.16
MBC, 2004	0.18	0.11
Native Perennial Cover, pre-fire	0.06	0.02
Native Perennial Cover, 2004	0.45[a]	0.37[c]
Native Perennial Grass Cover, pre-fire	0.31[d]	0.21
Native Perennial Grass Cover, 2004	0.57[a]	0.48[a]
Large Native Perennial Grass Cover, pre-fire	0.26	0.13
Large Native Perennial Grass Cover, 2004	0.46[a]	0.35[c]
Native Perennial Richness, pre-fire	0.32[d]	0.25
Native Perennial Richness, 2004	0.21	0.11

The first four axes of detrended correspondence analysis (DCA) account for 32.7% of the variance in the species composition data, with the first axis accounting for 15.7% (Table 1.35). The first axis, DCA-1, is nearly 3.5 half-changes long, reflecting the high species diversity associated with the relatively wide range of habits represented in the sample data (Fig. 1.7). DCA -1 appears to approximate an elevation gradient, with low elevation stands near the origin or left-hand side of the diagram and elevation of sites increasing generally from left to right. The second axis, DCA-2, is just over three half-changes long and appears to approximate a gradient in total shrub cover, with shrub cover decreasing from bottom to top across the diagram.

Wyoming big sagebrush shrublands in pre-fire condition are clustered in the lower left-hand corner of the diagram and have the lowest scores on both the first (DCA-1) and second (DCA-2) axes of the ordination. The five plots associated with BRMaP 20, big sagebrush-spiny hopsage plots on sandy soils, have the most extreme low scores. Post-fire (2004) scores for the same plots have just slightly higher scores along DCA-1 but are separated by one to two full half-changes from their pre-fire counterparts along DCA-2. Differences between other big sagebrush plot pairs are less extreme, but are still generally greater than differences for other habitat types. Differences between the pre- and post-fire composition of winterfat plots (samples 44, 47, and 50 near the upper left-hand corner of the diagram) were far less extreme than for big sagebrush plots.

A tight cluster of sites on the right-hand side of the diagram represents a mixture of grasslands and threetip sagebrush/ Idaho fescue communities. Site scores for these habitats range from middle to high scores along DCA-1 but are clustered within a narrow range along DCA-2. In general, pre- and post-fire conditions of grasslands and threetip sagebrush/ Idaho fescue plots are clustered comparatively closely together on the ordination diagram. Overall, pre-fire site scores are much more widely dispersed along both ordination axes than 2004 values, suggesting a coparative homogenization of vegetation among sites following the 24 Command Fire.

Table 1.35. Summary of axis scores from detrended correspondence analysis (DCA), ALE Reserve vegetation plots, pre-fire vs. 2004 data.

Axes →	1	2	3	4
Eigenvalues	.414	.218	.142	.090
Cumulative % variance of species data	15.7	23.9	29.3	32.7

Sum of all unconstrained eigenvalues 2.643

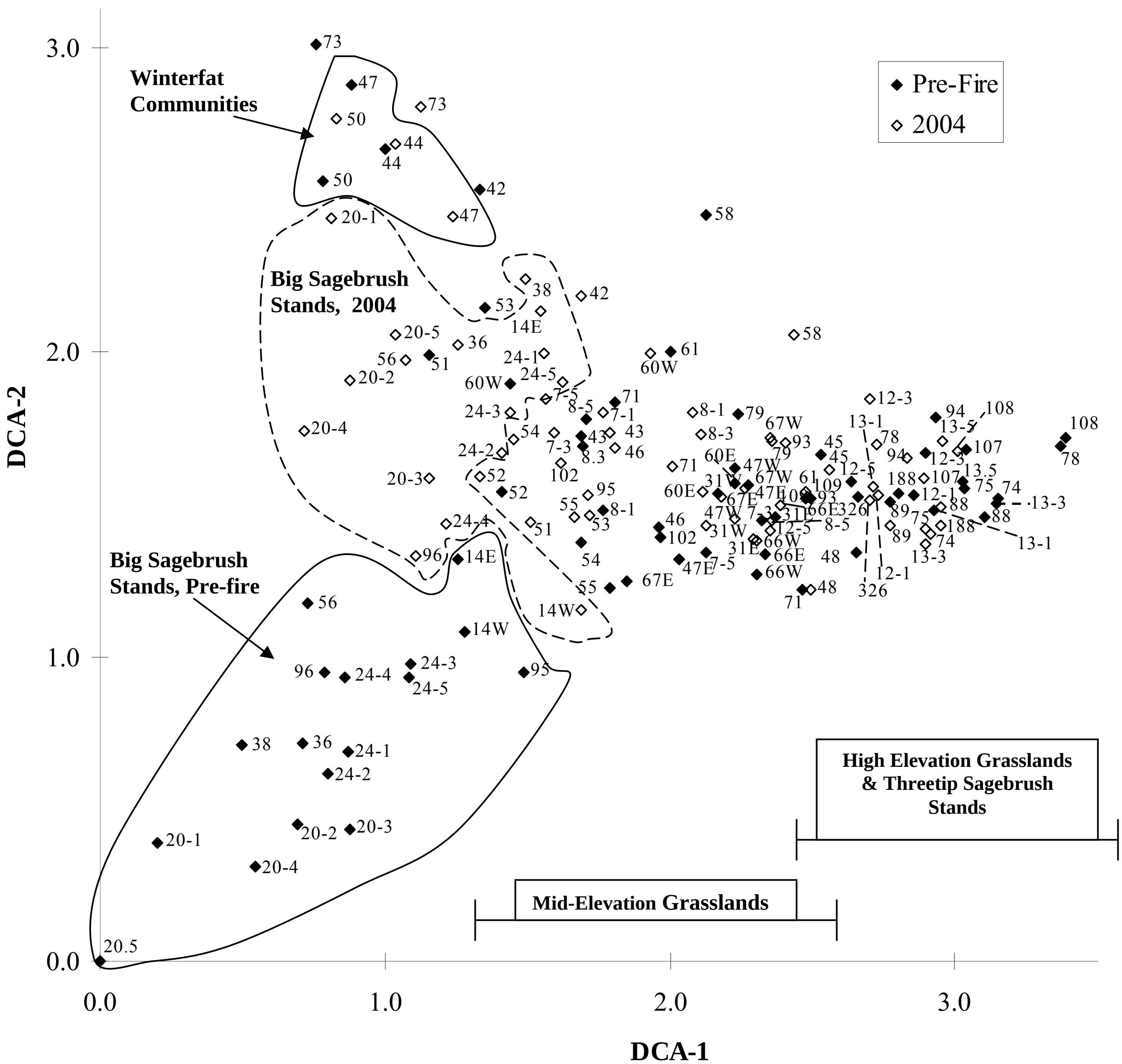


Fig. 1.7. Detrended correspondence analysis (DCA) of perennial vegetation on ALE Reserve study plots, pre-fire vs. 2004. Scatterplot of pre-fire (♦) and 2004 (◊) sample plots.

DISCUSSION

Invasive species. Cheatgrass abundance on the ALE Reserve varies from dense stands with few or any native perennial plant species (Fig. 1.8) to areas of low cheatgrass abundance where native habitats are still in relatively good condition. Cheatgrass abundance declined sharply in the year following the 24 Command Fire. At many sites, percent cover of cheatgrass had recovered to at or near pre-fire levels by 2002, while percent frequency already exceeded pre-fire values by that year. By 2004, overall cheatgrass cover was statistically similar to pre-fire levels. Overall frequency was greater for all types of plots, suggesting that the species was more evenly distributed within plots and was perhaps more ubiquitous than prior to the fire. Unfortunately, no pre-fire density data were available to allow for the same direct comparisons with this parameter. Close correlations between cheatgrass cover and density in all other years suggest that low densities in 2001 represented a considerable reduction compared to pre-fire conditions. Comparisons of densities between unburned refugia and adjacent burned areas support this conclusion. However, density values in both vegetation plots and Transition density plots in 2004 were the highest figures recorded between 2001 and 2004, though the difference between 2003 and 2004 values was significant only for the vegetation plots.

Our data indicate a pattern of cheatgrass response to wildfire similar to that described in shrub-steppe vegetation in a number of previous studies (for example Humphrey and Schupp 2001, Peters and Bunting 1994, Rickard et al. 1988, West and Hassan 1985, Young and Evans 1985, 1978, Evans and Young 1978). Wildfire destroys a portion of the cheatgrass seedbank and clears areas of the dense surface litter that provides the most favorable seedbed for cheatgrass germination and establishment. Very severe wildfires, usually associated with shrublands and the woody fuels that characterize them, may destroy nearly the entire cheatgrass seedbank within the affected area (Young et al. 1976). However, a percentage of viable cheatgrass seed inevitably survives. Due to the species' enormous reproductive potential, survival of a very small fraction of the original seedbank is sufficient to restore cheatgrass abundance to pre-fire levels within 2-4 years.

The spike in cheatgrass cover and frequency in Steppe-in-Time (SIT) plots on the ALE Reserve in 2000 may have been influenced by precipitation more than 40% above normal at Hanford during winter and spring of 2000 (HMS 2004)[1]. Cheatgrass abundance is likely to vary with precipitation or other moisture related factors, with abundance increasing during years of above-average precipitation, and with abundant spring precipitation is especially favorable for cheatgrass establishment (Blackshaw et al. 2001, Young and Longland 1996Young and Evans 1985, Young et al. 1969).

---

[1] The failure of cheatgrass cover in SIT plots in southern Benton County plots to track these increases on ALE in 2000 may be explained by local variability in precipitation. While the entire Lower Columbia Basin received higher-than-normal precipitation for winter and spring 2000 overall, only Hanford and Richland received significant precipitation during April of that year. The Hanford Meteorological Station and the Richland Station of the National Weather Service (NWS) recorded 0.57" (30% above normal) and 0.86" (56% above normal) respectively during April 2000, while NWS stations just to the south (Kennewick, Prosser, McNary Dam, Boardman, OR), all recorded less than 0.05" (92% - 97% below normal; HMS 2004, NWS 2004). Thus cheatgrass cohorts in areas south of the Hanford Site likely experienced drought during a critical period of development at the same time that sites on the ALE Reserve were benefiting from above-normal precipitation.

It is less clear how above-normal precipitation may have affected cheatgrass abundance in 2003 and 2004. During the critical December – April periods of 2003 and 2004 precipitation at Hanford has been 204.3% and 151.4% of normal, respectively (HMS 2004). However, below normal temperatures and prolonged snowcover at low elevations during winter 2003 – 2004 may have negatively influenced production and abundance of cheatgrass during the past year.

Abundance and recovery of cheatgrass were not uniform across the ALE landscape. Cheatgrass was strikingly less abundant in vegetation plots at elevations above 1400 ft. (427 m; Fig. 1.9). High elevation sites display a set of characteristics that may render them more resistant to the proliferation of cheatgrass. These sites support relatively productive native perennial plant communities that are considerably higher in total percent cover and species richness of native perennials than habitats at lower elevations. Higher precipitation and more moderate growing season temperatures in these habitats (Rotenberry et al. 1976, Hinds and Thorp 1969), enhanced by northerly to northeasterly aspects on moderate to steep slopes, may promote greater resilience in these more nearly closed stands in the aftermath of wildfires and other disturbances. The high percent cover of native perennial vegetation was correlated with reduced cover of cheatgrass, as has been noted in other recent studies (West and York 2002, Anderson and Inouye 2001), and may indicate a degree of competitive control of cheatgrass abundance. Native perennials appear to maintain dominance or to co-dominate with cheatgrass on more mesic exposures even at lower elevations on Hanford (Sauer and Rickard 1979, Rickard 1975, Rickard et al. 1973).

Characteristics of the winter environment at higher elevations may also contribute to reduced performance by cheatgrass. A critical advantage of this invasive winter annual is its capacity for root growth during the cool months of winter when soil moisture is less limiting and native perennial species are dormant (Harris 1967). The progressively lower temperatures and lengthening periods of persistent snow cover that are associated with increasing elevation probably reduce this competitive advantage. Over-winter mortality of fall-germinating seedling cohorts, reduced productivity of survivors, and delayed germination of spring-germinating seedling cohorts may all be significant consequences of winter conditions (Pierson and Mack 1990, Mack and Pyke 1984). An extreme example of winter conditions unfavorable to cheatgrass comes from January 2004, when snow cover was persistent on the Central Plateau at Hanford at only 600' (183 m) elevation for nearly the entire month of January (HMS 2004) and pocket gopher snow tunnel castings were observed at all elevations. Undoubtedly the duration of this snowpack was considerably longer above 2000'. Other environmental factors, including pH, soil organic matter, and total soil carbon also increase with elevation on ALE (Smith et al. 2002) and may play a role in overall community resistance to cheatgrass proliferation.

The moderate to high negative correlations of cheatgrass abundance with species richness of native perennials supports theories that species diversity in native communities confers resistance to invasion by alien species (DiTomasi 2000, Rosentreter 1994) . This position has been challenged in recent years (Anderson and Inouye 2001, Stohlgren et al. 1999, and others). The possible merits of this relationship aside, its reverse seems intuitive: that natural diversity may express itself more fully in areas where the abundance of invasive species is low, and more ecological niches are potentially available to native species.

At lower elevations, cheatgrass abundance remained lower than pre-fire values in 2004 where big sagebrush stands occurred on sandy soils (Fig. 1.10). Cheatgrass cover was still only 54 % of pre-fire values in these habitats four years after the 24 Command Fire, in contrast to other sites where cheatgrass cover had returned to pre-fire values much earlier. Cheatgrass abundance has been observed to respond positively to finer textured soils (Durham et al. 2001, Miller et al. 2001). The coarse texture of sandy soils allows moisture from winter storms to penetrate quickly through surface layers, rendering these sites less favorable for winter growth of cheatgrass seedlings than silt loam soils, which hold moisture nearer to the soil surface where it is more available to seedling roots. While recolonization is slower on these sites, high cheatgrass abundance in pre-fire data underscore this invader’s ability to dominate such sites over the long term, however.

The ecosystem-altering effects of cheatgrass have been well documented. Invasion of this Mediterranean annual into shrub-steppe communities initiates a suite of changes in critical ecosystem properties such as community structure, species diversity, and moisture and nutrient regimes. Cheatgrass exploits the niches previously occupied by native annual plant species, and outcompetes the seedlings of perennials (Brooks and Pyke 2001, Rosentreter 1994, Whisenant 1990, Young and Evans 1978, Harris 1977, 1967). Cheatgrass occupies the interspaces between perennial vascular plants and smothers microbiotic crusts (Belnap and Philips 2001, Belnap et al. 2001) while disrupting mycorrhizal associations (Wicklow-Howard 1994). Its winter annual habit alters seasonal patterns of production and adds unusually large amounts of dead above and belowground biomass to the system annually (Belnap and Phillips 2001). Copious additions of litter contribute to smothering effects aboveground and, coupled with below ground root decay, increase carbon: nitrogen ratios and ultimately reduce nitrogen availability in soils (Evans et al. 2001). Associated changes in the soil biota include a reduction in diversity of soil invertebrates and changes in diversity of soil microbes and fungi (Belnap and Phillips 2001).

The annual buildup of a continuous mat of dry litter contributes to increases in the frequency, extent, and severity of wildfires (USFS 2001, Brooks and Pyke 2001, Young and Evans 1985, 1978) which reinforces the trends outlined above. Wildfires contribute to the wholesale alteration of the ecosystem by altering the availability and distribution of nutrients, and by further reducing the vigor of surviving native perennials and contributing to the fragmentation of microbiotic crusts (Brooks and Pyke 2001, Belnap et al. 2001, Whisenant 1990). Areas dominated by cheatgrass are associated with declining wildlife habitat value (Brooks and Pyke 2001, Knick et al. 1999, Soll et al. 1999, Dobler 1994, Schuller et al. 1993, Groves and Steenhof 1988). The entire successional sequence may be attenuated into a brief episode of alien annual forb abundance followed by cheatgrass dominance (Young and Longland 1996). Recurring wildfires only serve to reduce the vigor of relict native perennials and reinforce this brief cycle. In many areas this cheatgrass-wildfire cycle has resulted in the conversion of extensive native shrub-steppe ecosystems to alien annual grasslands within a few decades (Monsen 1994a, Peters and Bunting 1994, Whisenant 1990, Young and Evans 1985). Once dominance has been achieved, cheatgrass has been observed to persist for many decades with no

Fig 1.8. Dense cheatgrass (*Bromus tectorum*) with Carey's balsamroot (*Balsamorhiza careyana*). Iowa Flats area, elevation ca. 700 ft. (215m).

a.

b.

Fig. 1.9, High elevation native plant communities on silt loam and stony silt loam soils exhibited high percent cover and richness of native perennial plant species, factors associated with lower percent cover of cheatgrass in ALE study plots: (a) bluebunch wheatgrass-Sandberg's bluegrass grassland on upper slopes of Rattlesnake mountain, ca. 2500 ft. (760 m); (b) forb-rich bluebunch wheatgrass-Idaho fescue community near the crest of Rattlesnake Mountain, ca. 3200 ft. (975 m).

a.

b.

Fig 1.10. Wyoming big sagebrush – spiny hopsage/ Sandberg's bluegrass shrubland on sandy soils (BRMaP Plot 20, Transect PC 1), elevation 640 ft. (195 m): (a) 2001; (b) 2004. No pre-fire image is available. Prior to 2000, percent cover of large shrubs and cheatgrass was greater than 20% and 35%, respectively, in this community. Percent cover of cheatgrass was still little more than 50% of pre-fire values four years after the 24 Command Fire. Substantial cover in 2004 was provided by the alien annual forbs tumble mustard and Russian thistle.

indication of natural recovery towards perennial dominated systems (Rickard and Sauer 1982, Daubenmire 1975).

Cheatgrass is almost ubiquitous on ALE. It is, of course, well established along roadsides, but invades surrounding lands along nearly every old rutted track or fireline, as well as along wildlife trails. Along with tumble mustard (*Sisymbrium altissimum*) and other non-native forbs, it has colonized and dominated nearly every landscape concavity, from deep washes to minor depressions, and in this way has made alien corridors into and surrounding the remaining perennial shrublands and grasslands. There are likely no places on ALE that are completely free of cheatgrass. Even the best remaining grasslands we have observed already contain occasional individuals or, here and there, a dense stand upon a gopher mound or other small natural disturbance. The presence of these small infestations likely serve as seed sources for increasing cheatgrass abundance in the surrounding landscape when fires or other disturbances create favorable conditions.

The ALE Reserve is host to a suite of alien annual forb species that is typical of western rangelands (Evans and Young 1970). Like cheatgrass, these alien forbs are common and widespread across all major habitat types on the Reserve. The abundance and distribution of alien annual forbs increased substantially following the 24 Command Fire. While overall cover and frequency of alien annual forbs were highest in 2003, abundance values for tumble mustard (*Sisymbrium altissimum*), Russian thistle (*Salsola kali*), and redstem filaree or storksbill (*Erodium cicutarium*) in 2004 were still significantly greater than pre-fire values, and the number of plots in which these species were recorded represented a 4- to 6-fold increase over pre-fire distributions.

These broadleaf annual weeds may dominate degraded rangelands for a brief period following wildfires or other disturbances, but typically decline within a few years, yielding to competition with cheatgrass and other species (Rickard et al. 1988, Daubenmire 1975, Evans and Young 1970). Interpreted in this light, the declines in abundance of both alien and native annual forbs observed in 2004 are not unexpected.

Some important invasive annuals are underrepresented in our records either due to seasonal phenology or because particular habitats were not sampled. Russian thistle achieves maximum development in late summer and our data records represent only seedling stages of this warm season annual. This species currently dominates some former big sagebrush stands at low to middle elevations especially on sandier soils. Storksbill is abundant along roadsides and on warmer aspects in disturbed draws, habitats where no vegetation reference plots were located. In some of these disturbed areas storksbill appears to vie with cheatgrass for dominance.

Roads and other corridors (Gelbard and Belnap 2003, Trombulak and Frissell 2000, Tyser and Worley 1992), grazing and associated water developments (Belsky and Gelbard 2000, Fontaine et al. 2004), wildfire (Pellant 1990, Young and Evans 1985, Bushey 1995), and homestead, agricultural, and other human development are well-known to promote the proliferation of cheatgrass and other invasive species (Carpenter and Murray 1999, Mack 1981, Stewart and Hull 1949). Lack of correlation between cheatgrass abundance and land use and disturbance measures in this study may be attributable to one or several of a number of factors, including sketchy or incomplete records of historic activities, insufficiently detailed fire history data and the diffuse impacts of grazing and other agricultural, military, and scientific activities away from established homesteads or facilities. Another such factor is the proliferation of roads on or surrounding ALE. There are more than 80 miles of currently active roads (both paved and unpaved) and nearly 60 miles of abandoned roads on the ALE Reserve, and the site is bordered on three sides by busy state highways. In addition to roads, nearly 20 miles of powerlines cross the Reserve (HRNM 2004). Few areas are more than 2.0 km from one of these features, so that the influence of any single factor may be difficult to tease out. It may also be that the relationships between these factors and cheatgrass abundance are not so linear as to be revealed by simple correlation.

Native species and communities. The 24 Command Fire burned with varying intensity as it swept across the Arid Lands Ecology Reserve (Interagency Fire Team 2000) and had varying effects upon the plant communities in its path.

The 24 Command Fire had substantial impacts at all structural levels within the big sagebrush stands on ALE (Fig. 1.11). Changes in stand structure, species abundance, and community composition were strongly evident four years after the wildfire and the impacts of these changes can be expected to affect ecosystem processes and functions in both the short term and for many years to come.

The dramatic loss of Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) in the 24 Command Fire continued a trend that has seen the majority of ALE shrublands reduced by fire over the past three to five decades (Rickard at all 1988, J.L. Downs pers. comm., T. Skinner pers. comm.) The 24 Command Fire consumed most of the remaining high quality, large, and contiguous stands of big sagebrush within the ALE Reserve. Portions of stands that escaped the fire, mainly north of Rattlesnake Springs and the Gate 118 Road, are small and fragmented. Both Wyoming big sagebrush and its important associate spiny hopsage (*Atriplex* = *Grayia spinosa*)were almost entirely removed from within the footprint of the fire. The loss of these keystone shrub species over wide areas on the ALE Reserve is exacerbated by the lack of significant natural sources of sagebrush seed over most of the affected area and by the vanishingly small rate of reproductive success of spiny hopsage in the Columbia Basin in recent decades.

The removal of these large deep-rooted shrubs alters important ecosystem functions such as water utilization and storage, resource allocation, biological production, and nutrient cycling (Link et al 1990, Bunting 1985, Cline et al. 1977, Harniss and Murray 1973) and reduces habitat value for large and small mammals, birds, and terrestrial invertebrates (Welch and Criddle 2003, Soll et al. 1999, Dobler 1994, H. Newsome pers. comm). Numerous vertebrate and invertebrate species feed upon big sagebrush, and sagebrush-obligate wildlife species such as the greater sage grouse (*Centrocerus urophasianus* – until recently a candidate for federal endangered species status, state threatened), sage sparrow (*Amphispeza belli* – state candidate), loggerhead shrike (*Lanius ludovicianus* – federal species of concern, state candidate), sage thrasher (*Oreoscoptes montanus* – state candidate) and others require big sagebrush habitat for shelter, nest sites, food, or habitat for prey species. Many other wildlife species, including the burrowing owl (*Athene cunicularia* -- federal species of concern, state candidate), the black-tailed jackrabbit (*Lepus californicus* – state candidate), and others are most common and abundant in the vicinity of big sagebrush stands (Newsome and

LaFramboise 2004, Welch and Criddle 2003, Wambolt 2001, Connelly et al. 2000, Soll et al. 1999, Dobler 1994, McCorquodale 1987).

These former Wyoming big sagebrush or big sagebrush – spiny hopsage habitats on the ALE Reserve will not recover without extensive and persistent restoration efforts. The loss of big sagebrush from the last extensive portions of its former range on the ALE Reserve represents the crossing of a threshold in ecosystem condition (Hemstrom et al. 2002, Stringham et al. 2001, Laycock 1991, Westoby et al. 1989). Even under the most favorable conditions, natural recovery of big sagebrush canopies may take 30 years or more (Wambolt et al. 2001, 1999, Nelle et al. 2000, Harniss and Murray 1973). The same factors that make unassisted recovery of Wyoming big sagebrush habitats unlikely – the semiarid climate of south central Washington, the high frequency of wildfires, and the potential for increase of cheatgrass and other invasive species – will present extreme challenges to the restoration and long-term maintenance of these critical habitats.

Big sagebrush is entirely dependent upon establishment from seed to recolonize areas following fire. The majority of big sagebrush seeds typically remain viable in the soil seed bank only through the spring of the year in which they are produced (Meyer 1994, Young and Evans 1989) and abundant recruitment from seedbanks following a summer wildfire is unlikely. However, a small but perhaps ecologically significant fraction of the big sagebrush seed bank may persist beyond this brief period (Booth 2002, Young and Evans 1989) and may account for the scattered big sagebrush juveniles observed in former shrublands on ALE these past few years.

Natural reestablishment of big sagebrush on ALE is further complicated by problems of seed dispersal from the few remaining naturally occurring sources. Seed production in surviving islands of Wyoming big sagebrush may contribute to the long-term recovery of the ALE shrublands (Longland and Bateman 2002). However, maximum dispersal of big sagebrush seeds from the parent plant is only 30 m (Welch 2004), and most seed produced by big sagebrush falls within only one meter of the parent (Young and Evans 1989, Bunting 1985). Much of the area formerly dominated by big sagebrush and severely burned during the 24 Command Fire is currently remote from significant stands of reproductive survivors

Although spiny hopsage has been reported to be capable of vegetative regeneration following fire (Daubenmire 1970), no evidence either of resprouting or of seedling establishment of this species has been observed on ALE in the four years following the 24 Command Fire. Resprouting of spiny hopsage appears to be extremely uncommon in the Columbia Basin (Simmons and Rickard 2003, Rickard and McShane 1984). Reproduction by this species from seed, while not unheard of at Hanford (J. Downs pers. comm., D. Salstrom pers. comm.), is rare enough to prompt concerns over potential local extirpation of this species in south central Washington (Simmons and Rickard 2003).

Impacts of the 24 Command Fire within shrublands on ALE were not limited to the canopy layer. All other structural layers of the big sagebrush plant community – hemishrubs, grasses, forbs, and microbiotic crusts – exhibited evidence of decline and mortality beyond that experienced in grasslands and threetip sagebrush communities. Multivariate analyses support this assessment. Sorenson's coefficient of similarity (***Sc***) and detrended correspondence analysis (DCA) clearly separated big sagebrush shrublands from the majority of grasslands and threetip sagebrush communities and indicated greater dissimilarity between pre-fire and post-fire composition in big sagebrush stands compared to these other community types. This distinction held for most of the big sagebrush stands even when dominant shrub species were removed from the calculation of ***Sc***, supporting observations of substantial changes in understory assemblages as well as in the shrub canopies.

The severity of the fire in these stands may have resulted in depletion of the soil seed bank as well (Young et al. 1976) affecting both non-native and native species. Along one heavily burned transect (BRMP 20, PC 1) cheatgrass frequency declined from 100% to 5.0% in 2001, and remained at a very low level (15.0%) in 2002 (By 2004, frequency had recovered to 100%). Less extreme variations in cheatgrass abundance were observed elsewhere; however, even where cheatgrass abundance is still low, the trend is towards increasing abundance of this invasive annual.

Recovery of native perennial vegetation in former big sagebrush shrublands was still far below pre-fire levels in 2004. In a few places perennial forbs such as Carey's balsamroot (*Balsamorhiza careyana*), Cusick's sunflower (*Helianthus cusickii*), and globemallow (*Sphaealcea munroa*) became abundant during the springs of 2003 and 2004. In most cases, however, increases in the abundance of perennial forbs and hemishrubs amounted to only a few percent in absolute cover and failed to compensate for large declines in perennial grasses and the complete absence of dominant shrubs.

In the absence of significant cover by perennial plant species, invasive annual forbs such as tumble mustard, along with some native annuals such as tansy mustard (*Descurainia pinnata*), and other annual forbs have colonized many of these sites, although cover remains sparse in many areas. As noted above, this suite of broadleaf annuals often dominate a a brief early successional stage following stand-clearing wildfires, prior to cheatgrass attaining dominance on a site. One or a few generations of these annuals contribute to the development of a plant litter layer that facilitates seed germination and establishment of cheatgrass (Young and Longland 1996, Evans and Young 1970). Cheatgrass cover and frequency have increased annually since 2001 in these burned-over sagebrush stands, and density also increased in 2004, adding to concerns about the trajectory of succession on these sites.

While cheatgrass is the species that appears best suited to exploit these post-fire conditions, other aliens are capable of exploiting these conditions as well (Grace et al. 2001, Bushey 1995). Diffuse knapweed (*Centaurea diffusa*) is found along roads bordering heavily impacted areas, while rush skeletonweed (*Chondrilla juncea*) and other invasive composites are capable of long-distance wind-aided dispersal from locations on ALE and elsewhere on the Hanford Site (Evans et al. 2003).

The potential for soil erosion on these heavily impacted sites has apparently been moderated to some degree by the establishment of annual vegetation and the slow recovery of perennials since 2000. Very few observations of dust storms, dust devils, sand over roadways, and other evidence of drifting soil particles were made in 2003 and 2004 compared to the two years immediately following the fire. However, the potential for long-term degradation and erosion of surface soils will remain so long as perennial vegetation does not reclaim the site. The extensive loss of perennial vegetation and microbiotic crust cover in arid lands initiates a cycle of soil degradation that results in reduced hydrological function (less infiltration of precipitation into soils, shorter retention time, and increased runoff), decreased biotic production, reduced input of

a.

b.

c.

Fig. 1.11. Wyoming big sagebrush/ Sandberg's bluegrass shrubland on silt loam soils (Biodiversity Plot 96), elevation 820 ft. (250 m): (a) pre-fire (1994); (b) 2001; (c) 2004. Cover of native perennial plants remains very low four years after the 24 Command Fire, while abundance of alien annual forbs has increased. Microbiotic soil crusts are entirely missing or, at best, present only in trace amounts.

organic matter, more extreme soil temperatures, lower biological diversity and activity, and increased erosion (Whisenant 1999, 1995, Belnap and Gillette 1998, Allen 1995, Milton et al 1994). Conversion to annual vegetation will slow but not stop this process. Substantial loss of topsoil through erosion would represent the crossing of another ecosystem threshold from which recovery can be effected only at great cost and over a long period of time (Laycock 1991, Westoby 1989). Impacts are not limited to 'donor' areas; areas receiving erosional deposits suffer burial of vegetation, seed banks, and other soil biota, in addition to alteration of other soil characteristics. Microbiotic crust is particularly intolerant of burial (Belnap et al. 2001, Belnap 1999).

Threetip sagebrush shrublands did not exhibit the same degree of impact upon the associated herbaceous understory, except within limited areas. Based on pre-fire shrub cover, fuel loads appear to have been lower in these stands than in the big sagebrush communities. Since these stands occur in more mesic habitats, at higher elevations and on northerly aspects, community resilience was likely favored in these sites compared to lower elevation shrublands. Threetip sagebrush stands may recover structural characteristics more quickly than big sagebrush stands, since *A. tripartita* is capable of resprouting from crowns following wildfire, although mortality may occur as a result of severe fires (McArthur and Taylor 2004, Akinsoji 1988). While threetip sagebrush continues to persist on these sites, canopy covers were still significantly below pre-fire levels in 2004, and Wyoming big sagebrush, historically a common associate of threetip sagebrush in these habitats, was almost entirely missing.

Winterfat is a long-lived shrub that is more common in the Great Basin than in our area (Yensen and Smith 1984). Like threetip sagebrush, winterfat is capable of resprouting following wildfire, but dense stands may experience high mortality as a result of severe fires (Pellant and Reichert 1984). Winterfat stands in our samples suffered dramatic declines in the shrub canopy apparently related to shrub mortality following the 24 Command Fire and also exhibited high mortality of perennial grasses and forbs in the understory (Fig. 1.12). Percent cover and density of cheatgrass in winterfat stands on ALE have increased substantially since 2001. Unlike threetip sagebrush, winterfat communities occur at lower elevations (below 1200'/ 365 m) on the ALE Reserve where warmer, drier conditions are less favorable for the recovery of perennial vegetation. Winterfat communities are in decline in the Great Basin where winterfat reproduction appears to be threatened by competition from invasive annual plant species (Kitchen and Jorgenson 1999). Population trajectories for this species in south central Washington are not well known, but winterfat communities occupied less than 1100 acres on the ALE Reserve prior to the 24 Command Fire (Wilderman 1994).

Observations in formerly dense stands of both winterfat and threetip sagebrush support evidence from other sources that these resprouting shrub species can suffer substantial mortality as a result of severe fires (McArthur and Taylor 2004, Akinsoji 1988, Pellant and Reichert 1984). Moreover, with the alteration of fire regimes by invasive species, even resprouting shrub and hemishrub species will be killed by the recurrence of fires at ever more frequent intervals (Simmons and Rickard 2002, Brooks and Pyke 2001, Whisenant 1990, Rickard 1989, Bunting 1985, Young 1983).

Recovery of the native perennial grasslands of the ALE Reserve varied by dominant species and plant community type as well as from plot to plot. The substantial reduction of native perennial grass cover in 2001 is consistent with other observations made during the initial year following wildfire in the shrub-steppe (Rickard et al. 1988, Young 1983, Uresk et al. 1976, Harniss and Murray 1973). The response of native bunchgrasses to wildfire varies by species, season of burning, amount and timing of precipitation during the first growing season following the fire, and other factors (Bunting 1985, Uresk et al. 1980). The 24 Command Fire struck in early summer, a time when some native species may be at their most vulnerable (Wright 1985). Large bunchgrasses such as bluebunch wheatgrass (*Agropyron spicatum*) and needle-and-thread (*Stipa comata* and *S. thurberiana*) are still actively growing at this time and are highly susceptible to injury.

The condition of bluebunch wheatgrass grasslands varied across the Reserve, with habitats above 1000 – 1200 ft. (305 -365 m) tending to be in better condition than those below this range. The herbaceous component in many mid-elevation stands appeared to be headed towards recovery in 2004 (Fig. 1.13). The best of these stands may approach the condition they were in immediately before the 24 Command Fire within the next few years. Stands below 1000 ft. were in the poorest condition in terms of cheatgrass abundance and the virtual absence of microbiotic crusts (Fig. 1.14). The prospect of recurring catastrophic wildfires and resultant stepwise increases in cheatgrass abundance prompt concerns about ecosystem integrity, despite the recovery of perennial bunchgrasses. From a long-term perspective, key elements of the shrub-steppe ecosystem, such as large shrubs and microbiotic crusts, are still missing or in poor condition in all of these habitats and will require extensive and persistent restoration efforts if these habitats are to again approach historical levels of productivity and wildlife habitat value.

Outside of shrublands, most bunchgrasses on the ALE Reserve survived the 24 Command Fire (Fig. 1.15). Overall cover of bluebunch wheatgrass (*Agropyron spicatum*), the dominant large bunchgrass over most of the middle and upper slopes of ALE, has increased substantially since 2001, but in 2004 was still significantly below pre-fire levels. Many studies have reported more rapid and vigorous recovery of bluebunch wheatgrass following wildfire than we have observed on ALE (Rickard et al. 1988, Young 1983, Uresk et al. 1976, Harniss and Murray 1973, Conrad and Poulton 1966). The coarse stems and moderate amounts of leafy material characteristic of tussock of Bluebunch wheatgrass (as well as squirreltail [*Sitanion histrix*]), tend to burn relatively quickly compared to other bunchgrasses, thus minimizing the transfer of heat to the plant's crown (Young 1983, Wright 1971). So long as individuals are not killed outright, bluebunch wheatgrass has been reported to match or exceed pre-fire production by the second or third year following fire (Britton et al. 1990, Rickard et al. 1988, Uresk 1980, Harniss and Murray 1973). Successive wildfires at short intervals will gradually reduce vigor, especially when coupled with increased competition from cheatgrass and other invasive species. Cheatgrass seedlings outcompete *Agropyron* and other native bunchgrass seedlings, strongly inhibiting the establishment of new cohorts to replace weakened or senescent individuals (Harris 1967, 1977).

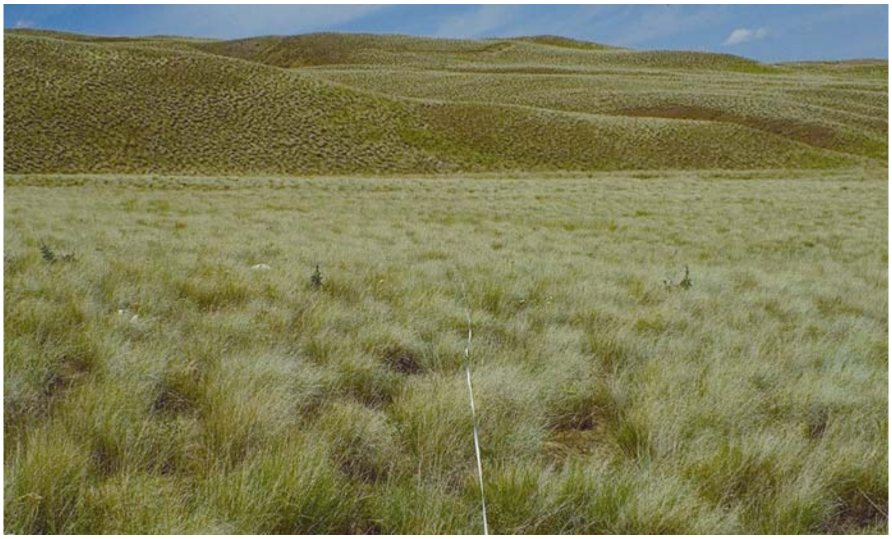
Compared to bluebunch wheatgrass, *Stipa* species (*S. comata* and *S. thurberiana*) and Idaho fescue (*Festuca idahoensis*) suffered more severe losses of aboveground cover and exhibited only limited recovery by 2004. Both Idaho fescue and *Stipa* develop more densely leafy tufts that burn longer and generate higher temperatures than bluebunch wheatgrass (Young 1983, Wright 1971; but see Robberecht and Defosse 1995). These

a. **b.**


c. **d.**


Fig. 1.12. Winterfat-Sandberg's bluegrass community (Biodiversity Plot 44), elevation 900 ft. (275 m): (a) pre-fire (1994); (b) 2001; (c) 2004.; (d) unburned remnant of mature winterfat community along Gate 111 Road. Winterfat stands exhibited large reductions in shrub canopy cover along with mortality of individual winterfat shrubs as a result of the 24 command Fire. Percent cover of cheatgrass in 2004 was nearly double pre-fire values.

a.



b.

c.

Fig. 1.13. Bluebunch wheatgrass-Sandberg's bluegrass grassland (Biodiversity Plot 79), elevation 1540 ft. (470 m): (a) pre-fire (1994); (b) 2001; (c) 2004. Many mid-elevation grasslands are in relatively good condition, with recovering native perennials and a low abundance of cheatgrass. The canopy layer of Wyoming big sagebrush, present prior to wildfires in the 1970s and 1980s, is still missing, however, and the condition of microbiotic soil crusts varies from high cover of early seral species to almost entirely lacking.

a.

b.

c.

Fig. 1.14. Bluebunch wheatgrass-Sandberg's bluegrass grassland (Biodiversity Plot 102), elevation 760 ft. (230 m): (a) pre-fire (1994); (b) 2001; (c) 2004. Grasslands at lower elevations (below 1000 ft./ 305m) exhibited increasing abundance of cheatgrass.

Fig. 1.15. Surviving bluebunch wheatgrass tussock in 2001, one year after the 24 Command Fire. Dense fuels in the interiors of large bunchgrasses may generate killing heat when burned; however, temperatures at the perimeter of the tussock remain lower, allowing meristems (growing points) there to survive.

species are highly susceptible to fire damage, especially in early summer, and may take many years to recover to pre-fire levels of production (Britton et al. 1990, Bunting 1985, Wright 1985, Uresk et al. 1980 Harniss and Murray 1973, Conrad and Poulton 1966). Frequency of *Stipa*, much reduced in 2001, did recover statistically to pre-fire values in 2002. Idaho fescue suffered both severely reduced cover and frequency following the fire, and percent cover was still only 35% of pre-fire values in 2004.

Sandberg's bluegrass (*Poa sandbergii*) is more susceptible to damage from spring burning than from summer wildfires (Young 1983). Initial reductions of Sandberg's bluegrass following the 24 Command Fire were less than those for the larger bunchgrass species in 2001, and by 2002 cover and frequency of this cool season perennial had recovered to pre-fire values. This early-maturing species had completed its seasonal growth and was largely dormant by the time the wildfire occurred (Rickard et al. 1988, Bunting 1985, Wright 1985). The small stature and diameter of Sandberg's bluegrass tussocks provide less biomass to generate killing heat than in the more robust warm season bunchgrasses (Wright and Klemmedson 1965). Where dense woody fuels contributed to fire intensity, however, a decline in all perennial understory species was observed regardless of stature and season of maturation.

Owing to increased moisture, lower evaporative demand, and fewer woody fuels, herbaceous plant communities at elevations greater than 1200 – 1400 ft. (365 – 427 m) elevation were less impacted than those at lower elevations. This effect was especially apparent above 2000 ft (610 m). Invasive species, although still present, also occupy a smaller proportion of the landscape at higher elevations. These two factors taken together suggest that higher elevation communities should display the most direct trajectory towards recovery to pre-fire conditions. However, the absence of significant stands of big sagebrush or threetip sagebrush at these elevations indicates unoccupied niches and lower ecological function in plant communities compared to historic conditions and site potential (Wilderman 1994).

Rare Plant Species. Occurrences of Piper's daisy (*Erigeron piperianus*) were scattered generally within the area of occurrence outlined by Caplow and Beck (1995, 1997). Observations suggest that, with the possible exception of former sagebrush stands where fire severity was high, the species is probably in good condition across most of its range on ALE. However, repeated fires at short intervals within the habitat area could reduce the competitive vigor of this species, along with other hemishrubs, and could threaten local populations.

Based on vegetation plot records, basalt milkvetch (*Astragalus conjunctus* var. *rickardii*) appears to be stable and recovering from a slight decline in abundance following the 24 command Fire. Basalt milkvetch's relatively mesic, high elevation habitats support plant communities that appear to be somewhat resilient following fire, and which exhibit lower levels of invasion by non-native species than shrub steppe communities at lower elevations.

No quantitative data and no recent observations are available on which to evaluate the status of Columba milkvetch (*Astragalus columbianus*) on the ALE Reserve. This species may benefit from moderate disturbance, such as low intensity fires or roadside erosion (WNHP 2000, F. Caplow pers. comm.).

While vegetation data and observations do not indicate significant changes in the abundance and distribution of these species as a result of the 24 Command Fire, the available information is insufficient to adequately assess the status of these populations, particularly Columbia milkvetch. Rare plant populations are likely to fluctuate even in the absence of disturbance (Soll et al. 1999) and ongoing monitoring of populations is necessary in order to characterize population trends accurately and to assess threats to the viability of local populations.

Microbiotic crust. Inexperience and varying search images probably led to inconsistencies in estimating microbiotic crust (MBC) cover in 2001. It seems not unlikely that the field botanists who provided pre-fire baseline data for reference plots may also have struggled with this problem. Therefore, comparisons between pre- and post-fire crust abundance must be made with caution. Difficulties for the TNC field crew in 2001 included distinguishing biological soil crusts from physical surface crusts, and applying 'forensic' techniques to record MBC that would probably not have been noted by a more general survey. Search protocols were refined and standardized in subsequent years, and estimates from 2002 through 2004 should be much more reliable and comparable to pre-fire estimates.

Despite the inconsistencies discussed above, MBC abundance was clearly reduced in all habitat types following the 24 Command fire, with the most dramatic reductions occurring in shrublands. This shrubland effect was not limited to big sagebrush stands but occurred within shrublands of all types where pre-fire shrub cover was greater than 5.0%. The continued decline in crust cover after 2002 indicated by the data is somewhat puzzling. It is conceivable that in some areas dead crust remained physically intact for a time following the fire and was mistakenly identified as living MBC. More recently decomposition and surface abrasion may have worn away these patches so that field staff no longer misrecorded them. Surviving patches of living crust may also have suffered mortality sometime after the fire as a result of fire-induced changes in the physical or chemical environment.

Changes in the abundance of MBC following wildfire are accompanied by qualitative changes as well. Microbiotic crusts can be severely degraded or killed by high temperature wildfires, with losses in biomass and species diversity accompanying reductions in cover (Johansen et al. 1993, Whisenant 1990). While reducing the species richness of crust assemblages, burning also alters species composition, favoring a few species of short mosses over tall mosses and lichens (Hilty et al. 2004). Lichens are the crust components most easily damaged by wildfire, as well as the slowest to recover (Belnap 1993, Johansen et al. 1984). The role of microbiotic crusts in shrub-steppe ecosystems is still incompletely understood; however, any loss or degradation of MBC will likely affect important ecosystem functions such as soil stabilization, soil water relations, nutrient capture and cycling and, possibly, resistance to biological invasion (Belnap et al. 2001, Belnap 1999, 1994, Belnap and Gillette 1998, Whisenant 1999, Kaltenecker et al. 1999).

The condition of crusts across the ALE Reserve is generally poor. Even areas that currently support extensive cover of MBC are characterized primarily by rudimentary crust assemblages in early stages of recovery from disturbance (McIntosh 2003 and pers. comm.). Microbiotic crusts exhibiting the dark coloration, species diversity, and complex microtopographic relief characteristic of mature crusts (Belnap et al. 2001, Johnston and Belnap 1997) are found only in a few of the unburned sagebrush refugia. While no data are available on post-fire ungulate use of these habitats, it is plausible to conjecture that elk use of these habitats (McCorquodale 1987) may concentrate in the remaining few areas, with deleterious impacts to biological crusts.

Attempts to assess crust recovery based solely on visual surface cover, as in this study, fail to account for critical properties such as crust thickness, biotic diversity, and most functional attributes, and may greatly underestimate recovery time (Belnap 1993).

Conclusions and recommendations. The 24 Command Fire has had a profound effect on the structure and function of plant communities of the ALE Reserve, and exacerbated or underscored ecosystem trends that threaten biodiversity. Although early evidence suggests the fire resulted in significant declines in shrub-steppe quality over a large area, long-term trajectories of plant community recovery are not yet clear only four years out from the fire. Effects of the fire may be relatively transitory in some areas; in others, the effects will be felt for decades to come. Region-wide trends of decline in shrub-steppe ecosystems due to altered fire regimes, invasive species, and habitat conversion and fragmentation (Knick 1999, Knick et al. 2003) add to concerns about the long-term integrity of the plant communities and their associated biota on the ALE Reserve and the Hanford Reach National Monument. Careful management will be required if the Reserve is to continue its critical role as a premier reservoir of shrub-steppe biodiversity. A long-term commitment to integrated and adaptive approaches to invasive species management, fire management and restoration practices will be required to successfully manage the ALE Reserve and other shrub-steppe ecosystems in the coming years.

The status of invasive species and the condition of native plant communities following the 24 Command Fire vary across the Reserve. Trajectories for recovery of native plant communities vary both between and within major plant community types. High elevation shrublands and grasslands (threetip sagebrush/ Idaho fescue and threetip sagebrush/ bluebunch wheatgrass potential natural community types), comprising approximately 15% of the ALE landscape (Wilderman 1994) are in the best condition overall and in general appear to be capable of recovery of most of the ecological structure and function that was present prior to the 24 Command fire within 3 – 5 additional years with minimal management intervention. Minimization of soil disturbance, continued monitoring of cheatgrass abundance, and aggressive treatment of noxious weed infestations in these areas is strongly recommended.

Wyoming big sagebrush shrublands that existed prior to the 24 Command fire continue to be in poor condition with virtually no shrubs, low cover of native perennials and microbiotic crusts, and increased abundance of alien forbs and an apparent trajectory towards increasing abundance of cheatgrass. The success of rehabilitation efforts in the former big sagebrush shrublands (Section II, this volume) will help to determine the recovery trajectory for these sites. Continued monitoring of overall condition and invasive species status of these stands is strongly recommended. Aggressive management activity to control cheatgrass and to enhance the recovery of natural structure and function of these stands will be critical to the long-term ecological integrity of these habitats

Spiny hopsage is an important associate of Wyoming big sagebrush in low elevation shrublands. While spiny hopsage may be established from seed (Shaw and Hafercamp 1994), based on its low rate of reproductive success locally this species may require active intervention to assure its continued presence in these habitats. Planting of bare root or containerized seedlings propagated from locally collected seed (or cuttings; Everett et al. 1978) is strongly recommended in future rehabilitation and restoration plans.

Winterfat communities are an important component of habitat diversity on the lower slopes of the ALE Reserve. The reproductive success of winterfat in the Great Basin has proven vulnerable to competition from invasive annuals. Not enough is known about the reproductive ecology of this species locally to determine if it is at risk, but the areas in our samples do not appear to be resilient. Continued monitoring of overall condition and invasive species status winterfat communities, along with reproductive and demographic studies of winterfat, are strongly recommended to determine whether active intervention is called for to assure the ecological integrity of this locally uncommon community type.

The conditions of bluebunch wheatgrass – Sandberg's bluegrass and needle-and-thread -- Sandberg's bluegrass grasslands are extremely variable. In general, grasslands are in good condition, with recovering perennial grasses and overall cheatgrass abundance comparable to pre-fire records. These community types occupy more than 25,000 acres (10,120 ha) or more than 30% of the ALE landscape (Wilderman 1994), and range in elevation from as low as 670 ft. (204 m) to over 1800 ft. (550 m). While condition is variable at all elevations, sites in the upper portion of this range are generally in the best condition, while most sites below 1000 ft (305 m) exhibit increasing abundance of cheatgrass. Reintroduction of Wyoming big sagebrush into selected favorable areas within these community types will be necessary to restore the structure and function of an intact shrub-steppe ecosystem. Continued monitoring of cheatgrass abundance of these stands and timely treatment of satellite occurrences of noxious weeds is highly recommended.

A truly accurate portrayal of abundance and condition of microbiotic crusts (MBC) on the ALE Reserve may require the work of specialized crust ecologists. Microbiotic crust is a critical functional element of shrub-steppe ecosystems, one that is threatened regionally by wildfire and by invasive species. The few remaining areas of undisturbed crusts on ALE and the Monument as a whole are small and isolated. These areas represent critical reference stands and refugia of biodiversity, and should be managed very carefully. Management should consider the following recommendations:

a) Areas of intact 'climax' crust should be identified and recognized in resource management and fire management plans as irreplaceable biological reservoirs of moss and lichen species (and perhaps other soil organisms) that occur nowhere else.

b) Following identification of critical conservation areas for 'climax' crust assemblages, wildlife exclosures should be constructed around selected areas as part of a program to determine the potential of threats to this resource from habitat use by ungulates.

c) In order to establish baselines against which to assess change in biological crust communities, quantitative inventories of MBC in 'climax' areas and along existing permanent plots on ALE and around the Monument are strongly recommended. Such studies can build upon work that has already been done in the area (McIntosh 2003, Link et al. 2000).

d) The Monument should encourage controlled experimental studies to assess the effects on MBC of herbicide treatments commonly used for control of cheatgrass and other invasive species, in order to determine potential threats to this resource that may be associated with these operations.

Restoration of native bunchgrasses appears to assist in the recovery of biological soil crusts (Hilty et al. 2004). No proven techniques exist at present for the outright restoration of MBC at a landscape scale (J. Belnap pers. comm.). Therefore, all management activities related to restoration, invasive species, and fire management, along with general road and facilities maintenance, should be conducted in such a way as to minimize or eliminate any adverse effect on existing high-quality microbiotic crusts (Belnap 1994, Dobler 1994).

Filling data gaps regarding land use and disturbance histories on the ALE Reserve may be difficult. However, reconstruction of a much more detailed fire history is possible and is strongly recommended. This can be accomplished by Monument personnel in cooperation with the Department of Energy and Pacific Northwest National Laboratory and in consultation with other local experts. Fire history reconstruction should be completed as soon as possible, before information recorded only in the memories of individuals is irretrievably lost.

The altered wildfire regime promoted by the proliferation of cheatgrass poses a threat to biodiversity not just on the ALE reserve but throughout the entire Lower Columbia Basin. Successive large-scale wildfires over the last three decades on the ALE Reserve continue to set back plant succession on potential sagebrush habitat and create conditions that favor invasive plant species and communities over native ones (Whisenant 1990). Fire management needs to focus on practices which, to the degree possible, restore fire regimes to their historical range of variation in terms of frequency, season, and extent of area burned. Particular emphasis should be placed on limiting the number of ignitions, enhancing capacity for rapid response to suppress ignitions and limit the spread of fires, developing and enhancing less flammable buffers near likely points of ignition (such as roadways) and reducing the abundance of highly flammable fuels, such as cheatgrass, that facilitate frequent, catastrophic burns. Maintaining fire frequencies at appropriate intervals for native stands is perhaps the most critical tool that can be applied in the defense of the Monument's outstanding native plant communities and the wildlife that depends upon them. Failure to effectively manage wildfire in the future will negate years of costly restoration efforts.

The creation or enhancement of fuel breaks in order to confine fires to limited areas can be an effective part of an integrated fire management plan (Pellant 1999). Fuel breaks may serve to protect rare species habitats or high quality stands of native vegetation, or to break annual grasslands into more manageable blocks for fire suppression efforts. Care must be taken in the planning and creation of fuel breaks. The efficacy of fuel breaks depends directly upon both width and state of maintenance (Wilson 1988). Fuel breaks may act as refuges and migratory corridors for alien plant species (Keeley 2001), may facilitate erosion (Pellant 1999), and may serve as barriers to the movement of native wildlife (Ingelfinger et al. 2001).

Many opportunities for the enhancement of fuel breaks exist along roads within the Reserve. Carefully sited, such enhancements would create little or no new disturbance. In some areas roadsides are already heavily disturbed as a result of previous fire suppression activities, and clearing and/ or restoring perennial vegetation in these corridors would accomplish weed management objectives at the same time. Opportunities of this sort exist along the 1200 Foot Rd., the Wooden Powerline (Gate 113) Rd., and elsewhere where vegetation at least one bulldozer-blade width from a road is primarily alien, sometimes owing to disturbance from previous fire-management efforts. Removal or intensive control of roadside vegetation in selected areas would result in virtually no additional direct impact on native plants and should decrease alien densities in the corridor. Roadside conditions are not uniform, however. Where intact native perennial vegetation exists along roadsides these areas should be maintained as natural firebreaks. Disturbance of such areas would be counterproductive, encouraging colonization by flammable invasive species. Involvement of biological staff at all stages of the planning and installation of fuel breaks is strongly recommended.

Fire resistant vegetation or greenstripping may be considered in the fuel break program (Pellant 1999, 1994). Greenstrips are less effective as fuel breaks than vegetation-free areas, but are less expensive to maintain, minimize erosion dangers and barriers to wildlife, and may offer some resistance to invasive species. Selection of plant material for fuel breaks must be considered carefully. Greenstrip vegetation, in addition to being adapted to arid sites, should be fire resistant through as much of the fire season as possible, competitive with invasive species, tolerant of burning (Pellant 1994), and non-invasive. Non-native species such as crested wheatgrass (*Agropyron cristatum)* and desert wheatgrass (*A. desertorum*) are frequently selected due to availability and ease of establishment and maintenance. Crested wheatgrass has already been introduced in several areas on the ALE Reserve. The introduction of non-native species into or adjacent to high quality areas must be viewed with extreme caution. Such species may be most appropriate in degraded areas adjacent to annual grasslands or as temporary, place-holding measures elsewhere. Recent researchers have suggested a transitional role for crested wheatgrass plantations in the restoration of degraded shrub-steppe lands to perennial vegetation (Cox and Anderson 2004) although this view conflicts with many assessments that suggest crested wheatgrass outcompetes native species and suppresses biodiversity (e.g., Stevens and Monsen 2004, Harrison et al. 1996, Chambers et al. 1994, Allen and Jackson 1992). Existing crested wheat plantations around the ALE Labs/ Nike Site and at the Gate 117 Rd. in the vicinity of the Benson Ranch may represent suitable sites in which to test this hypothesis. These plantations should be maintained until they can be replaced with native assemblages and any expansion of crested wheat from these sites carefully monitored. Some greenstrip species such as annual kochia (*Kochia scoparia*) have proven to be invasive; such species must be eradicated or controlled and their future use avoided.

Greenstripping with native species will protect the biological integrity of areas where installed and is highly appropriate for use in and adjacent to high quality natural areas. Native Columbia Basin species may have lower moisture content, remain green for a shorter portion of the fire season, and are not as resistant to invasive species as crested wheat. Greenstrips that function during some but not all of the fire season will still contribute towards the goal of lessening fire frequency and extent (Monsen 1994b).

Management of cheatgrass infestations will also have a strong influence on ALE fire regimes. The maintenance of native plant communities with low densities of cheatgrass, and the replacement of continuous cheatgrass mats with native bunchgrasses can inhibit the spread of surface fires. In some areas on the middle slopes below the 1200

Ft. Rd. and elsewhere *de facto* greenstrips of native species may be created by treating cheatgrass infestations and maintaining less fire-prone native plant communities. Such maintenance of selected portions of native communities will be less destructive to the landscape, may well be less expensive than installing new greenstrips, and may help to preserve biodiversity of the communities treated. Planning, treatment, and monitoring of such native strips should be done experimentally at first to determine effects on all components of the ecosystem.

Any activity associated with fuel management should be scrutinized carefully and undertaken with the minimum possible disruption of the landscape. Any clearing of fuel breaks must be undertaken with full consideration of the potential impacts on the spread of invasive species and with a commitment to maintain or restore where impacts are created.

Fire response plans should be developed that identify likely areas of ignition, sensitive resource areas, and develop policies and procedures for the construction of emergency firebreaks. Construction of such breaks should be avoided wherever possible, but especially in areas of intact native vegetation. Once constructed, it can be difficult or even impossible to reverse the damage caused to native plants and microbiotic crusts. Response plans should identify locations where, because of the presence of rare species or high quality communities, firebreak construction should be avoided, and identify alternative locations that may be acceptable.

The problem of cheatgrass must be addressed in relation to native plant community health and fire management practices. There are no simple answers; no permanent solution to the problem of cheatgrass control is currently available (Youtie et al. 1998), and management is extremely challenging. Methods of physical and chemical control of cheatgrass have had variable success in facilitating the maintenance or establishment of native plant communities in infested areas (Carpenter and Murray 1999, Allen 1995, Matisse and Scholten 1994, Monsen 1994a). Efforts to limit cheatgrass infestations or the conditions that favor them in existing high quality plant communities are much more likely to succeed than efforts at control and restoration once an infestation has become serious (Belnap and Phillips 2001).

Assessment of post-fire patterns of cheatgrass abundance as outlined in this study can help identify top priority areas where restoration will be most effective and is most urgently needed. While no habitats on the ALE Reserve appear to be immune to cheatgrass invasion, environmental and plant community characteristics may confer a degree of resistance upon some types of habitats. Our findings suggest that, following wildfires in big sagebrush stands, a brief window of opportunity may be available for restoring native perennial plant species with minimal competition from cheatgrass (Stevens and Monsen 2004, Evans and Young 1978). Restoration efforts that take advantage of this window can minimize site preparation expenses dedicated for invasive species control, while at the same time minimizing risks of herbicide effects on desirable remnant native species. Following the 24 Command Fire, this opportunity appeared to last only for a single season on finer-grained silt-loam soils. Where sagebrush stands were cleared on sandy soils, however, restoration opportunities appeared to exist even four years following wildfire.

Monument managers must have timely information regarding habitat condition and the status of invasive species populations in order to adaptively manage at-risk habitats. Additional management intervention is likely to be required in former sagebrush stands between the Gate 117 and Gate 118 roads, and in winterfat stands on the lower slopes above the Cold Creek Valley. In these areas continued annual monitoring of a portion of the network of permanent plots used in this study is strongly recommended. Outside of these heavily impacted areas, monitoring of a selected array of existing permanent plots should become a periodic undertaking at no greater than 5-year intervals. Continued monitoring of cheatgrass abundance and the status of native perennial plant communities is necessary to provide timely indication of population trends and to allow managers to evaluate levels of threat to habitat resources around the Reserve and to respond appropriately.

The analyses contained within this report would not have been possible without the existence of a network of permanent vegetation plots on the ALE Reserve and the availability of relevant pre-fire data from those plots. Permanent plots are invaluable in detecting and documenting change in wildland vegetation and are sources of critical information for the managers of dynamic landscapes.

The Hanford Reach National Monument is fortunate to have three sets of reference vegetation plots available on the ALE Reserve. Each type of plot used in this study has its particular merits, as well as some drawbacks. The transect-and-microplot methodologies of the BRMaP and Steppe-in-Time plots provide much more precise ocular estimates of plant cover, particularly regarding annual species. Steppe-in-Time Plots provide the longest, albeit intermittent, record of any of the plot types, having been read from 1992-1993, 1997, and 2000 – 2004. Biodiversity Plots provided coverage of winterfat communities, needle-and-thread grasslands, and of forb-rich communities along the crest of Rattlesnake Mountain that was not available from the other methodologies, and enhanced the coverage of high quality bluebunch wheatgrass grasslands at all elevations.

The network of BRMaP Plots extends across the entire Hanford Site. Nine BRMaP plots are located on Monument Lands outside the boundaries of the ALE Reserve (Fig. 1.16, Table 1.36). Five macroplots (Each BRMaP macroplot includes three to five 100 m vegetation transects) are located on the North Slope, on lands currently managed by the U.S. Fish and Wildlife Service (USFWS), and two each are placed within the McGee-Riverlands Unit and the Hanford Dunes, areas managed by the U.S. Department of Energy (DOE). These plots have not been resampled since 1996, the year they were installed. Periodic resampling of these plots is necessary in order to evaluate trends in habitat condition and invasive species populations, as well as to update baseline information in advance of potential wildfires or other disturbances, or climatically induced vegetation change. It is strongly recommended that the USFWS and DOE, in concert as co-managers of the Hanford Reach National Monument, make certain that these plots are located and resampled within the next one to two years, and that resampling of these plots thereafter become part of a periodic effort.

While the existing BRMaP plots will provide valuable information, there is a need for better coverage on a landscape the size of the Hanford Reach National Monument. It is recommended that Monument biologists install additional plots in selected areas around the Monument (exclusive of the ALE Reserve, where coverage is generally very good). It is recommended sampling methodologies follow the BRMaP model. However, transects should be located individually in selected habitats, rather than in clusters of three to five. Winterfat habitats on the McGee-Riverlands Unit and big sagebrush/ bluebunch wheatgrass habitats on the Wahluke Unit are examples of habitats that should be considered for inclusion in this expanded network of monitoring plots. Recent vegetation mapping efforts on the McGee Ranch – Riverlands Unit and on the North Slope (Salstrom and Easterly 2003, 2004) can be helpful in selecting appropriate community types and sample placement sites for additional monitoring units.

Effective monitoring programs are essential to the adaptive management of natural resources. Effective programs need not be prohibitively expensive. Early detection of threats to conservation targets will contribute to considerable budgetary efficiencies when mitigation efforts are undertaken in a timely fashion (Milton et al. 1994, Moody and Mack 1988). Sampling schedules can be staggered so that only a fraction of the plot network is sampled during each year of, say, a five year monitoring cycle. While professional expertise is invaluable, volunteers can make strong contributions to vegetation monitoring efforts and offer an opportunity to further reduce the costs of a monitoring program. A pool of capable volunteers, many of whom have very strong botanical skills, is already available in the Columbia Basin and has made strong contributions to the data collection efforts recounted in this volume. The Steppe-in-Time project was an entirely volunteer effort after 1992, and numerous skilled volunteers contributed hundreds of hours towards The Nature Conservancy's efforts on the ALE Reserve from 2001-2004. In addition to their hard work, volunteers provide inspiration and a link to the community, and add to the constituency for conservation on the Monument. Incorporating proven volunteers in vegetation monitoring programs, and recruiting more volunteers from groups such as the Columbia Basin chapters of the Washington Native Plant Society and Audubon Society is strongly recommended.

The foregoing recommendations are offered with a sober understanding of the considerable challenges managers of the Hanford Reach National Monument face. Implementation of these and other sound principles across a complex and dynamic landscape, utilizing limited resources, will continue to present a daunting challenge to fire and resource management professionals throughout the Columbia Basin, and across the arid west.

Table 1.36. BRMaP macroplots on the Hanford Reach National Monument. Vegetation types are taken from Easterly and Salstrom (2003) and Soll et al. (1999).

Plot no.	**Management Unit**	**Vegetation**
11	River Corridor	bitterbrush/ Indian ricegrass dune complex
14	Wahluke	bitterbrush/ Indian ricegrass dune complex
16	Wahluke	Wyoming big sagebrush/ needle-and-thread
17	Wahluke	Wyoming big sagebrush/ Sandberg's bluegrass
18	McGee Ranch-Riverlands	Wyoming big sagebrush/ Sandberg's bluegrass-cheatgrass
21	Saddle Mt.	Wyoming big sagebrush/ cheatgrass
27	River Corridor	bitterbrush/ Indian ricegrass dune complex
29	McGee Ranch-Riverlands	Wyoming big sagebrush-stiff sagebrush/ bluebunch wheatgrass
30	Wahluke	spiny hopsage/ Sandberg's bluegrass

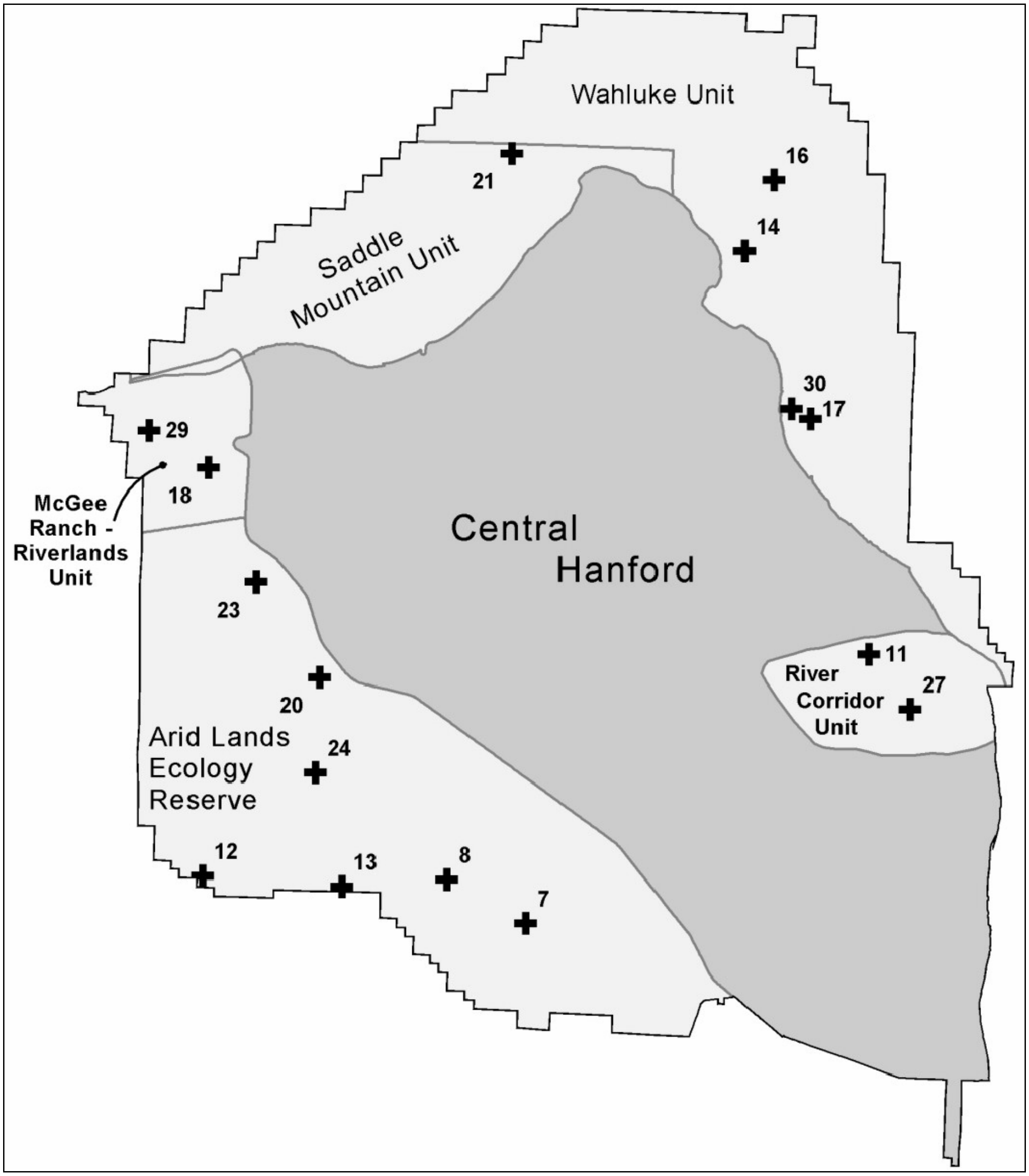


Fig. 1.16. BRMaP network of permanent plots on the Hanford Reach National Monument. Locations shown are macroplot centers. Management unit boundaries are approximate. Each macroplot includes three to five 100 m vegetation transects. Plot installation and vegetation surveys were performed in 1996. With the exception of plots on the ALE, Reserve, plots have not been revisited since that year. Plot coordinates are presented in Appendix A.

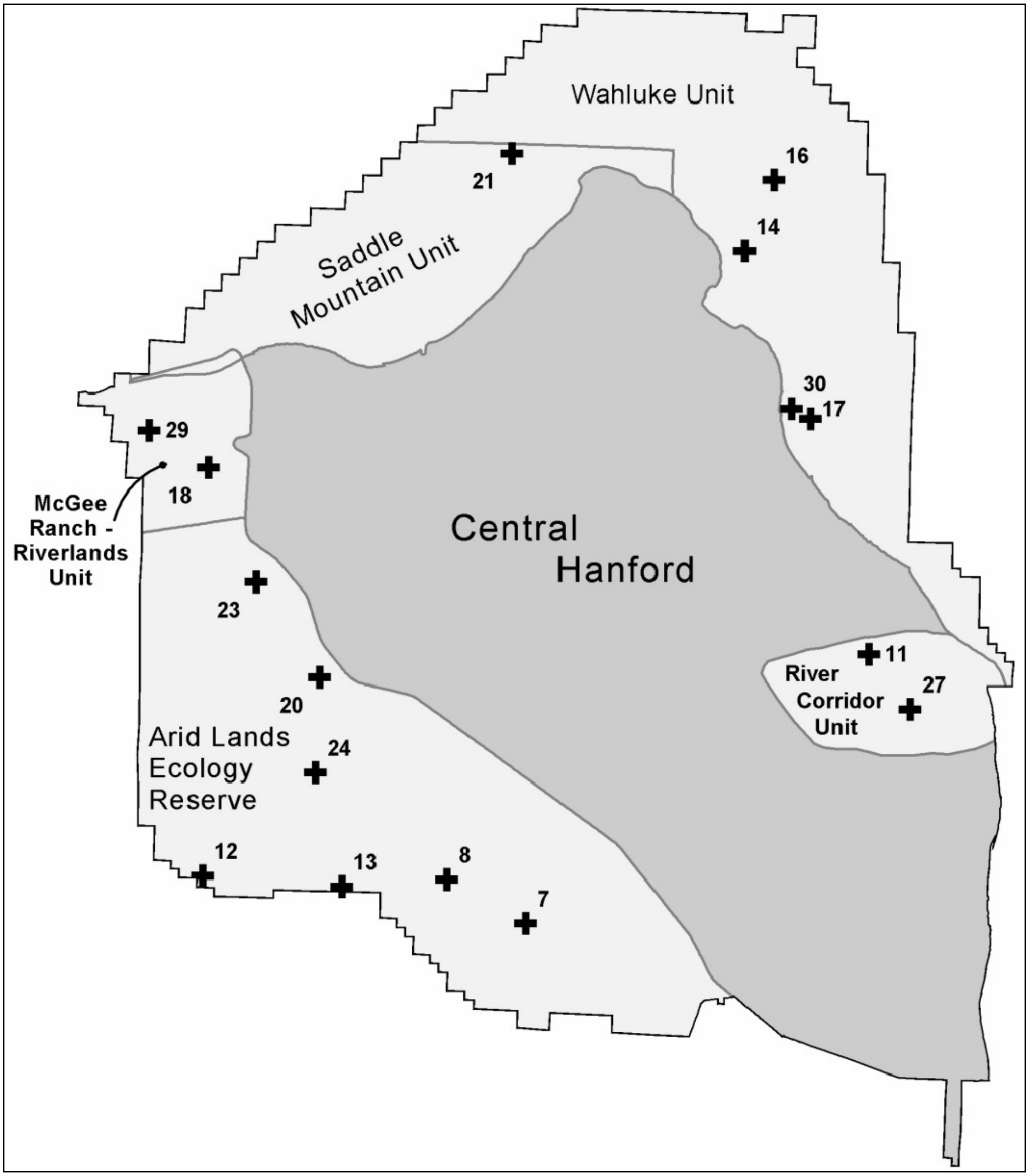


Fig. 1.16. BRMaP network of permanent plots on the Hanford Reach National Monument. Locations shown are macroplot centers. Management unit boundaries are approximate. Each macroplot includes three to five 100 m vegetation transects. Plot installation and vegetation surveys were performed in 1996. With the exception of plots on the ALE, Reserve, plots have not been revisited since that year. Plot coordinates are presented in Appendix A.

LITERATURE CITED

Allen, E.B. 1995. Restoration ecology: Limits and possibilities in arid and semiarid lands. In B.A. Roundy, E.D. McArthur, J.S. Haley, and D.K. Mann (compilers), Proceedings: Wildland Shrub and Arid Land Restoration Symposium. General Technical Report INT-GTR-315. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 7-15.

Allen, E. B., and L.L Jackson. 1992. The Arid West. Restoration and Management Notes 10: 56-59.

Akinsoji, A. 1988. Post-fire vegetation dynamics in a sagebrush steppe in southeastern Idaho. Vegetatio 78: 151-155.

Anderson, J.E., and R.S. Inouye. 2001. Landscape-scale changes in plant species abundance and biodiversity of a sagebrush steppe over 45 years. Ecological Monographs 71: 531-556.

BAER (Northern States Burned Area Emergency Rehabilitation) Team. 2000. 24 Command Fire Burned Area Emergency Rehabilitation (BAER) Plan. United States Department of Interior. 76 pp. plus appendices.

Belnap, J. 1999. Structure and function of biological soil crusts. In P.G. Entwhistle, A.M. DeBolt, J.H. Kaltenecker, and J. Steenhof (compilers), Proceedings: Sagebrush Steppe Ecosystems Symposium. Bureau of Land Management Publication No. BLM/ID/PT-001001+1150. Boise, ID. Pp. 55-64. Available online at: http://www.id.blm.gov/publications/data/SSSymp.pdf

Belnap, J. 1994. Potential role of cryptobiotic soil crusts in semiarid rangelands. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service, Intermountain Research Station, Ogden, UT. Pp. 179-185.

Belnap, J. 1993. Recovery rates of cryptobiotic crusts: Inoculant use and assessment methods. Great Basin Naturalist 53: 89-95.

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.

Belnap, J., J.H. Kaltenecker, R. Rosentreter, J. Williams, S. Leonard, and D. Eldridge. 2001. Biological Soil Crusts: Ecology and Management. Technical Reference 1730-2. US Dept. of Interior, Bureau of Land Management, Denver, CO. 110 pp.

Belnap, J., and D.A. Gillette. 1998. Vulnerability of desert biological soil crusts to wind erosion: the influences of crust development, soil texture, and disturbance. Journal of Arid Environments 39: 133-142.

Belsky, A.J., and J. L. Gelbard. 2000. Livestock Grazing and Weed Invasions in the Arid West. Oregon Natural Desert Association, Portland, OR. 31 pp.

Blackshaw, R.E., F.J. Larney, C.W. Lindwall, P.R. Watson, and D. A. Derksen. 2001. Tillage intensity and crop rotation affect weed community dynamics in a winter wheat cropping system. Canadian Journal of Plant Science 81: 805-813.

Booth, D.T. 2002. Seed longevity and seeding strategies affect sagebrush revegetation. Journal of Range Management 55: 188-193.

Bourgeron, P.S., R.L. DeVelie, L.D. Engelking, G. Jones, and E. Muldavin. 1992. Western Heritage Task Force Site and Community Survey Manual. Version 92B. Western Heritage Task Force, Boulder, CO. 24 pp.

Britton, C.M, G.R. McPherson, and F.A. Sneva. 1990. Effects of burning and clipping on five bunchgrasses in Eastern Oregon. Great Basin Naturalist 50: 115-120.

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 Miscellaneous Publication No. 11: 1-14.

Bunting, S.C. 1985. Fire in sagebrush-grass ecosystems: Successional changes. In K. Sanders and J. Durham (eds.), Rangeland Fire Effects: Proceedings of a Symposium Held 27-29 November, 1984. U.S. Bureau of Land Management, Boise, ID. Pp. 7-11.

Bushey, C. 1995. Fire effects on noxious weeds within the Columbia River Basin. Science report, Interior Columbia Basin Ecosystem Management Project. Available online at: http://www.icbemp.gov (under "Science Contract Reports, September 7, 1999").

Caplow, F., and K. Beck. 1995. A Rare Plant Survey of the Hanford Nuclear Reservation. Report to the Nature Conservancy of Washington, Seattle, WA. 65 pp. plus appendices.

Caplow, F., and K. Beck. 1997. A Rare Plant Survey of the Hanford Nuclear Reservation. Report to the Nature Conservancy of Washington, Seattle, WA. 38 pp. plus appendices.

Carpenter, A.T., and T.A. Murray. 1999. Element Stewardship Abstract for *Bromus Tectorum* L. (*Anisantha tectorum* (L.) Nevski). The Nature Conservancy Wildland Invasive Species Program. Available online at: http://tncweeds.ucdavis.edu

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

Cline, J.F., D.W. Uresk, and W.H. Rickard. 1977. Comparison of soil water used by a sagebrush-bunchgrass and a cheatgrass community. Journal of Range Management 30: 199-201.

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

Conrad, C.E., and C.E. Poulton. 1966. Effect of a wildfire on Idaho fescue and bluebunch wheatgrass. Journal of Range Management 19: 138-141.

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

Daubenmire, R. 1975. Plant succession on abandoned fields, and fire influences, in a steppe area in southeastern Washington. Northwest Science 49: 36-48.

Daubenmire, R. 1970. Steppe vegetation of Washington. Washington Agricultural Experiment Station. Technical Bulletin 62. Pullman, WA.

del Moral, R., J. H. Titus, & A. M. Cook. 1995. Early primary succession on Mount St. Helens, USA. Journal of Vegetation Science 6: 107-120.

DiTomasi, J.M., 2000. Invasive weeds in rangelands: Species, impacts, and management. Weed Science 48: 255-265.

Dobler, F.C. 1994. Washington State shrub-steppe ecosystem studies with emphasis on the relationship between nongame birds and shrub and grass cover densities. In S.B. Monsen and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp.149-161.

DOE-RL (U.S. Department of Energy, Richland Operations Office). 2001. Hanford Site Biological Resources Management Plan. DOE/RL 96-32. U.S. Department of Energy, Richland, WA.

Downs, J.L., W.H. Rickard, C.A. Brandt, L.L. Cadwell, C.E. Cushing, D.R. Geist, R.M. Mazaika, D.A. Neitzel, L.E. Rogers, M.R. Sackschewsky, and J.J. Nugent. 1993. Habitat types on the Hanford Site: Wildlife and plant species of concern. PNL-8942. Pacific Northwest Laboratory, Richland, WA.

Durham, R.E., B.A. Zamora, M.R. Sackschewsky, and J.C. Ritter 2001. Predicting first-year bare-root seedling establishment with soil and community dominance factors. In E.D. McArthur and D.J. Fairbanks (compilers), Shrubland Ecosystem Genetics and Biodiversity: Proceedings. Proceedings RMRS-P-21. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT. pp 204-209.

Easterly, R., and D. Salstrom. 2004. Current Vegetation Map of Saddle Mountain, Wahluke and Ringold Units, Hanford Reach National Monument. Report to the U.S. Fish and Wildlife Service, Hanford Reach National Monument, Richland, WA. 31 pp plus appendices.

Easterly, R., and D. Salstrom. 2003. Vegetation of the McGee Ranch – Riverlands Unit. In J.R. Evans, M.P. Lih, and P.W. Dunwiddie (eds.), Biodiversity Studies of the Hanford Site. The Natuer Conservancy, Seattle, WA. pp. 11-21.

Elzinga, C.L., D.W. Salzer, and J.W. Willoughby. n.d. Measuring and Monitoring Plant Populations. Technical Reference 1730-1. U.S. Bureau of Land Management, Denver, CO. 477 pp.

Evans, J.R., J.J. Nugent, and J. Meisel. 2003. Invasive Plant Species Inventory and Management Plan for the Hanford Reach National Monument. The Nature Conservancy, Seattle, WA. 192 pp.

Evans, R.A., and J.A. Young. 1978. Effectiveness of rehabilitation practices following wildfire in a degraded big sagebrush-downy brome community. Journal of Range Management 31: 184-188.

Evans, R.A., and J.A. Young. 1970. Plant litter and establishment of alien annual weed species in rangeland communities. Weed Science 18: 697-703.

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.

Everett, R.L., R.E. Meeuwig, and J.H. Robertson. 1978. Propagation of Nevada shrubs by stem cuttings. Journal of Range Management 31: 426-429.

Fontaine, A.L., P.L. Kennedy, and D.H. Johnson. 2004. Effects of distance from cattle water developments on grassland birds. Journal of Range Management 57: 238-242.

Franklin, J.F., and C.T. Dyrness. 1973. Natural Vegetation of Oregon and Washington. Oregon State University Press, Corvallis, OR.   417 pp.

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

Grace, J.B., M.D. Smith, S.L.Grace, S.L. Collins, and T.J. Stohlgren. 2001. Interactions between fire and invasive plants in temperate grasslands 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 Miscellaneous Publication No. 11: 40-65.

Groves, C.R., and K. Steenhof. 1988. Responses of small mammals and vegetation to wildfire in shadscale communities of southwestern Idaho. Northwest Science 62: 205-210.

Hajek, B.F. 1966. Soil Survey: Hanford Project in Benton County Washington. BNWL – 243. Pacific Northwest Laboratory, Richland, WA. 16 pp.

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

Harris, G.A. 1967. Some competitive relationships between *Agropyron spicatum* and *Bromus tectorum*. Ecological Monographs 37: 89-111.

Harrison, R.D., N.J. Chatterton, R.J. Page, M. Curto, K.H. Asay, K.B. Jensen, and W.H. Horton. 1996. Competition, Biodiversity, Invasion, and Wildlife Usage of Selected Introduced Grasses in the Columbia and Great Basins. Research Report 155. Utah Agricultural Experiment Station, Utah State University, Logan UT. 86 pp.

Hassan, M.A., and N.E. West. 1986. Dynamics of soil seed pools in burned and unburned sagebrush semi-deserts. Ecology 67: 269-272.

Harniss, R.O., and R.B. Murray. 1973. Thirty years of vegetal change following burning of sagebrush-grass range. Journal of Range Management 26: 322-325.

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

Hill, M.O., and H.G. Gauch, Jr. 1980. Detrended correspondence analysis: An improved ordination technique. Vegetatio 42: 47-58.

Hilty, J.H., D.J. Eldridge, R. Rosentreter, M.C. Wicklow-Howard. and M. Pellant. 2004. Recovery of biological soil crusts following wildfire in Idaho. Journal of Range Management 57: 89-96.

Hinds, N.R., and L.E. Rogers. 1991. Ecological Perspectives of Land Use History: The Arid Lands Ecology (ALE) Reserve. PNL 7750. Pacific Northwest Laboratory, Richland, WA. 18 pp.

Hinds, W.T., and J.M. Thorp. 1969. A climatological survey of temperature-precipitation-vegetation relations on the ALE Reserve. In D.W. Pearce (ed.), Pacific Northwest Laboratory Annual Report for 1968 to the USAEC Division of Biology and Medicine. BNWL – 1050, Volume 1: Life Sciences; Part 2: Biological Sciences. Pacific Northwest Laboratory, Richland, WA. Pp. 4.4 – 4.16.

Hitchcock, C. L., and A. Cronquist. 1973. Flora of the Pacific Northwest. University of Washington Press, Seattle, WA. 730 pp.

HMS (Hanford Meteorological Services). 2004. Pacific Northwest National Laboratory, Richland, WA. Available online at http://hms.pnl.gov

HRNM (Hanford Reach National Monument). 2004. Data from geographic information system database. Hanford Reach National Monument/ Saddle Mountain National Wildlife Refuge, Richland, WA.

Humphrey, L.D., and E.W. Schupp. 2001. Seed banks of *Bromus tectorum*-dominated communities in the Great Basin. Western North American Naturalist 61: 85-92.

Ingelfinger, F.M., and S.H. Anderson. 2001. The effects of infrequently traveled roads on sagebrush obligate passerines. Abstracts of The Wildlife Society 8th Annual Conference, Reno/ Tahoe, NV.

Interagency Fire Team. 2000. 24 Command Fire: A National Level Review. U.S. Fish and Wildlife Service Region 1. Portland, OR. 24 pp. plus appendices.

Johansen, J.R., J. Ashley, and W.R. Rayburn. 1993. Effects of rangefire on soil algal crusts in semiarid shrub-steppe of the lower Columbia Basin and their subsequent recovery. Great Basin Naturalist 53: 73-88.

Johansen, J.R.,, L.L. St. Clair, B.L. Webb, and G.T. Nebeker. 1984. Recovery patterns of cryptogamic soil crusts in desert rangelands following fire disturbance. The Bryologist 87: 238-243.

Johnston, R.L., and J. Belnap. 1997. Potential for influencing native bunchgrass restoration by manipulating soil biota. In T.N. Kaye, A. Liston, R.M. Love, D.L. Luoma, R.J. Meinke, and M.V. Wilson (eds.), Conservation and Management of Native Plants and Fungi. Native Plant Society of Oregon, Corvallis. Pp. 99-104.

Kaltenecker, J.H., M.C. Wicklow-Howard, and K. Larsen. 1999. Biological soil crusts: Natural barriers to *Bromus tectorum* establishment in the Northern Great Basin, USA. In P.G. Entwhistle, A.M. DeBolt, J.H. Kaltenecker, and J. Steenhof (compilers), Proceedings: Sagebrush Steppe Ecosystems Symposium. Bureau of Land Management Publication No. BLM/ID/PT-001001+1150. Boise, ID. Pp. 115. Available online at: http://www.id.blm.gov/publications/data/SSSymp.pdf

Kartesz, J.T.. 1999. A Synonymized Checklist and Atlas with Biological Attributes for the Vascular Flora of the United States, Canada, and Greenland. First edition. In J.T. Kartesz and C.A. Meacham, Synthesis of the North American Flora. CD-ROM version 1.0. North Carolina Botanical Garden, Chapel Hill, NC.

Keeley, J.E. 2001. Fire and Invasive Species in Mediterranean-climate ecosystems of California. 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 Miscellaneous Publication No. 11: 81-94.

Kent, M., and P. Coker. 1992. Vegetation Description and Analysis. John Wiley and Sons, NY. 363 pp.

Kitchen, S.G, and G.L. Jorgensen. 1999. Annualization of rodent burrow clusters and winterfat decline in a salt-desert community. In E.D. McArthur, W.K. Ostler, and C.L. Wambolt, Proceedings: Shrubland Ecotones. RMRS-P-11. U.S. Forest Service, Rocky Mountain Research Station, Ogden, UT. pp 175-180.

Knick, S.T. , D.S. Dobkin, J.T. Rotenberry, Michael A. Schroeder, W. Mathew Vander Hagen, and C. van Riper. 2003. Teetering on the edge or too late? Conservation research issues for avifauna of sagebrush habitats. The Condor 105: 611-634.

Knick, S., J.T. Rotenberry, and B. Van Horne. 1999. Effects of distrubance on shrub steppe habitats and raptor prey in the Snake River Birds of Prey National Conservation Area, Idaho. In P.G. Entwhistle, A.M. DeBolt, J.H. Kaltenecker, and J. Steenhof (compilers), Proceedings: Sagebrush Steppe Ecosystems Symposium. Bureau of Land Management Publication No. BLM/ID/PT-001001+1150. Boise, ID. Pp. 98-99.

Knick, S.T. 1999. Requiem for a sagebrush ecosystem? Northwest Science 73: 53-57.

Laycock, W.A. 1991. Stable states and thresholds of range condition on North American rangelands: A viewpoint. Journal of Range Management 44: 427-433.

Link, S.O., B.D. Ryan, J.L. Downs, L.L.Cadwell, J. Soll, M.A. Hawke, and J. Ponzetti. 2000. Lichens and mosses on shrub-steppe soils in southeastern Washington. Northwest Science 74: 50-56.

Link, S.O., G.W. Gee, M.E. Theide, and P.A. Beedlow. 1990. Response of a shrub-steppe ecosystem to fire: Soil water and vegetational change. Arid Soil Research and Rehabilitation 4: 163-172.

Longland, W.S., and S.L. Bateman. 2002. The ecological value of shrub islands on disturbed sagebrush rangelands. Journal of Range Management 55: 571-575.

Mack, R.N. 1981. Invasion of *Bromus tectorum* L. into western North America: An ecological chronicle. Agro-Ecosystems 7: 145-165.

Mack, R.N., and D.A. Pyke. 1984. The demography of *Bromus tectorum*: The role of microclimate, grazing and disease. Journal of Ecology. 72: 731-748.

Marsh, Michael. 1999. Landowner Report: Fitzner-Eberhardt Arid Lands Ecology Reserve. Benton County Shrub-Steppe Study. Unpublished report.

McArthur, E.D., and J.R. Taylor. 2004. *Artemisia tripartita* Rydb. In J.K. Francis (ed.), Wildland Shrubs of the United States and its Territories: Thamnic Descriptions. Vol. I. General Technical Report IITF-GTR-26. U.S. Forest Service, International Institute of Tropical Forestry, San Juan PR. Pp. 88-90.

McCorquodale, S.M. 1987. Fall-winter habitat use by elk in the shrub-steppe of Washington. Northwest Science 61: 171-173.

McCune, B., and J.B. Grace. 2002. Analysis of Ecological Communities. MjM Software design, Glendenen Beach, OR. 300 pp.

McCune, B., and D. Keon. 2002. Equations for potential annual direct incident radiation and heat load. Journal of Vegetation Science 13: 603-606.

McIntosh, T. 2003. Biological Soil Crusts of the Hanford Reach National Monument. Pp. 23-42 in J.R. Evans, M.P. Lih, and P.W. Dunwiddie, Biodiversity Studies of the Hanford Site: Final Report 2002-2003. The Nature Conservancy, Seattle, WA.

Matisse, S. N., and G. Scholten. 1994. Mechanical control of undesirable annuals on the Boise Front, Idaho. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 190-193.

Meyer, S.E. 1994. Germination and establishment ecology of big sagebrush: Implications for community restoration. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 244-251.

Miller, M., J. Belnap. S. Bealty, and B. Webb. 2001. Components of spatial and temporal soil variation at Canyonlands National Park: Implications for P dynamics and cheatgrass (*Bromus tectorum*) performance. In E.D. McArthur and D.J. Fairbanks (compilers), Shrubland Ecosystem Genetics and Biodiversity: Proceedings. Proceedings RMRS-P-21. U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT. pp 154-162..

Milton, S.J., W. Richard, J. Dean, M.A. Du Plessis, and W.R. Siegfried. 1994. A conceptual model of arid rangeland degradation. Bioscience 44: 70-79.

Monsen, S.B. 1994a. The competitive influences of cheatgrass (*Bromus tectorum*) on site restoration. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service, Intermountain Research Station, Ogden, UT. Pp. 43-50.

Monsen, S.B. 1994b. Selection of plants for fire suppression on semiarid sites. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service, Intermountain Research Station, Ogden, UT. Pp. 363-373.

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.

Nelle, P.J., K.P.Reese, and J.W. Connelly. 2000. Long-term effects of fire on sage grouse habitat. Journal of Range Management 53: 586-591

Newsome, H., and W. LaFramboise. 2004. Effects of large-scale wildfire on grassland breeding birds on the Arid Lands Ecology Reserve, Hanford Reach National Monument. Presentation to the Northwest Scientific Association 77th Annual Meeting, March 26, 2004. Ellensberg, WA.

NWS (National Weather Service). 2004. Interactive Climate Information Tool, National Weather Service Forecast Office, Pendleton, OR. Available online at: http://www.wrh.noaa.gov/pdt/climate/interactiveClimateCounty.php?county=Benton

O'Connor, G., and W. Rickard. 2003. A History of the Fitzner-Eberhardt Arid Lands Ecology Reserve: Four Decades of Environmental Research. PNNL-SA-39326. Pacific Northwest National Laboratory, Richland, WA. 200 pp.

Palmer, M.P. 1993. Putting things in even better order: The advantages of Canonical correspondence analysis. Ecology 74: 2215-2230.

Pellant, M. 1999. Fuels management in the Snake River Birds of Prey National Conservation Area. In P.G. Entwhistle, A.M. DeBolt, J.H. Kaltenecker, and J. Steenhof (compilers), Proceedings: Sagebrush Steppe Ecosystems Symposium. Bureau of Land Management Publication No. BLM/ID/PT-001001+1150. Boise, ID. Pp. 103-107. Available online at: http://www.id.blm.gov/publications/data/SSSymp.pdf

Pellant, M. 1996. Cheatgrass: The Invader that Won the West. Science report, Interior Columbia Basin Ecosystem Management Project. Available online at: http://www.icbemp.gov (under "Science Contract Reports, September 7, 1999").

Pellant, M. 1994. History and applications of the Intermountain Greenstripping Program. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 63-68.

Pellant, M. 1990. The cheatgrass-wildfire cycle – are there any solutions? In E.D. MaArthur, E.M. Romney, S.D. Smith, and P.T. Tueller (eds.), Proceedings – Symposium on Cheatgrass Invasion, Shrub Die-off, and Other Aspects of Shrub Biology and Management. General Technical Report INT-GTR-276. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 7-11.

Pellant, M., and L. Reichert. 1984. Management and rehabilitation of a burned winterfat community in southwestern Idaho. In A.R. Tiedman, E.D. McArthur, H.C. Stutz, R. Stevens, and K.L. Johnson (compilers), Proceedings: Symposium on the Biology of Atriplex and Related Chenopods. General Technical Report INT-GTR-172. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 281-285.

Peters, E.F., and S.C. Bunting. 1994. Fire conditions pre- and postoccurrence of annual grasses on the Snake River Plain. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service, Intermountain Research Station, Ogden, UT. pp. 31-36.

Pierson, E.A., and R.N. Mack. 1990. The population biology of *Bromus tectorum* in forests: Distinguishing the opportunity for dispersal from environmental restriction. Oecologia 84: 519-525.

PNNL (Pacific Northwest National Laboratory). 1997. Hanford Site 1996 Environmental Report. PNNL-11472. Pacific Northwest National Laboratory and U.S. Department of Energy, Richland, WA.

Rasmussen, J.J. 1971. Soil Survey of Benton County Area, Washington. US Department of Agriculture, Soil Conservation Service, Washington, D.C.

Rickard, W.H. 1989. Response of round-headed buckwheat to summer burning. Northwest Science 63: 144-145.

Rickard, W.H. 1975. Vegetation of knob and kettle topography in south-central Washington. Northwest Science 49: 147-152.

Rickard, W.H., and L.D. Poole. 1989. Terrestrial wildlife of the Hanford Site: Past and future. Northwest Science 63: 183-193.

Rickard, W.H., L.E. Rogers, B.E. Vaughan, and S.F. Liebetrau. 1988. Shrub-Steppe: Balance and Change in a Semi-Arid Terrestrial Ecosystem. Elsevier, NY. 272 pp.

Rickard, W.H., and M.C. McShane. 1984. Demise of spiny hopsage following summer wildfire: An authentic record. Northwest Science 58: 282-285.

Rickard, W.H., and R.H. Sauer. 1982. Self-revegetation of disturbed ground in the deserts of Nevada and Washington. Northwest Science 56: 41-47.

Rickard, W.H., J.F. Cline, and R.O. Gilbert. 1973. Behavior of winter annuals as influenced by microtopography and elevation. Northwest Science 47: 44-49.

Robberecht, R., and G.E. Defosse. 1995. The relative sensitivity of two bunchgrass species to fire. International Journal of Wildland Fire 5: 127-134.

Rosentreter, R. 1994. Displacement of rare plants by exotic grasses. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service, Intermountain Research Station, Ogden, UT. pp. 170-175.

Rotenberry, J.T., W.T. Hinds, and J.M. Thorp. 1976. Microclimatic patterns on the Arid Lands Ecology Reserve. Northwest Science 50: 122-130.

Sackschewsky, M.R., and J.L. Downs. 2001. Vascular Plants of the Hanford Site. PNNL-13688. Pacific Northwest National Laboratory, Richland, WA.
Available online at http://www. pnl.gov/ecology/Library/PNNL13688.pdf

Sauer, R.H., and W.H. Rickard. 1979. Vegetation of steep slopes in the shrub-steppe region of Southcentral Washington. Northwest Science 63: 5-11.

Schuller, C.A., W.H. Rickard, and G.A. Sargent. 1993. Conservation of habitats for shrubsteppe birds. Environmental Conservation 20: 57-64.

Simanton, J.R., G.D. Wingate, and M.A. Weltz. 1990. Runoff and sediment from a burned sagebrush community. In J.S. Krammes, Effects of Fire Management of Southwestern Natural Resources. US Forest Service, Rocky Mountain Research Station, Ogden, UT. General Technical Report RM-GTR-191. Fort Collins, CO. Pp 180-185.

Simmons, S., and W.H. Rickard. 2003. Fire effects on spiny hopsage in south central Washington. Western North American Naturalist 63: 524-528.

Simmons, S., and W.H. Rickard. 2002. Demise of an isolated buckwheat stand by repetitive wildfires. Northwest Science 76: 183-184.

Shaw, N.L., and M.R. Hafercamp. 1994. Spiny hopsage seed germination and seedling establishment. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 252-256.

Soll, J., J.A. Hall., R. Pabst, and C. Soper. 1999. Biodiversity Inventory and Analysis of the Hanford Site. Final Report: 1994-1999. The Nature Conservancy of Washington, Seattle, WA. 179 pp.

Smith, J.L., J.J. Halvorson, and H Bolton Jr. 2002. Soil properties and microbial activity across a 500 m elevation gradient in a semi-arid environment. Soil Biology and Biochemistry 34: 1749-1757.

Stevens, R., and S.B. Monsen. 2004. Guidelines for restoration and rehabilitation of principal plant communities. Chapter 17, in S.B. Monsen, R. Stevens, and N.L. Shaw (compilers), Restoring Western Ranges and Wildlands, v. 1. . General Technical Report RMRS-GTR-136. U.S. Forest Service, Rocky Mountain Research Station., Ft. Collins, CO. pp. 199-294.

Stewart, G. and A,C, Hull, 1949. Cheatgrass (*Bromus tectorum* L.): An ecological intruder in southern Idaho. Ecology 30: 58-74.

Stohlgren, T.J., D. Binkley, G.W. Chong, M.A. Kalkhan, L.D. Schell, K.A. Bull, Y. Otsuki, G. Newman, M. Bashkin, and Y. Son. 1999. Exotic plant species invade hot spots of biodiversity. Ecological Monographs 69: 25-46.

Stringham, T.K., W.C. Krueger, and P.L. Shaver. 2001. States, transitions, and thresholds: Further refinement for rangeland applications. Special Report 1024. Agricultural Experiment Station, Oregon State University, Corvallis, OR. 15 pp.

ter Braak, C. J. F., and P. Smilauer. 1999. CANOCO v. 4.02. Center for Biometry, Wageningen, The Netherlands.

ter Braak, C. J. F. 1986. Canonical correspondence analysis: A new eigenvector technique for multivariate direct gradient analysis. Ecology. 67**:** 1167-1179.

ter Braak, C. J. F. 1987. The analysis of vegetation-environment relationships by canonical correspondence analysis. Vegetatio. 69: 69-77.

Trombulak, S.C., and C. A Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology 14: 18-30.

Tyser, R.W., and C.A. Worley. 1992. Alien flora in grasslands adjacent to road and trail corridors in Glacier National Park, Montana (U.S.A.).   Conservation Biology 6: 253-262.

Uresk, D.W., J.F. Cline, and W.H. Rickard. 1976. Impact of wildfire on three perennial grasses of south-central Washington. Journal of Range Management 29: 309-310.

Uresk, D.W., W.H. Rickard, and J.F. Cline. 1980. Perennial grasses and their response to a wildfire in south-central Washington. Journal of Range Management 33: 111-114.

USFS (U. S. Forest Service). 2001. Fire Effects: *Bromus tectorum*. U. S. Forest Service, Fire Effects Information System. Available online at: http://www.fs.fed.us/database/feis/plants/graminoid/brotec/fire effects.html.

Valentine, N. 2001. Danger pits location. Email correspondence to J. Haas, Deputy Project Leader, Hanford Reach National Monument, 4/16/2001.

Walker, S.C., R. Stevens, S.B. Monsen, and K.R. Jorgensen. 1995. Interaction between native and seeded introduced grasses for 23 years following chaining of juniper-pinyon woodlands. In B.A. Roundy, E.D. McArthur, J.S. Haley, and D.K. Mann (compilers), Proceedings: Wildland Shrub and Arid Land Restoration Symposium. General Technical Report INT-GTR-315. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 372-380.

Wambolt, C.L. 2001. Mule deer foraging preference among five sagebrush (*Artemisia* L.) taxa. Western North American Naturalist 61: 490-494.

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

Welch, B.L. 2004. *Artemisia tridentata* Nutt. In J.K. Francis (ed.), Wildland Shrubs of the United States and its Territories: Thamnic Descriptions. Vol. I. General Technical Report IITF-GTR-26. US Forest Service, International Institute of Tropical Forestry, San Juan PR. Pp. 85-87.

Welch, B.L., and C. Criddle. 2003. Countering misinformation concerning big sagebrush. Research Paper RMRS-RP-40. U.S.Department of Agriculture, Forest Service, Rocky Mountain Research Station, Ogden, UT. 28 pp.

West, N.E., and T.P. Yorks. 2002. Vegetation responses following wildfire on grazed and ungrazed sagebrush semi-desert. Journal of Range Management 55: 171-181.

West, N.E., and M.A. Hassan. 1985. Recovery of sagebrush-grass vegetation following wildfire. Journal of Range Management 38: 131-134.

Westoby, M., B. Walker, and I. Noy-Meir. 1989. Opportunistic management for rangelands not at equilibrium. Journal of Range Management 42: 266-274.

Wicklow-Howard, M.C. 1994. Mycorrhizal ecology of shrub-steppe habitat. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service Intermountain Research Station, Ogden, UT. Pp. 207-210.

WNHP (Washington Natural Heritage Program). 2000. Field Guide to Washington's Rare Plants. Washington Department of Natural Resources Natural Heritage Program, Olympia, WA., and U.S. Bureau of Land Management.

WNHP (Washington Natural Heritage Program). 1997. Endangered, Threatened and Sensitive Vascular Plants of Washington – with Working Lists of Non-Vascular Species. Washington Department of Natural Resources Natural Heritage Program, Olympia, WA.

Whisenant, S.G., 1999. Repairing Damaged Wildlands: A Process-Oriented, Landscape-Scale Approach. Cambridge University Press, Cambridge, UK. 312 pp.

Whisenant S.G. 1995. Landscape dynamics and arid land restoration. In B.A. Roundy, E.D. McArthur, J.S. Haley, and D.K. Mann (compilers), Proceedings: Wildland Shrub and Arid Land Restoration Symposium. General Technical Report INT-GTR-315. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 26-34.

Whisenant S.G. 1990. Changing fire frequencies on Idaho's Snake River Plains: ecological and management implications. In E.D. MaArthur, E.M. Romney, S.D. Smith, and P.T. Tueller (eds.), Proceedings – Symposium on Cheatgrass Invasion, Shrub Die-off, and Other Aspects of Shrub Biology and Management. General Technical Report INT-GTR-276. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 1-7.

Wilderman, D.W. 1994. Plant Communities of the Fitzner/Eberhardt Arid Lands Ecology Reserve and the North Slope of the Hanford Site: Findings of the 1994 Inventory. The Nature Conservancy of Washington, Seattle, WA. 51 pp. plus appendix.

Wilson, A.A.G. 1988. Width of firebreak that is necessary to stop grass fires: Some field experiments. Canadian Journal of Forest Research 18: 682-687.

Wright, H.A. 1985. Effects of fire on grasses and forbs in sagebrush-grass communities. In K. Sanders and J. Durham (eds.), Rangeland Fire Effects: Proceedings of a Symposium Held 27-29 November, 1984. U.S. Bureau of Land Management, Boise, ID. Pp. 12-21.

Wright, H.A. 1971. Why squirreltail is more tolerant to burning than needle-and-thread. Journal of Range Management 24: 277-284.

Wright, H.G., L.F. Neuenschwander, and C. M. Britton. 1979. The Role and Use of Fire in Sagebrush-Grass and Pinyon-Juniper Plant Communities: A State-of-the-Art Review. US Forest Service, Intermountain Research Station, Ogden, UT. General Technical Report GTR-INT-58. 48 pp.

Wright, H.A., and J.O. Klemmedson. 1965. Effect of fire on bunchgrasses of the sagebrush-grass region of southern Idaho. Ecology 46: 680-688.

Yensen, D.A., and G.W.Smith. 1984. Big sagebrush-winterfat and big sagebrush-Nuttall saltbush mosaic vegetation in southwestern Idaho. In A.R. Tiedman, E.D. McArthur, H.C. Stutz, R. Stevens, and K.L. Johnson (compilers), Proceedings: Symposium on the Biology of Atriplex and Related Chenopods. General Technical Report INT-172. U.S. Forest Service Intermountain Research Station, Ogden, UT. Pp. 28-33.

Young, J.A., and R.A. Evans. 1989. Dispersal and germination of big sagebrush (*Artemisia tridentata*) seeds. Weed Science 37: 201:206.

Young, J.A., and R.A. Evans. 1985. Demography of *Bromus tectorum* in *Artemisia* communities. In White, J. (ed.), The Population Structure of Vegetation. Der Junk, Dordrecht. Pp. 489-502.

Young, J.A., and R.A. Evans. 1978. Population dynamics after wildfires in sagebrush grasslands. Journal of Range Management 31: 283-289.

Young, J.A., R.A. Evans, and R.E. Eckert. 1969. Population dynamics of downy brome. Weed Science 17: 20-26.

Young, J.A., R.A. Evans, and R.A.Weaver. 1976. Estimating potential downy brome competition after wildfires. Journal of Range Management 29: 322-325.

Young, J.A., and W.S. Longland. 1996. Impact of alien plants on Great Basin Rangelands. Weed Technology 10: 384-391.

Young, R. P. 1983. Fire as a vegetation management tool in rangelands of the intermountain region. In S.B. Monsen and N. Shaw (compilers), Managing Intermountain Rangelands: Improvement of Range and Wildlife habitats. General Technical Report GTR-INT-157. U.S. Forest Service, Intermountain Research Station, Ogden, UT. pp 18-31.

Youtie, B., J. Ponzetti, D. Salzer, and J. Soll. 1998. Controlling exotic grasses on relict grasslands (Oregon). Restoration and Management notes 16: 209.

Zar, J.H. 1996. Biostatistical Analysis. Prentice Hall. New Jersey. 718 pp.

PERSONAL COMMUNICATIONS

Caplow, F. Botanist. Washington Dept. of Natural Resources, Natural Heritage Program, Olympia, WA.

Downs, J.L. Senior Research Scientist, Plant Ecology. Pacific Northwest National Laboratory, Richland, WA.

Gaston, G. Cultural Resource Manager. US Fish and Wildlife Service, Hanford Reach National Monument, Richland, WA.

McIntosh, T. Bryologist, microbiotic crust ecologist and consultant. Biospherics, Inc., Vancouver, BC.

Newsome, H. Wildlife biologist. US Fish and Wildlife Service, Hanford Reach National Monument., Richland, WA.

Salstrom, D. SEE Botanical Consulting, Tenino, WA.

Skinner, T. Fire Management Officer. US Fish and Wildlife Service, Arid Lands Unit, Pasco, WA.

II. Monitoring Post-fire Rehabilitation on the Fitzner-Eberhardt Arid Lands Ecology Reserve, Hanford Reach National Monument

INTRODUCTION

The 24 Command Fire of June 27 – July 2, 2000, burned nearly all of the 77,000 acre Fitzner-Eberhardt Arid Lands Ecology (ALE) Reserve, a federal Research Natural Area located on the Hanford Reach National Monument (BAER Team, 2000; see also Section I, this volume). Rehabilitation plans for selected portions of the affected area were developed under the authorization of the Burned Area Emergency Rehabilitation (BAER) program following the results of post-fire vegetation assessments (BAER Team 2000, Evans 2001, USFWS 2001, Evans et al. 2002, Smith et al. 2003). Rehabilitation efforts implemented between December 2002 and February 2003 targeted 10,000 acres of the native habitats most seriously impacted by the 24 Command Fire.

In November 2002 The Nature Conservancy (TNC) was engaged to monitor the first two years of rehabilitation efforts and to assess potential threats to the long-term recovery of restored plant communities. Monitoring began in November 2002, during the installation phase of the rehabilitation project, and continued until fall 2004.

MATERIALS AND METHODS

Survival of sagebrush outplantings. More than 700,000 seedlings of Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) were planted on the ALE Reserve in late fall 2002 (Smith et al. 2003). The purpose of these plantings was to "protect and restore the ecological integrity and site productivity of shrub-steppe sagebrush plant communities within the ALE and DOE lands in accordance with established management plan guidelines (BAER Team 2000)". Both bare root and tube-grown sagebrush seedlings were used in the installation. Bare root plants were supplied by Lucky Peak Nursery (U.S. Forest Service, Boise, ID). Four-inch tube-grown seedlings were supplied by Bitterroot Restoration, Inc. (Corvallis, MT) and Buffaloberry Nursery (McCall, ID). All seedlings were grown from wild-collected seed from the Lower Columbia Basin.

Seedlings were planted at a specified stocking rate of 450 plants/ acre (1112 plants/ ha) in 12 polygons covering approximately 1500 acres in total (Fig. 2.1). The polygons, which ranged in size from 33 acres to 600 acres, were dispersed along the 1200 Foot Rd., the main internal access way on the Reserve. The locations of sagebrush planting polygons were selected from areas in high-quality bluebunch wheatgrass (*Agropyron spicatum* = *Pseudoroegneria spicata*) – Sandberg's bluegrass (*Poa sandbergii* = *P. secunda*) grasslands reflecting characteristics of native sagebrush steppe understories. Polygons were located between 900 ft. and 1400 ft. (275-427 m) elevation[1] on silt loam soils on the gently sloping low to middle slopes of Rattlesnake Mountain and the Rattlesnake Hills. Plants were installed by three experienced independent contractors in late November through mid-December 2002 (Table 2.1). Bare root and tubling stocks were planted in separate polygons. Immediately prior to planting, bare root plants in four

[1] Most of the planted areas fell within a narrower range, between approximately 1000 - 1300 ft. (305 – 396 m).

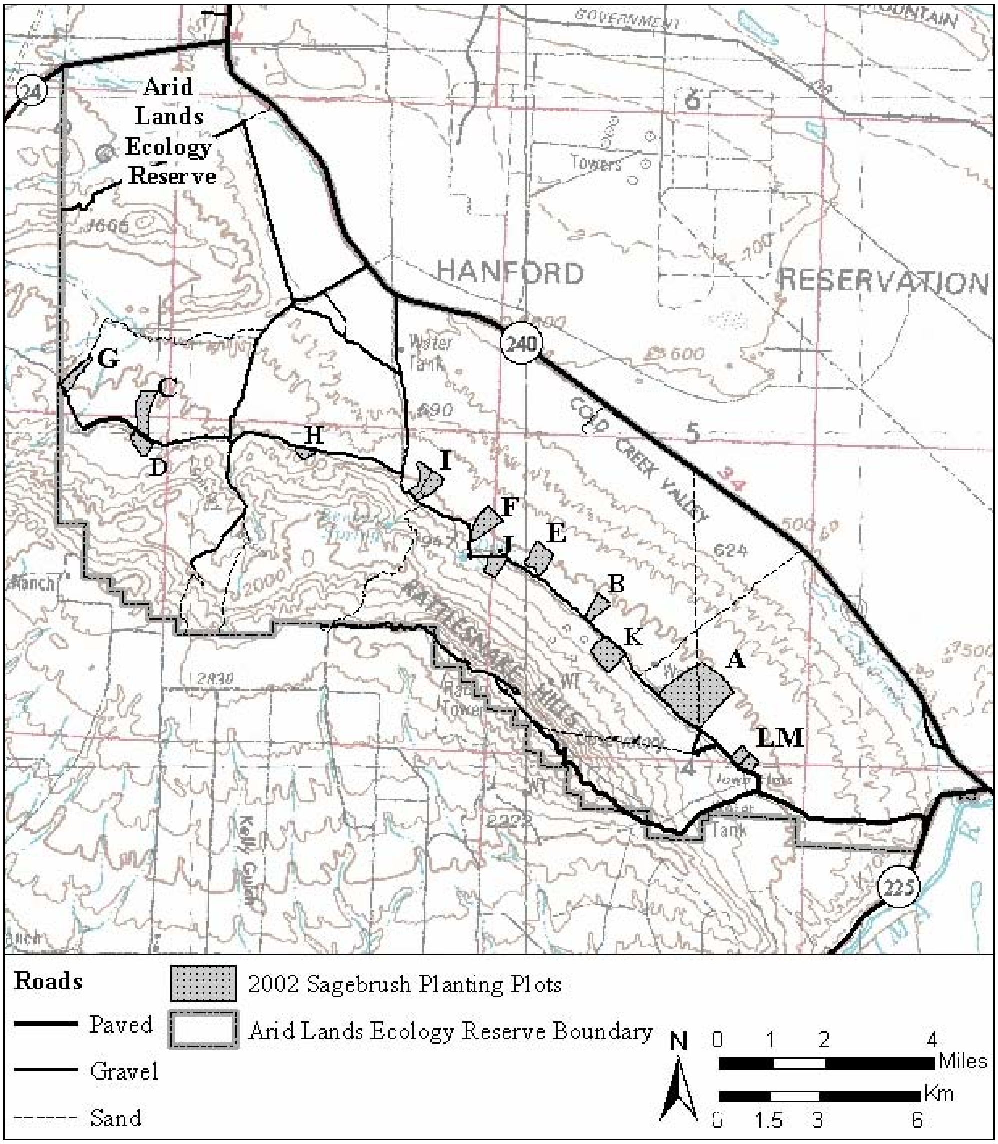


Figure 2.1. Location of Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) outplanting polygons on the Arid Lands Ecology Reserve, November – December, 2002. Monitoring plots were located in polygons A, B, C, D, F, J, K, and LM. Figure courtesy of Hanford Reach National Monument.

polygons were treated with Plant Success™ endomycorrhizal inoculum root dip hydrogel (Mycorrhizal Applications, Grants Pass, OR). Bare root plants in the remaining two polygons were treated with Terra Sorb™ (Plant Health Care, Inc., Pittsburgh, PA), a nonmycorrhizal hydrogel.

During winter 2002-2003 a total of 26 plots were established within eight[2] of the sagebrush polygons to monitor the survival of outplanted big sagebrush stock (Table 2.1). Plot locations within polygons were determined randomly using GIS (Hooge and Eichenlaub 2000). Random points that fell within unplanted areas within polygons (e.g., weedy draws which planting crews were instructed to avoid) were passed over in favor of the next random point. Three plots were installed in each of seven polygons; an eighth polygon, 3-4 times the size of the next largest polygon, received 5 plots. All polygons were stratified into three segments of roughly equal size in order to assure dispersion of plots across the polygons. Plots within polygons were similar in soil type and other readily observable environmental variables (Table 2.2). Annual precipitation throughout the planting zone is approximately 7.0 in. – 8.0 in. (178 – 203 mm).

Monitoring methodology closely followed protocols used by Monument personnel for monitoring survival of shrub seedlings planted in 2000 and 2001. Sample plots consisted of a 100 m x 12 m belt transect bisected lengthwise by a 100 m baseline. Baseline transects were uniformly oriented towards magnetic north from the randomly selected origins. The position of individual sagebrush plants was recorded in terms of distance along the baseline from the origin and perpendicular distance from the baseline at that point. Position right or left of the baseline was recorded as plus (+) or minus (-) respectively. The aim was to capture approximately 100 seedlings within each plot. Actual plots contained a total of 2814 seedlings or 108.2 (± 13.8 SD) seedlings / plot. A few plots were planted so densely that recording was halted before reaching 100 m.

To assess the effects of seedling size on survival, plants in four polygons (polygons B, C, F, and J) were recorded in one of two size classes at time of planting: (a) > 15cm tall, and (b) < 15cm tall. Individual plants whose size was on the border between the two size classes were assigned to a class based on morphology: multibranched individuals were placed into the larger of the two size classes, while single-stemmed individuals were placed into the smaller size class.

In order to assess first and second year survival, sagebrush plots were resampled during October in both 2003 and 2004. Each seedling was relocated according to its grid coordinates and plant condition was recorded in one of the following three states: alive, appearing healthy ('alive'); alive, appearing stressed ('stressed'); and dead (Table 2.3). Both total percent survival ('alive' plants + 'stressed" plants) and the percentage of 'alive' plants only were used in comparisons between treatments.

In order to examine the potential influence of interspecific competition on the survival of sagebrush outplantings, percent cover of cheatgrass and native perennial bunchgrasses was estimated using a line intercept method during October 2004. Canopy cover of individual clumps of native bunchgrasses and generalized canopy cover of

---

[2] Four of the sagebrush outplanting polygons were planted as mitigations for construction of the Environmental Restoration and Disposal Facility (ERDF) on Central Hanford. These polygons were monitored by Bechtel, Inc., and results are not included in this report.

Table 2.1. Summary of Wyoming big sagebrush outplanting polygons with associated treatment variables.

Polygon	Size (Acres)	# plots	Condition	Hydrogel	Mycorrhizae	Contractor
A	600	5	Tube			Bitterroot Restoration
LM	85	3	Tube			Bitterroot Restoration
B	101	3	Bare Root	X		Wildlands, Inc.
K	102	3	Bare Root	X		Wildlands, Inc.
C	132	3	Bare Root	X	X	Frank Meduzia
D	76	3	Bare Root	X	X	Frank Meduzia
F	119	3	Bare Root	X	X	Frank Meduzia
J	68	3	Bare Root	X	X	Frank Meduzia

Table 2.2. Environmental variables associated with Wyoming big sagebrush outplanting plots on the ALE Reserve, 2002-2003.

Plot	Condition	Hydrogel	Mycorrhizae	Slope (^{o})	Aspect (^{o})	Heat Load Index	Elevation (ft.)
A4	Tube			8	115	0.8475	1000
A5	Tube			5	330	0.8702	1080
A9	Tube			2	30	0.8741	1200
A15	Tube			3	135	0.8874	1040
A17	Tube			5	290	0.8989	1020
B5	BR	X		2	100	0.8807	1120
B6	BR	X		5	40	0.8393	1180
B9	BR	X		2	40	0.8736	1120
C1	BR	X	X	5	55	0.8397	1200
C4	BR	X	X	3	200	0.9103	1200
C11	BR	X	X	5	140	0.8848	1120
D1	BR	X	X	5	360	0.8514	1280
D2	BR	X	X	10	50	0.7791	1350
D3	BR	X	X	8	380	0.8096	1360
F1	BR	X	X	5	360	0.8514	1110
F3	BR	X	X	5	30	0.8405	940
F6	BR	X	X	3	40	0.8623	1080
J1	BR	X	X	5	30	0.8405	1280
J2	BR	X	X	5	25	0.8416	1320
J4	BR	X	X	8	380	0.8096	1390
K4	BR	X		5	45	0.8391	1240
K5	BR	X		5	40	0.8393	1200
K24	BR	X		5	380	0.8430	1260
LM2	Tube			2	70	0.8751	1120
LM3	Tube			2	380	0.8751	1080
LM5	Tube			2	90	0.8784	1080

Table 2.3. Criteria for survival scores of Wyoming big sagebrush outplantings.

Condition	Description
Alive	Plants generally robust. Stems flexible. Foliage supple, expanded; color mostly green and grey-green.
Stressed	Live stems still flexible. If multiple stems one or more stems may be dead; if single stems, one or more branches may be dead. Foliage somewhat dry and brittle, sparse, or incompletely expanded; color mostly grey, yellow-grey, or white.
Dead	Stems brittle, sometimes broken. Foliage white to brown and brittle where present, or lacking.

cheatgrass patches were measured to the nearest 5.0 cm along five randomly selected 10 m sections of the 100 m baseline transect of each sagebrush survival plot.

Seeding density and seedling emergence. The purposes of aerial seeding treatments were to stabilize soils, suppress invasive species, and promote ecological recovery of the shrub-steppe ecosystem on ALE (USFWS 2001). Areas targeted for treatment by direct seeding totaled approximately 10,500 acres (4250 ha). Seed mixtures were developed for low and high elevation habitats within the target areas. Mixes were composed primarily of native grasses and cultivars of native grasses (Table 2.4). A seeding density of 50 seeds/ $ft.^2$ (538 seeds/m^2) or 16 - 18 lbs. pure live seed (PLS)/ acre by weight was specified in the rehabilitation plan. Seed mixes were supplied by L&H Seed Company (Connell, WA). Seeds were aerially applied to 5 polygons in December 2002 and January 2003 (Fig. 2.2). Polygon 3 was divided into high elevation (> 800 ft.) and low elevation (< 800 ft.) seed mix areas. The shrub seed polygon (SS) was dedicated for the aerial seeding of Wyoming big sagebrush. Because of difficulties feeding the minute, extremely light seeds of Wyoming big sagebrush through the seeding apparatus, leftover low elevation seed mix, along with inert material, was used as a carrier for the sagebrush seeds in this polygon.

A sixth polygon (DS), located along the SR 240 highway corridor in Cold Creek Valley, was drill seeded using a rangeland drill. Two cultivars and a special collection of bluebunch wheatgrass were installed in approximately equal areas of this corridor. Anatone bluebunch wheatgrass (*Agropyron spicatum* = *Pseudoroegneria spicata*; NSN 2004a) was installed from Gate 120 south to milepost 8; Secar bluebunch wheatgrass (also known as Snake river wheatgrass, *Elymus wawaiensis*; NSN 2004b) was installed from milepost 8 south to milepost 14; and a special collection of bluebunch wheatgrass from the Hell's Canyon area (R. Herman pers. comm.) was installed from milepost 14 south to Horn Rapids Rd., then west to gate 106 (J. Meisel pers. comm.).

Both previously-established permanent reference plots[3] and newly-established plots were used to monitor seed rain, seedling emergence, and potential recruitment. New 'rehabilitation' plots were established as permanent plots in December 2002 and in April – June 2003. Long-term permanent plots already established within the rehabilitation area had the advantage of pre-treatment vegetation data including percent cover and density of cheatgrass. Newly-established rehabilitation plots were located in a stratified random fashion (Hooge and Eichenlaub. 2000) in areas within polygons that were not adequately represented in the long-term reference plot array. Rehabilitation plots were oriented around 100 m baseline transects uniformly oriented towards magnetic north from randomly selected origins.

The number of transects within polygons was determined by a combination of factors including polygon size, landscape and cover type diversity, and access. The largest polygon, P3, was also the most diverse in terms of topography, vegetation and landscape condition, and had a large array of long-term permanent plots. Polygon P4 was also large, but was less diverse and had no previously established vegetation plots within its boundaries. Lacking sagebrush in 2000, the vegetation in Polygon P4 experienced a less severe burn during the 24 Command Fire than did the rest of the project area, and, as a result, still exhibited high cover of native bunchgrasses, although cheatgrass abundance was also relatively high. This polygon was the most remote from access points.

Seed traps were built of ¾ inch x 1½ inch lumber with bottoms made from plastic contact paper secured to the frame with duct tape. Traps were constructed so that inside dimensions were 20 cm x 20 cm. Contact paper bottoms and 1½ inch sides minimized seeds blowing out of traps. Three seed traps were randomly located along transects in 27 new and selected pre-existing plots in December 2002 (Table 2.5). Traps were collected as quickly as possible, in most cases within 24 hours following seeding operations in December 2002 and January 2003. Unfavorable flying conditions interrupted aerial seeding efforts in late December 2002 and disrupted the sample collection schedule. As a result, some traps went uncollected for up to a week. Trapped seeds were later identified to species and counted for comparison to seed mix specifications.

Vegetation data were collected from all permanent and newly established plots within the rehabilitation project area during spring 2003 and 2004. Long-term permanent vegetation plots on ALE were established using several different methodologies (Table 2.6). Methods used in sampling newly-established rehabilitation plots were consistent with those used in Hanford Biological Resources Management Plan (BRMaP) Plots and Steppe-in-Time (SIT) Plots (see Section I, this volume). In each rehabilitation plot, visual estimates of percent cover of vascular plant species, microbiotic crust, and plant litter were recorded to the nearest full percent within 20cm x 50cm ($0.1m^2$) microplots (n = 20) arrayed randomly within 5m stratifications along the baseline transect. The minimum score for a trace occurrence within each microplot was 1.0%. Biodiversity

[3] Permanent reference plots installed under the Hanford Biological Resources Management Plan (BRMaP), the Steppe-in-Time project (SIT) and the Hanford Biodiversity Project are described in Section I of this report.

Plots used a cover-class system to estimate cover of vascular plant species and MBC (see Section I, this volume).

Surveys for initial seedling emergence (2003) and potential establishment (2004) within a subset of microplots (n = 10) in all rehabilitation plots and permanent vegetation plots within the project area. In BRMaP and Rehabilitation plots surveys were conducted in every second microplot along each vegetation transect. In SIT Plots seedling surveys were conducted in every microplot within the first 100 m of the 200 m transects. In Biodiversity Plots surveys were conducted in every third microplot of cheatgrass density arrays (Section I, this volume). Seedlings were identified to species and counted within the entire 20 x 50 cm microplot. Recruitment densities were counted in 2004 in the same microplots that were surveyed for emergence during 2003. During the 2004 field season we were unable to distinguish consistently and confidently between seedlings surviving from 2003 and those newly emerged in 2004. For this reason 2004 seedling densities are reported as potential recruitment. Potential recruitment may not strictly represent second-year survival of seedlings emerged in 2003 but may be inflated by seedlings that were newly emerged in 2004. Specific instances where emergence of large numbers of new seedlings during fall 2003 or spring 2004 may affect recruitment figures will be discussed in later sections.

A simple emergence rate was calculated for seedlings emerged in 2003 by calculating the average seedling density /m^2 per polygon and dividing this figure by the average seed density /m^2 per polygon. Potential recruitment was calculated by dividing the density of seedlings surveyed in 2004 by the average seed density /m^2 per polygon. While these figures cannot be compared statistically between polygons, they do allow for general comparisons between species and treatments as well as comparisons with other studies.

Cheatgrass abundance and the effects of herbicide treatments. In order to temporarily relieve native seedlings from competition with cheatgrass (*Bromus tectorum*) during the establishment phase of rehabilitation efforts, glyphosate (Roundup™) herbicide was applied to all seeding polygons at 3.5 oz./acre between November 14 and December 10, 2002. A late winter followup treatment was also applied between February 18 and 20, 2003, to portions of the project area below 800 ft. (244 m) elevation only. Cover and density of cheatgrass within the project area were measured during the course of vegetation surveys in all permanent and newly established plots during spring 2003 and 2004. Density of cheatgrass was counted within the 20 x 20 cm portion of each microplot nearest to transect baselines. To assess changes attributable to herbicide effects, post-treatment values from long-term permanent plots were compared to pre-treatment values from the same plots. Values from the long-term plots were also compared to newly established plots to qualify generalizations inferred from trends observed in permanent plot data. Long-term permanent plots from throughout the ALE Reserve outside the rehabilitation project area were used as control comparisons for cheatgrass abundance measurements in treatment areas. Cheatgrass cover and density data from 53 permanent long-term vegetation plots and density data only from 14 transition density plots outside the rehabilitation area were collected both in the course of a reserve-wide monitoring effort to assess cheatgrass abundance and plant community recovery following the 24 Command Fire (Section I this volume).

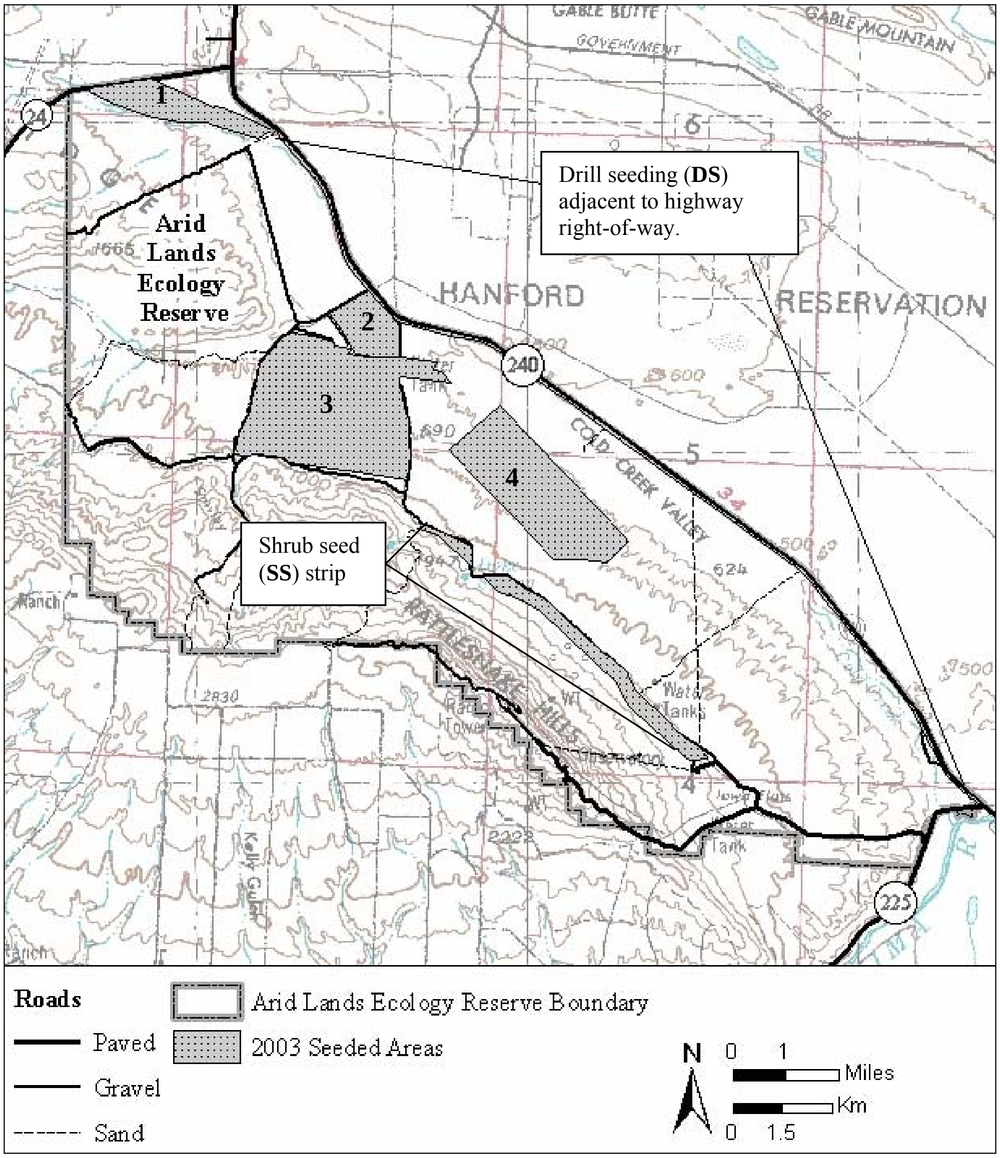


Figure 2.2. Location of aerial and drill seeding polygons on the Arid Lands Ecology Reserve, December 2002 – January 2003. See Table 2.6 and text for description. Figure courtesy of Hanford Reach National Monument.

Recovery of bulldozer fireline. During the June 2000 wildfire a short (1/4 mile) bulldozer fireline was cleared in otherwise high-quality grassland vegetation in upper Snively Basin. The fire subsequently burned around the suppression swath and through the surrounding vegetation. During the spring of 2001, USFWS personnel and volunteers manually restored the dozer berms to natural contours and hand-broadcast seed of Sandberg's bluegrass (*Poa sandbergii* = *P. secunda*) over the disturbed area. Rehabilitation activities were mandated by the BAER plan "to protect habitats from noxious weed infestation, visual intrusion on the landscape, and to minimize fragmentation of ecological areas (BAER Team 2000)." The affected area is located on silt loam soils on moderate slopes (10-15º) between 2200 – 2500 ft. (670 – 760 m).

The Snively Basin treatment area and the adjacent grassland were surveyed during June 2003 and June 2004 to assess the effectiveness of restoration methods and the recovery of the area. Three non-permanent 48 m transects were installed down the center of the rehabilitation swath in three locations selected to represent the range of variation in vegetation within the feature. Vegetation percent cover data were collected within six pairs of 20 cm x 50 cm microplots placed randomly within 8 m stratifications along each transect. Cheatgrass density data were collected in 20 cm x 20 cm sections within each microplot. One microplot of each pair was located within the center of the disturbed area (internal microplots) while the other microplot of the pair was located a distance of 6 m from the baseline in grassland that had not been affected by the suppression efforts (external microplots). External microplots alternated from side to side along the baseline transect. Vegetation cover and cheatgrass density data from internal microplots were compared with data from external microplots to assess differences between the treatment area and the surrounding grassland. Data from 2003 were compared to data from 2004 and to evaluate changes in each area over time.

OTHER METHODS

GPS Technology and Plot Locations. Plot location coordinates were recorded using a Garmin *etrex* Personal Navigator portable GPS unit. Location coordinates for all of the plots used in this study are presented in Appendix A. Operational settings used with the portable GPS unit were as follows:

Time: Zone – Pacific
Format – 12 Hr
Position format: UTM/ UPS
Map Datum: NAD27 CONUS
North Reference: Grid

Environmental Variables. Environmental variables associated with sample plots are presented in Appendix B. Slope (degrees), aspect, and elevation were recorded for every plot visited. Aspect values were recorded as magnetic bearings in the field and later corrected for declination. Prior to correlation analyses, aspect was rescaled symmetrically about the NE-SW axis to a scale of zero (the coolest aspects, 45^o) to one (the warmest aspects, 225^o) according to the method of McCune and Keon, (2002). A heat load index (HLI) incorporating the influence of slope angle as well as aspect upon solar incidence was also calculated following McCune and Keon (2002, Eqn. 3). See

Section I (this volume) for more detailed explanations of the calculations for these parameters.

Each plot was identified as belonging to one of three general soil types, silt-loam, stony silt-loam, or sand, based on Hajek (1966) and Rasmussen (1971). Data on monthly and seasonal precipitation were taken from the Hanford Meteorological Station (HMS 2004).

Percent cover of litter was visually estimated along with cover of vegetation in BRMaP and SIT microplots. Litter was recorded in Biodiversity Plots only in 2004, when it was visually estimated in 20cm x 50cm microplots that were extensions of the 20cm x 20cm microplots used for cheatgrass density counts in those plots (Section I, this volume). Since litter data was not available for all plots prior to 2004, only 2004 litter data was used as a variable in correlation tests.

Elevations within the treatment areas ranged from approximately 550-1000 ft. (165 – 305 m). Average annual precipitation varyied from about 6.25” (160 mm) per year at the lower elevations to no more than 8” (203 mm) per year at the highest elevations within the project area.

Statistical analyses. Within-year comparisons between polygons, plots, and treatments were tested for significance using single factor analysis of variance (ANOVA). Following significant results from ANOVA, pairwise comparisons were made using Tukey’s multiple comparisons test ($\alpha = 0.05$); when pairwise comparisons exceeded 20 the Scheffé test ($\alpha = 0.05$) was used. Between-year comparisons were made using two-tailed paired-sample *t* tests. Comparisons between non-paired plots were made using an independent two-sample *t* test. Comparisons between more than two years (as with some data from long-term permanent plots) were made using repeated measures ANOVA. Significant between-year differences indicated by repeated measures ANOVA were further investigated using a single paired-sample *t* test comparing only the two years of primary interest. Vegetation differences between the rehabilitated bulldozer fireline and the surrounding grassland in Snively Basin were also analyzed using two-tailed paired-sample *t* tests.

Correlations between treatment data and community and environmental factors were explored using Pearson’s product-moment correlation. The significance of correlations greater than $r = 0.30$ was tested using simple regression (Zar 1996, Elzinga et al. n.d.).

Scientific nomenclature. Vascular plant taxonomy generally follows Hitchcock and Cronquist (1973). Revised nomenclature is taken from Kartesz (1999; see Appendix C).

Table 2.4. Seeding mixtures used in the 24 Command Fire rehabilitation project: a) aerial seeding mixes; b) drill seeding mixes. Quantities are expressed in lbs. pure live seed (PLS) /acre.

a. Aerial seeding mixes

LOW ELEVATION MIX

Common Name	Scientific Name	Source or Variety	PLS/acre	% of Mix
Thickspike Wheatgrass	*Agropyron dasystachyum* (=*Elymus lanceolatus*)	'Schwendimar'	6	33
Indian Ricegrass	*Oryzopsis hymenoides*		6	33
Sandberg's bluegrass	*Poa sandbergii*	Hanford or E. WA.	5	28
Squirreltail	*Sitanion hystrix*	'Sandhollow'	1	5
Needle and Thread	*Stipa comata* (= *Hesperostipa comata*)	Hanford or E. WA.	0.25	1
Yarrow	*Achillea millefolium*	*lanulosa* var. *lanulosa*	0.1	T

HIGH ELEVATION MIX

Common Name	Scientific Name	Source or Variety	PLS/acre	% of Mix
Bluebunch Wheatgrass	*Agropyron spicatum* (=*Psuedoroegneria spicata*)	Secar, Anatone, Hells Canyon	10	63
Sandberg's bluegrass	*Poa sandbergii*	Hanford or E. WA.	5	31
Squirreltail	*Sitanion hystrix*	'Sandhollow'	1	6
Yarrow	*Achillea millefolium*	*lanulosa* var. *lanulosa*	0.1	T

SHRUB MIX

Common Name	Scientific Name	Source or Variety	PLS/acre	% of Mix
Wyoming Big Sagebrush	*Artemisia tridentata*	ssp. *wyomingensis*	0.2	--

b. Drill seeding mixes

Common Name	Scientific Name	Source or Variety	PLS/acre	% of Mix
Bluebunch Wheatgrass	*Agropyron spicatum* (=*Psuedoroegneria spicata*)	Secar, Anatone, Hells Canyon*	10	76.3
Sandberg's bluegrass	*Poa sandbergii*	Hanford or E. WA.	2	15.3
Squirreltail	*Sitanion hystrix*	'Sandhollow'	1	7.6
Yarrow	*Achillea millefolium*	*lanulosa* var. *lanulosa*	0.1	0.8

* Individual wheatgrass stocks were drill seeded into separate portions of the drill seed area, with each stock planted over approximately one-third of the total area.

Table 2.5. Summary of aerial and drill seeding polygons and seed and vegetation transects within the rehabilitation project area. Vegetation, seedling emergence, and cheatgrass density were sampled in all plots. Numbers in parentheses indicate number of plots used for seed trapping.

Polygon	Size (Acres)	Soil Type	Seed Mix (Elev.)		Number of Permanent Transects in Polygon (# used as seed plots)				
					New	Permanent Transects			
			High	Low	Transects	BRMaP	Biodiversity	SIT	T plots
1	865	Loam		X	3 (2)		2 (1)		
2	696	Sand		X	3 (1)	5 (2)			
3	3609	Loam	X	X	6 (6)	5 (2)	3 (1)	2 (2)	4 (3)
4	3047	Loam		X	3 (3)				
Shrub Seed	1369	Loam	Shrub Seed Mix		2 (2)	1 (1)			1 (1)
Drill Seed	900	Sand, Loam	drill seed mix		12				
Totals	10,486				29 (14)	11 (5)	5 (2)	2 (2)	5 (4)

Table 2.6. Summary of permanent plot characteristics, ALE Reserve. Complete descriptions of vegetation and density sampling methods for permanent reference plots are presented in Section I of this report.

Plot Type	Plot Description	Vegetation microplot unit	Vegetation microplot array	Cheatgrass density data				Density microplot array*
				2001	2002	2003	2004	
BRMaP	100m transect	20x50cm microplot	Regular, @ 5m	X	X	X	X	With vegetation microplots
SIT	200m transect	20x50cm microplot	Regular, @ 10m		X	X	X	With vegetation microplots
Biodiversity	5x20m plot	5x20m plot	no microplots	X	X	X	X	30 microplots arrayed randomly along parallel 50m transects along & beyond long sides of plot
Transition	500-1500m transect	None	NA	X	X	X	X	At 25m intervals
Rehab-ilitation	100m transect	20x50cm microplot	Random, w/in 5m stratifications			X	X	With vegetation microplots

*All density microplots are 20cm x 20cm units or 20cm x 20cm portions of vegetation microplots

FINDINGS

Big sagebrush survival. Overall planting density of Wyoming big sagebrush averaged 410.2 plants/ acre (± 128.9 SD [1013.3 plants/ ha ± 318.3 SD]) within the study plots compared with a target planting density of 450 plants / acre (1111.5 plants/ ha). Density within sample plots ranged between 269.9 and 857.0 plants/ acre, with values for 22 of 26 sample plots falling within one standard deviation of the mean (281.3 – 539.1 plants /acre). Average planting density by polygon varied between 311.7 plants/ acre (± 16.4 SD) in polygon D and 530.3 plants/ acre (± 218.6 SD) in Polygon B. Differences between polygons were not significant ($P = 0.251$; Table 2.7). Sample densities that matched or exceeded project specifications were recorded in seven of 26 plots (26.9%), but more than half of all plots had densities of at least 80% of specifications, and 21 plots (80.8%) had densities of at least 70% of specifications.

Overall mean percent survival of big sagebrush seedlings two years after planting was 36.3% (± 20.6 SD; Table 2.8a). Percent survival of sagebrush outplantings by polygon through October 2004 ranged from as low as 4.8% (± 6.6 SD) to as high as 76.6% (± 10.5 SD). The highest rates of survival were exhibited in bare root polygons B and K, and in tubling polygon LM. Differences between-polygon differences were similar whether survival was calculated using all live seedlings (alive + stressed plants) or using only robust live seedlings (table 2.8a,b).

Bare root seedlings that were *not* treated with a mycorrhizal inoculant exhibited a significantly higher rate of survival (71.6% ± 8.8% SD) than either tubling plants or bare root seedlings that were treated with a mycorrhizal inoculant prior to planting (Table 2.9a). Bare root plants inoculated with mycorrhizae exhibited the lowest overall survival (13.5% ± 11.2 SD). The survival rate of tube-grown stock was intermediate between that of bare root plants in the mycorrhizal and non-mycorrhizal treatments.

Survival of bare root seedlings did not differ among plants in tall and short size classes and control (tall + short seedlings) values regardless of whether survival was calculated using all live seedlings (alive + stressed plants) or using only robust live seedlings (Table 2.10a,b). Plots with seedlings treated with mycorrhizal inocula made up 9 of twelve plots used in this analysis. The mean survival rate over the entire size class sample (< 30.0%) was strongly influenced by low rates of survival (< 20.0%) in the mycorrhizal treatment plots. Survival rates of seedlings treated only with a non-mycorrhizal hydrogel exceeded 60% (Table 2.10b).

Sagebrush outplantings that appeared healthy and robust in 2003 generally remained so in 2004: 87.4% of outplantings rated ‘alive’ in 2003 were also recorded as ‘alive’ the following year, and fewer than 10.0% of these live individuals either died (7.3%) or turned missing (1.0%) between years. For seedlings recorded as ‘stressed’ in 2003, more than half were observed to be either dead (48.1%) or missing (3.7%) in 2004. Just under 40.0% of the stressed seedlings were recorded as having transitioned into ‘alive’ condition between years, while 8.9% remained in stressed condition.

Mean percent cover of cheatgrass in the 26 sagebrush survival plots was 4.7% (± 7.7 SD). Cheatgrass cover ranged between zero and 23.6% in the 26 plots and mean percent cover in the eight sagebrush polygons ranged from 0.1% to 19.8% (Table 2.11). Percent cover of cheatgrass in Polygons C (19.8% ± 5.1 SD) and B (10.8% ± 8.4 SD) was significantly greater than cover in all other polygons. Percent cover of cheatgrass was significantly greater in polygons where bare root seedlings were planted compared to tubling polygons (Tabele 2.12).

Bluebunch wheatgrass, Sandberg's bluegrass, and, occasionally, Cusick's bluegrass constituted the native perennial bunchgrass component of the sagebrush survival plots. Percent cover of bluebunch wheatgrass in the 26 plots ranged between 10.8% and 28.8%, with an overall mean cover of 19.5% (± 6.5 SD). Percent cover of all native perennial bunchgrasses ranged between 23.4% and 40.3%, with a mean cover of 30.4% (± 7.1 SD). These two parameters did not differ statistically either between polygons or between treatments (Tables 2.11 and 2.12).

Percent cover of all (alien annual plus native perennial) grasses in the 26 sagebrush survival plots ranged between 23.6% and 47.0%, with a mean cover of 33.4% (± 7.8 SD). Polygon C exhibited the highest value for total cover, and this value was significantly greater than total cover in Polygons A, B, F, J, and K.

The presence of the mycorrhizal inoculation treatment was strongly negatively correlated with total survival of big sagebrush outplantings ($r = -0.81$; $P < 0.0001$; Table 2.13). Slope was the only other variable that was even moderately correlated with outplanting survival. Percent cover of native perennial and alien annual grasses, variables that might be indicators of interspecific competition, were only weakly or not at all correlated with sagebrush survival.

Table 2.7. Density of Wyoming big sagebrush outplantings in ALE Reserve rehabilitation plots, November-December 2002: a) by plot; b) by polygon. Planting specifications were 450 plants/ acre (1112 plants/ ha).

a.

Polygon	Transect	Density plants/ha	Density plants/A
A	4	825.0	334.0
A	5	2116.7	857.0
A	9	758.3	307.0
A	15	1240.7	502.3
A	17	1333.3	539.8
B	5	1241.7	502.7
B	6	1033.3	418.4
B	9	1654.8	669.9
C	1	833.3	337.4
C	4	725.0	293.5
C	11	950.0	384.6
D	1	746.7	302.3
D	2	816.7	330.6
D	3	746.4	302.2

Polygon	Transect	Density plants/ha	Density plants/A
F	1	950.0	384.6
F	3	1055.6	427.4
F	6	883.3	357.6
J	1	950.0	384.6
J	2	841.7	340.8
J	4	1218.8	493.4
K	4	1125.0	455.5
K	5	966.7	391.4
K	24	666.7	269.9
LM	2	991.7	401.5
LM	3	850.0	344.1
LM	5	825.0	334.0
	Avg	1013.3	410.2
	SD	318.3	128.9

b.

	n	Density (plants/ acre) Mean	SD
A	5	508.0	(219.9)
B	3	530.3	(128.1)
C	3	338.5	(45.6)
D	3	311.7	(16.4)
F	3	389.9	(35.2)
J	3	406.3	(78.6)
K	3	372.2	(94.2)
LM	3	359.9	(36.4)
	P	0.251	

Table 2.8. Percent survival of Wyoming big sagebrush outplantings by polygon: a) all live plants ("alive" + "stressed" plants); (b) 'alive' plants only. Treatments include nursery tublings (Tube), and bare root stock (BR) with (+) and without (--) mycorrhizal inoculation. Numbers within columns accompanied by the same superscript were identified as significantly different by single factor ANOVA followed by Scheffé pairwise multiple comparisons ($\alpha < 0.05$).

a. Alive + Stressed Plants

Polygon	n	Condition	Percent Survival (± SD) 2002 - 2003		2003 - 2004		Overall (2002-04)	
A	5	Tube	46.1	(7.9)	79.0^{ghi}	(9.7)	36.5^{el}	(8.2)
LM	3	Tube	63.1	(15.5)	57.3	(17.3)	57.1^{ijk}	(17.6)
B	3	BR --	82.4^{a}	(2.9)	81.8^{def}	(3.6)	66.5^{fgh}	(2.2)
K	3	BR --	88.2^{b}	(4.1)	86.5^{abc}	(7.4)	76.6^{abcde}	(10.5)
C	3	BR +	40.4	(9.8)	61.0^{j}	(12.2)	25.2^{af}	(12.4)
D	3	BR +	17.0^{ab}	(21.6)	24.9^{adg}	(4.9)	4.8^{bgil}	(6.6)
F	3	BR +	48.9	(16.4)	22.5^{behj}	(17.3)	12.0^{chj}	(12.5)
J	3	BR +	51.7	(23.4)	24.9^{cfi}	(6.1)	11.9^{dk}	(2.4)
P value			0.0002		<0.0001		<0.0001	
Overall Mean			47.6	(22.6)	56.6	(26.9)	36.3	(26.0)

b. Alive Plants Only

Polygon	n	Condition	Percent Survival (± SD) 2002 - 2003		2003 - 2004		Overall (2002-04)	
A	5	Tube	16.1^{a}	(7.1)	67.4^{ghi}	(13.0)	31.0^{a}	(8.1)
LM	3	Tube	29.6	(9.7)	51.8	(19.6)	51.6^{jkl}	(19.8)
B	3	BR --	44.2^{bc}	(5.2)	75.8^{def}	(4.1)	61.7^{fghi}	(2.7)
K	3	BR --	55.4^{adef}	(19.5)	81.1^{abc}	(10.6)	72.0^{abcde}	(13.4)
C	3	BR +	25.0	(8.1)	57.4	(11.2)	23.7^{bf}	(11.4)
D	3	BR +	5.6^{bd}	(5.5)	16.1^{adg}	(14.0)	3.6^{cgj}	(5.4)
F	3	BR +	11.1^{e}	(11.8)	17.1^{beh}	(17.5)	9.6^{dhk}	(12.1)
J	3	BR +	10.0^{cf}	(4.9)	18.0^{cfi}	(4.5)	8.9^{eiml}	(3.0)
P value			<0.0001		<0.0001		<0.0001	
Overall Mean			18.4	(14.1)	49.6	(27.3)	32.6	(25.1)

Table 2.9. Percent survival of Wyoming big sagebrush outplantings by treatment: a) all live plants ('alive' + 'stressed' plants); (b) 'alive' plants only. Treatments include nursery tublings, and bare root stock with and without mycorrhizal inoculation. *P* values are results of single factor ANOVA. Treatments with identical superscripts are significantly different (Tukey, $\alpha = 0.05$).

a. Alive + Stressed Plants

		Percent Survival (± SD)					
Treatment	**n**	**2002-2003**		**2003-2004**		**2002-2004**	
Tublings	8	52.6^{a}	(13.6)	70.9^{b}	(16.3)	44.2^{bc}	(15.5)
Bare root - no mycorrhizae	6	85.3^{ab}	(4.5)	84.1^{a}	(5.8)	71.6^{ab}	(8.8)
Bare root – mycorrhizae added	12	39.5^{b}	(21.3)	33.3^{ab}	(19.3)	13.5^{ac}	(11.2)
P value		< 0.0001		< 0.0001		< 0.0001	

b. Alive Plants Only

		Percent Survival (± SD)					
Treatment	**n**	**2002-2003**		**2003-2004**		**2002-2004**	
Tublings	8	21.2^{a}	(10.2)	61.5^{b}	(16.5)	38.7^{bc}	(16.2)
Bare root - no mycorrhizae	6	48.8^{ab}	(14.1)	78.5^{a}	(7.8)	66.8^{ab}	(10.4)
Bare root – mycorrhizae added	12	12.9^{b}	(10.2)	27.2^{ab}	(21.2)	11.5^{ac}	(10.9)
P value		< 0.0001		< 0.0001		< 0.0001	

Table 2.10. Percent survival of Wyoming big sagebrush outplantings by size class: (a) all plots in sample polygons; (b) non-mycorrhizae plots only (Polygon B); (c) mycorrhizae plots only (Polygons C, F, and J). *P* values are results of single factor ANOVA.

a. All plots	**n**	**Alive + Stressed**		**Alive Only**	
Tall	12	29.8	(27.8)	27.1	(27.7)
Short	12	28.5	(23.8)	25.3	(22.3)
Control	12	28.9	(24.6)	26.0	(23.6)
P value		0.992		0.984	

b. Non-mycorrhizae plots	**n**	**Alive + Stressed**		**Alive Only**	
Tall	3	71.7	(6.7)	69.3	(7.0)
Short	3	63.9	(7.8)	57.6	(7.7)
Control	3	66.5	(2.2)	61.7	(2.7)
P value		0.348		0.144	

c. Mycorrhizae plots	**n**	**Alive + Stressed**		**Alive Only**	
Tall	9	15.8	(13.1)	13.1	(12.4)
Short	9	16.7	(11.5)	14.5	(12.0)
Control	9	16.4	(11.1)	14.1	(11.1)
P value		0.988		0.965	

Table 2.11. Percent cover of alien and native grasses in sagebrush outplanting polygons. Results of line-intercept sampling, October 2004. *P* values are the results of single factor ANOVA. Superscripts indicate significant differences (Scheffé test, α <0.05); see footnotes for explanation.

Polygon	n	Percent Cover: Cheatgrass		Bluebunch wheatgrass		All perennial grasses		All grasses	
A	5	1.82	(3.0)	17.9	(6.2)	27.2	(5.3)	29.1	(6.5)
B	3	10.83†	(8.4)	19.5	(4.9)	32.0	(5.7)	32.6	(6.1)
C	3	19.80‡	(5.1)	21.2	(5.7)	27.7	(6.5)	42.4^{f}	(8.0)
D	3	2.73	(3.7)	23.9	(6.4)	35.4	(6.9)	38.4	(7.1)
F	3	1.50	(2.0)	20.7	(5.8)	27.7	(6.5)	32.7	(5.5)
J	3	0.07	(0.2)	16.3	(3.6)	32.8	(6.8)	29.9	(3.3)
K	3	1.60	(5.7)	17.4	(9.3)	29.4	(9.1)	31.0	(8.8)
LM	3	1.27	(4.2)	20.4	(7.1)	32.6	(6.9)	33.9	(6.8)
P value		< 0.0001		0.029		0.003		<0.0001	

† --Significantly different from all other polygons except C.
‡ --Significantly different from all other polygons except B.
f--Significantly different from all other polygons except D and LM.

Table 2.12. Mean percent cover of alien and native grasses in sagebrush outplanting polygons by treatment. Results of line-intercept sampling, October 2004. *P* values are the results of single factor ANOVA. Values within columns with the same superscripts are significantly different (Tukey test, α < 0.05)

Polygon	n	Percent Cover: Cheatgrass		Bluebunch wheatgrass		All perennial grasses		All grasses	
Bare root M+	12	6.0^{a}	(8.7)	20.5	(6.0)	30.9	(7.3)	35.8^{ab}	(7.8)
Bare root M-	6	6.2^{b}	(8.5)	18.5	(7.4)	30.7	(7.6)	31.8^{a}	(7.5)
Tublings	8	1.6^{ab}	(3.5)	18.9	(6.5)	29.3	(6.4)	30.9^{b}	(6.9)
P value		0.008		0.268		0.504		0.003	

Table 2.13. Correlations between Wyoming big sagebrush outplanting survival 2002-2004 and environmental, community, and treatment variables. The symbol ± indicates presence-absence variables. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b - $P < 0.05$.

	Survival (%)	
	Alive + Stressed	**Alive only**
Elevation	-0.23	-0.20
Slope	-0.45[b]	-0.45[b]
Aspect	-0.09	-0.10
Heat load index	0.33	0.31
Condition (±: BR = 1; tubling = 0)	-0.20	-0.16
Hydrogel (±)	-0.20	-0.16
Mycorrhizae (±)	-0.81[a]	-0.78[a]
Sagebrush planted density	0.12	0.07
Percent cover		
Cheatgrass	0.00	0.03
Bluebunch wheatgrass	-0.27	-0.30
All perennial bunchgrasses	-0.08	-0.11
All grasses	-0.31	-0.31

Aerial seeding density. Aerial seeding densities exceeded the specified density (538.2 seeds/m^2) in all polygons (Table 2.14). Seeding densities of all species ranged from 1420.0 seeds/m^2 (± 1307.1 SD) in Polygon 4 to as high as 3577.8 seeds/m^2 (± 2932.0 SD) in Polygon 2. A significant difference among treatments (P = 0.065) was driven primarily by densities of Sandberg's bluegrass (*Poa sandbergii* = *P. secunda*) in Polygon 2 that were 50 – 150% higher than other treatments (P = 0.065). There were no significant differences in density between polygons for *Agropyron* species (P = 0.331) or for Indian ricegrass (*Oryzopsis hymenoides* = *Achnatherum hymenoides*; P = 0.252). Yarrow (*Achillea millefolium*) was a minor component of all seed mixes. Yarrow densities in most polygons were statistically similar, with the exception of very low densities in the Polygon 3 low elevation mix.

Aerial seeding densities of Wyoming big sagebrush seeds (*Artemisia tridentata* ssp. *wyomingensis*) were less than 90 seeds/m^2 in three of four plots in Polygon SS (Table 2.15). Density in a single plot was considerably higher at 675.0 seeds/m2 (± 488.0 SD; P = 0.064) but multiple comparison tests did not identify this difference as significant.

Table 2.14. Seed densities of grasses in aerial seeding plots on the ALE Reserve, 2002-2003. Values with the same superscript within columns were found to be significantly different by a Scheffé test (α < 0.05) following single factor ANOVA.

		Seeds /m^2 ($\underline{+}$ SD)									
Polygon	**n**	**All Species**		***Agropyron* Spp.***		**Sandberg's bluegrass**		**ricegrass**		**yarrow**	
1	5	2365.0	(1946.2)	515.0	(494.5)	1595.0	(1399.3)	220.0	(147.3)	35.0	(65.2)
2	9	3577.8^{ab}	(2932.0)	663.9	(771.8)	2722.2^{ab}	(2316.3)	169.4	(211.3)	22.2	(31.7)
3-high	18	1505.6^{a}	(1466.1)	280.6	(276.3)	1109.7^{a}	(1301.8)	--	--	1.4	(5.9)
3-low	23	1472.8^{b}	(1580.2)	316.3	(318.9)	1121.7^{b}	(1247.5)	112.5	(190.4)	34.8^{a}	(64.3)
4	5	1420.0	(1307.1)	400.0	(555.9)	990.0	(834.5)	5.0	(11.2)	25.0	(43.3)
SS	11	1788.6	(1481.8)	370.5	(394.6)	1093.2	(807.9)	--	--	75.0^{a}	(43.3)
P		0.065		0.331		0.065		0.252		0.007	

* Agropyron spp. includes both bluebunch wheatgrass (*A.spicatum* = *Pseudoroegneria spicata*) and thickspike wheatgrass (*A. dasystachyum* = *Elymus lanceolatus*)

Table 2.15. Seed densities and seedling emergence of Wyoming big sagebrush in shrub seed (SS) treatment plots on the ALE Reserve, 2003. P values are results of single factor ANOVA. Between-plot differences were not significant (Scheffé α < 0.05). No recruitment of big sagebrush seedlings was recorded within the shrub seed treatment plots in 2004.

Plot	**Seed Density (Seeds /m^2 $\underline{+}$ SD)**		**Seedling Emergence (Seedlings/m^2 $\underline{+}$ SD)**	
SS7	62.5	(17.7)	3.0	(6.7)
SS13	83.3	(62.9)	7.0	(13.4)
B8	675.0	(488.0)	20.0	(29.8)
T7-1	58.3	(28.9)	12.0	(16.2)
P	0.0644		0.216	

Seedling emergence. The overall emergence rate for herbaceous seedlings in 2003 was 3.2% (± 2.5 SD). The high elevation treatment exhibited the highest overall emergence at 5.7 %. The emergence rate for bluebunch wheatgrass in this treatment was the highest for all grass species in all treatments at 10.4 %. Sandberg's bluegrass also exhibited its highest emergence in this treatment (Table 2.16a). The low elevation seed mix applied over sandy soils exhibited the lowest emergence rate of all treatments. No grasses exhibited as much as 1.0% emergence in this treatment. Yarrow exhibited a much higher emergence rate in all treatments than did grass species. Yarrow emergence at low elevation exceeded 50.0%, regardless of soil type.

The high elevation seed mix treatment exhibited the highest mean density of emerging seedlings of all species in 2003 (Table 2.17a). This density (84.1 seedlings/m^2 ± 70.9 SD) was significantly greater than seedling densities in the low elevation sandy soil treatment (23.4 seedlings/m^2 ± 13.0 SD) but did not differ significantly from mean density in control plots (50.2 seedlings/m^2 ± 36.3 SD).

The high elevation treatment exhibited the highest mean densities of emerging *Agropyron* species and Sandberg's bluegrass seedlings in 2003 (Table 2.17b,c). The mean density of bluebunch wheatgrass seedlings in this treatment (32.8 seedlings/m^2 ± 22.9 SD) was significantly greater than the density of naturally occurring bluebunch wheatgrass seedlings in control plots (9.8 seedlings/m^2 ± 6.0 SD) and was significantly greater than the density of emerging thickspike wheatgrass seedlings at low elevation on both silt-loam and sandy soils. Mean density of bluebunch wheatgrass in this treatment did not differ significantly from mean density of bluebunch wheatgrass in the drill seed treatment. Mean densities of emerging seedlings of Sandberg's bluegrass in the high elevation treatment (39.6 seedlings/m^2 ± 42.2 SD) were significantly greater than seedling densities in both the drill seed treatment and in the control, but did not differ statistically from either low elevation aerial seeding treatnment.

The highest mean densities of emerging yarrow seedlings in 2003 were recorded in control plots (39.3 seedlings/m^2 ± 37.6 SD). However, this figure did not differ statistically from densities in other treatments (Table 2.17d).

Emergence of Wyoming big sagebrush seedlings varied between 3.0 seedlings/m^2 (± 6.7 SD) to 20.0 seedlings / m^2 (± 29.8 SD; Table 2.15) in 2003. Differences between plots were not significant ($P = 0.216$). The emergence rate of big sagebrush seedlings in 2003 ranged from 3.0 % to 20.6 %. The lowest emergence rate occurred in BRMaP 8 (Transect 1) where both seed density and absolute emergence were greatest. This plot was the most densely weedy of the shrub seed sample plots, with high densities of cheatgrass and alien annual forbs.

No surviving or newly-emerged big sagebrush seedlings were recorded within the four shrub seeding treatment plots in 2004. Overall, seedling searches were conducted in 47 plots within all rehabilitation project areas as well as in 9 control plots in both 2003 and 2004. Out of the 560 microplots closely examined for seedlings in these surveys, only one big sagebrush seedling was observed outside of the shrub seed polygon in 2003 (plot BRMaP 24-1). In 2004 a total of six new big sagebrush seedlings were recorded in plots outside of the shrub seed polygon. All three plots where these seedlings were found were in Polygon 3 (BRMaP 24-1 [1 seedling], P3-2 [2 seedlings], P3-3 [3 seedlings]). The seedling recorded in 2003 in BRMaP 24-1 was not found in 2004, although another seedling was found in a nearby microplot.

Table 2.16. Performance rates of seeds of selected species in aerial seeding plots, ALE Reserve, 2003 – 2004: (a) emergence rate (average emerged seedlings 2003 /m^2 ÷ average seeds/m^2); (b) potential recruitment rate (average seedlings/m^2 2004 ÷ average seeds/m^2).

a.		**Emergence Rate (%)**			
Treatment	**Soil Type**	**All Seeds**	**Agropyron Spp.**	**Sandberg's bluegrass**	**yarrow**
High elevation	Silt loam	5.7	10.4	3.5	33.9
Low elevation	Silt loam	3.1	3.5	1.5	69.3
Low elevation	Sand	0.7	0.6	0.5	52.7
Overall		3.2 (± 2.5)	4.8 (± 5.0)	1.8 (± 1.6)	52.0 (± 17.7)

b.		**Potential Recruitment Rate (%)**			
Treatment	**Soil Type**	**All Seeds**	**Agropyron Spp.**	**Sandberg's bluegrass**	**yarrow**
High elevation	Silt loam	1.2	0.6	1.2	0.9
Low elevation	Silt loam	0.6	0.1	0.8	1.9
Low elevation	Sand	0.3	0.4	0.2	3.6
Overall		0.7 (± 0.4)	0.4 (± 0.2)	0.8 (± 0.5)	2.1 (± 1.4)

Table 2.17. Seedling emergence (2003) and potential recruitment (2004) densities in study plots within the ALE Reserve rehabilitation project area. Abundance units are seedlings/ m^2. ANOVA P values are results of single factor ANOVA. Values with the same superscript within columns are significantly different (Tukey, $\alpha < 0.05$); t test P values are the results of two-tailed paired-sample t-tests.

a. All Seedlings					
Seed Mix	**Soil Type**	**n**	**Emergence 2003**	**Potential Recruitment 2004**	***t* test *P***
High elevation	Silt loam	12	84.1[a] (70.9)	17.4 (12.6)	0.007
Low elevation	Silt loam	17	51.6 (42.6)	9.8 (9.0)	< 0.001
Low elevation	Sand	9	23.4[a] (13.0)	10.2 (13.8)	0.080
Drill seed	Sand/Sandy loam	9	38.6 (52.6)	33.2 (60.5)	0.256
Control	Silt loam	9	50.2 (36.3)	7.3 (6.3)	0.005
ANOVA *P*			0.071	0.191	

Continued

Table 2.17 (continued). Seedling emergence (2003) and potential recruitment (2004) densities in study plots within the ALE Reserve rehabilitation area. Abundance units are seedlings/ m^2. ANOVA *P* values are results of single factor ANOVA. Values with the same superscript within columns are significantly different (Tukey, $\alpha < 0.05$); *t* test *P* values are the results of two-tailed paired-sample *t*-tests.

b. *Agropyron* Spp.*			**Emergence 2003**		**Potential Recruitment 2004**		***t* test *P***
Seed Mix	**Soil Type**	**n**					
High elevation	Silt loam	12	32.8^{abc}	(22.9)	1.8	(1.5)	< 0.001
Low elevation	Silt loam	17	13.0^{a}	(16.5)	0.5	(0.8)	0.006
Low elevation	Sand	9	3.1^{b}	(2.3)	2.1	(3.0)	0.376
Drill seed	Sand/Sandy loam	9	21.8	(22.3)	1.8	(2.6)	0.024
Control	Silt loam	9	9.8^{c}	(6.0)	1.9	(2.1)	0.003
ANOVA *P*			0.002		0.194		

c. Sandberg's bluegrass			**Emergence 2003**		**Potential Recruitment 2004**		***t* test *P***
Seed Mix	**Soil Type**	**n**					
High elevation	Silt loam	12	39.6^{ab}	(42.2)	13.7	(11.6)	0.068
Low elevation	Silt loam	17	16.6	(23.7)	8.6	(9.1)	0.158
Low elevation	Sand	9	11.6	(6.8)	6.0	(9.3)	0.178
Drill seed	Sand/Sandy loam	9	2.7^{a}	(3.2)	32.2	(60.5)	0.170
Control	Silt loam	9	0.9^{b}	(1.5)	4.6	(4.5)	0.17
ANOVA *P*			0.003		0.138		

d. Yarrow			**Emergence 2003**		**Potential Recruitment 2004**		***t* test *P***
Seed Mix	**Soil Type**	**n**					
High elevation	Silt loam	12	11.8	(8.5)	0.3	(0.9)	< 0.001
Low elevation	Silt loam	17	22.1	(40.0)	0.6	(1.2)	0..040
Low elevation	Sand	9	8.8	(10.5)	0.6	(0.7)	0.044
Drill seed	Sand/Sandy loam	9	14.1	(29.7)	0.4	(0.7)	0.202
Control	Silt loam	9	39.3	(37.6)	0.9	(1.5)	0.013
ANOVA *P*			0.194		0.815		

* *Agropyron* spp. includes both bluebunch wheatgrass (*A.spicatum* = *Pseudoroegneria spicata*—high elevation and drill seed mixes only) and thickspike wheatgrass (*A. dasystachyum* = *Elymus lanceolatus*—low elevation mix only).

Potential seedling recruitment. Mean overall potential seedling recruitment in 2004 was 31.3% of 2003 emergence (Table 2.18). Mean potential recruitment was highest for Sandberg's bluegrass (73.9%). Overall numbers of seedlings were not significantly different between years for this species ($P = 0.410$). Percent recruitment was much lower for *Agropyron* species (7.9%) and for yarrow (22.1%). The number of seedlings of squirreltail, needle-and-thread, and Indian ricegrass recorded in either year were too few to yield meaningful statistics.

The overall rate of potential recruitment for herbaceous seedlings in this study was 0.7% (± 0.4 SD; Table 2.16b). The high elevation treatment provided the highest overall recruitment rate at 1.2%. Bluebunch wheatgrass exhibited its highest potential recruitment in this treatment, and the recruitment rate for Sandberg's bluegrass in this treatment was the highest for all grass species in all treatments, also 1.2%. The low elevation seed mix applied over sandy soils exhibited the lowest recruitment rate of all treatments, even though yarrow exhibited its highest absolute recruitment in this treatment. *Agropyron* species, in this case thickspike wheatgrass, exhibited the lowest rate of recruitment in the study (0.1%) in the low elevation seed mix on silt-loam soils.

Differences between potential recruitment densities by treatment type in 2004 were not significant (Table 2.17). Sandberg's bluegrass exhibited the highest potential recruitment densities of all seeded species (Table 2.17c). This was in large part due to extremely high emergence in spring 2004 (or fall 2003) in the drill seed strip, where densities in 2004 exceeded those of the previous year by more than 12-fold. Even within the drill seed plots, the magnitude of this difference was primarily driven by densities in the three northernmost plots. In these plots, which each have a substantial component of established mature Sandberg's bluegrass, the magnitude of increase from 2003 to 2004 was almost 30-fold.

Overall recruitment densities were largely a function of the high densities of Sandberg's bluegrass in all treatments relative to other seeded species (Table 2.17a, c). Potential recruitment of *Agropyron* species was lowest in the low elevation/ silt-loam treatment (0.5 seedlings/m^2) but was between 1.8 seedlings/m^2 and 2.1 seedlings/m^2 in all other treatments (Table 2.17b). Yarrow recruitment was highest in the control, but was less than 1.0 seedlings/m^2 in all treatments (Table 2.17d).

Correlations between environmental and community variables and seedling emergence and recruitment were moderate at best and were generally weak (Table 2.19). The strongest negative correlation was between density of *Agropyron* seedlings (potential recruitment) and total cover of all vascular plant species in 2004 ($r = -0.47$; $P < 0.005$). Percent cover and density of cheatgrass and percent cover of litter (2003) were also moderately negatively correlated with Agropyron density in 2004. Elevation and percent cover of both native perennial grasses and all native perennial species were moderately correlated with the density of yarrow seedlings in 2003 but not in 2004. Density of Sandberg's bluegrass seedlings exhibited no correlations greater than $r = -0.31$ during either 2003 or 2004.

New seedlings provided little estimated cover to plant communities in 2003 and 2004 (Table 2.20a). The mean percent cover of all seeded species in 2003 was 0.9% (± 0.5 SD). Only one plot exhibited greater than 2.0% cover of seeded species and fewer than half of all plots in the project area had greater than 1.0% cover that year. In 2004 cover declined to 0.4% (± 0.3 SD) as numbers of seedlings from the 2003 cohort suffered mortality. Only two of 35 plots in polygons 1through 4 exhibited cover of seeded species greater than 1.0% in 2004, and more than 70% of plots had less than 0.5% cover. These cover values are likely to be exaggerated because our sampling methodology by convention recorded trace quantities as a full 1.0% in sample microplots, when actual cover was often a fraction of that figure.

The mean percent frequency of seeded species was 68.9% (± 23.9 SD) in 2003, declining to 34.1% (± 18.0 SD) in 2004 (Table 2.20b). Most of this value in 2004 was due to Sandberg's bluegrass. No other species accounted for more than 8.5% frequency in 2004.

Table 2.18. Potential recruitment (2004) as a percentage of seedling emergence densities (2003) in study plots within the ALE Reserve rehabilitation area. All sites (n = 47). Abundance units in 2003 and 2004 are seedlings/ m^2. *P* values are the results of two-tailed paired-sample *t* tests.

Species or Species Group	**Emergence 2003**		**Potential Recruitment 2004**		**Potential Recruitment (%)**	***P***
All seedlings	52.0	(53.0)	16.3	(28.5)	31.3	< 0.0001
*Agropyron** spp.	17.8	(20.5)	1.4	(2.0)	7.9	< 0.0001
Sandberg's bluegrass	18.8	(28.4)	13.9	(28.3)	73.9	0.410
Yarrow	15.4	(27.9)	3.4	(27.6)	22.1	0.005
Squirreltail	0.7	(2.9)	0.0	(0.0)	---	0.095
needle-and-thread	0.04	(0.2)	0.5	(2.0)	---	0.096
ricegrass	0.0	(0.0)	0.06	(0.4)	---	0.323

* *Agropyron* spp. includes both bluebunch wheatgrass (*A.spicatum* = *Pseudoroegneria spicata*) and thickspike wheatgrass (*A. dasystachyum* = *Elymus lanceolatus*).

Table 2.19. Correlations of seedling emergence and recruitment density with environmental and community variables, ALE Reserve, 2003-2004. All variables are from 2003-2004 only. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; c - $P = 0.005$; d - $P = 0.01$; e - $P = 0.05$.

	All Seedlings		***Agropyron* spp.***		**Sandberg's bluegrass**		**Yarrow**	
	2003	2004	2003	2004	2003	2004	2003	2004
Elevation	0.37^e	0.09	0.24	-0.14	-0.07	0.12	0.61^a	0.14
Slope	0.18	-0.01	0.37^e	0.00	0.06	-0.01	0.01	-0.15
Aspect	-0.25	-0.26	-0.22	-0.22	-0.10	-0.20	-0.21	-0.01
Heat Load	-0.26	-0.08	-0.43^d	-0.05	-0.11	-0.06	-0.07	0.14
Litter % cover 2003	-0.17	-0.08	-0.15	-0.40^e	0.09	-0.02	-0.32	0.14
Litter % cover 2004	-0.09	-0.16	-0.06	-0.23	-0.13	-0.16	0.02	0.16
Cheatgrass % cover 2003	-0.35^e	-0.38^e	-0.32	-0.42^d	-0.25	-0.31	-0.16	0.07
Cheatgrass % cover 2004	-0.38^e	-0.22	-0.46^c	-0.42^d	-0.07	-0.16	-0.31	0.22
Cheatgrass density 2003	-0.27	-0.28	-0.30	-0.42^d	-0.13	-0.19	-0.15	0.00
Cheatgrass density 2004	-0.25	-0.16	-0.26	-0.37^e	-0.06	-0.15	-0.22	0.37^e
Total % cover, all aliens 2003	-0.31	-0.27	-0.20	-0.31	-0.20	-0.24	-0.23	0.04
Total % cover, all aliens 2004	-0.30	-0.14	-0.41^e	-0.37^e	0.03	-0.09	-0.31	0.18
Perennial grass % Cover 2003	0.29	0.04	0.16	-0.19	-0.02	0.08	0.46^c	0.15
Perennial grass % Cover 2004	0.30	0.09	0.15	-0.20	0.01	0.13	0.46^c	0.13
Total % perennial plant cover 2003	0.30	0.11	0.29	-0.06	-0.06	0.14	0.42^e	0.04
Total % perennial plant cover 2004	0.42^d	0.17	0.34^e	-0.13	0.08	0.21	0.46^c	0.00
Total % plant cover 2003	0.08	-0.06	0.25	-0.30	-0.12	-0.05	0.11	0.22
Total % plant cover 2004	-0.02	-0.04	-0.17	-0.47^e	0.07	0.04	0.01	0.22

* Agropyron spp. includes both bluebunch wheatgrass (*A.spicatum* = *Pseudoroegneria spicata*) and thickspike wheatgrass (*A. dasystachyum* = *Elymus lanceolatus*).

Table 2.20. Mean percent cover (a) and frequency (b) of seedlings in aerial seeding polygons 1, 2, 3, and 4 on the ALE Reserve, 2003 – 2004. n = 35.

a.	**Percent Cover (± SD)**	
	2003	2004
Agropyron spp.*	0.4 (0.3)	0.1 (0.1)
Sandberg's bluegrass	0.3 (0.2)	0.2 (0.1)
Yarrow	0.3 (0.3)	0.04 (0.1)
All	0.9 (0.5)	0.4 (0.3)

b.	**Percent Frequency (± SD)**	
	2003	2004
Agropyron spp.*	33.6 (26.5)	8.4 (6.5)
Sandberg's bluegrass	38.4 (20.1)	29.1 (17.5)
Yarrow	33.2 (20.4)	2.3 (3.0)
All	68.9 (23.1)	34.1 (18.0)

* Agropyron spp. includes both bluebunch wheatgrass (*A.spicatum* = *Pseudoroegneria spicata*) and thickspike wheatgrass (*A. dasystachyum* = *Elymus lanceolatus*).

Drill-seed treatment. In the drill seed treatment Anatone bluebunch wheatgrass exhibited greater seedling emergence (43.0 seedlings/m^2 ± 29.8 SD) in 2003 than either Secar or the Hells Canyon bluebunch wheatgrass, although differences were not significant (P = 0.124; Table 2.21). Seedlings from the Hell's Canyon collection exhibited the greatest potential recruitment (3.7 seedlings/ m^2 ± 3.5 SD) in 2004, however, though again differences between the seed stocks were not significant (P = 0.304; Table 2.21).

Potential recruitment rates of wheatgrass seedlings in drill seed plots were extremely variable, ranging from zero to 50.0%. The Hell's Canyon collection exhibited the highest recruitment at 25.0% (± 25.0 SD). Neither of the other seed stocks exhibited a recruitment rate greater than 2.0%. Differences between treatments were not significant (P = 0.163; Table 2.21).

Seedling densities of Sandberg's bluegrass in three plots north of Milepost 8 in 2004 were nearly 30 time greater than emergence densities in 2003. No comparable phenomena were observed in any other plots or treatments.

The most meaningful correlations between environmental and community variables and seedling density in the drill seed strip appeared to be with percent cover of alien forbs and with percent cover of both native perennial grasses and all native perennials (Table 2.22). Percent cover of alien forbs was strongly correlated with wheatgrass seedling emergence density in 2003 (r = 0.85; P < 0.005). This variable was also strongly correlated with seedling densities of yarrow in 2003 and of Sandberg's bluegrass in 2004. Percent cover of native perennial grasses and percent cover of all native perennials in both 2003 and 2004 were moderately to strongly negatively correlated with recruitment densities of both wheatgrass (r = - 0.68 to – 0.74; P < 0.05) and yarrow (r = - 0.60 to – 0.72) in 2004. There were no significant correlations between wheatgrass seed stocks and environmental variables, reflecting the relative similarity of these parameters throughout this polygon.

As in the aerial seeding polygons, new seedlings provided little estimated cover in sample plots within this polygon in 2003 and 2004, although values were somewhat larger than in aerially-seeded plots (Table 2.23a). The mean percent cover of all seeded species in 2003 was 1.4% (± 0.9 SD). In 2004 cover declined to 0.6% (± 0.5 SD) as numbers of seedlings from the 2003 cohort suffered mortality. None of the bluebunch wheatgrass stocks exhibited as much as 1.0% cover during either 2003 or 2004. Again, these cover values are likely to be exaggerated because our sampling methodology by convention recorded trace quantities as a full 1.0% in sample microplots, when actual cover was often a fraction of that figure. Here and there small patches within the drill seed polygon appeared to exhibit greater recruitment than was observed in plot surveys. These patches were never larger than a few square meters, however, and were limited to areas of recently disturbed soil where competition was reduced.

The mean percent frequency of seeded species was 66.7% (± 63.0 SD) in 2003, declining to 51.1% (± 46.8 SD) in 2004 (Table 2.23b). Most of this 2004 frequency value was due to Sandberg's bluegrass. No other species accounted for more as much as 10.0% frequency in 2004. Percent frequency of the bluebunch wheatgrass stocks ranged from 38.3% to 65.0% in 2003, but only between 5.0% and 18.3% in 2004.

Vegetation of the drill seed strip along the SR 240 highway corridor varied from areas dominated by invasive annuals to areas dominated or codominated by native perennials. Shrubs are uncommon along the entire length of this polygon and small quantities of longleaf phlox (*Phlox longifolia*) along three transects represented the only woody vegetation recorded during this study. Percent cover of non-native species, all annuals, in sample plots varied from 13.0% to 60.6% in 2004. Cheatgrass was present at all sites, with percent cover ranging from 7.3% to 34.3%. Areas with substantial infestations of invasive perennial plant species occur along this corridor but were not sampled in this study (see Evans et al. 2003). Percent cover of native perennials ranged from as low as 1.3% up to 24.8%. Areas notable for dominance or codominance of native species occurred between mileposts six and nine, and in the area encompassing mileposts 18 to 19. Perennial vegetation in these areas was dominated by bunchgrasses. Sandberg's bluegrass was the most common and abundant perennial, while needle-and-thread was abundant in several areas. Yarrow and hoary aster (*Macharaenthera canescens*) were the most common native perennial forbs; Carey's balsamroot (*Balsamorhiza careyana*) and sulfur lupine (*Lupinus suphureus*) were locally abundant. In the better areas the sandy to sandy loam soils of the Cold Creek Valley support native plant species adapted to these soil types which are uncommon or entirely absent elsewhere on the Reserve (Table 2.24).

Table 2.21. Emergence (2003) and potential recruitment (2004) densities of wheatgrass seedlings for different seed stocks in drill seed treatments, ALE Reserve. Abundance units in 2003 and 2004 are seedlings/ m^2. ANOVA *P* values are results of single factor ANOVA. *t*-test *P* values are the results of two-tailed paired-sample *t* tests between 2003 and 2004 densities.

Seed stock	**n**	**Seedling density (seedlings / m^2 ± SD)**		**Potential Recruitment (%)**	***t* test *P***
		Emergence 2003	**Potential Recruitment 2004**		
Anatone	3	43.0 (29.8)	1.3 (2.3)	1.8 (3.0)	0.121
Hell's Canyon	3	11.0 (7.0)	3.7 (3.5)	25.0 (25.0)	0.106
Secar	3	12.3 (6.0)	0.3 (0.6)	2.0 (3.4)	0.076
ANOVA *P*		0.124	0.304	0.163	

Table 2.22. Correlations of seedling emergence (2003) and potential recruitment (2004) densities with environmental and community variables in drillseeded treatment plots on the ALE Reserve. Values accompanied by the following superscripts are significant: a - $P < 0.0001$; b – $P < 0.001$; c – $P < 0.005$; d – $P < 0.01$; e – $P < 0.05$.

		Seedling density (seedlings/ m^2)					
		Wheatgrass seedlings		**Sandberg's bluegrass**		**Yarrow**	
		2003	2004	2003	2004	2003	2004
Elevation		0.64	-0.11	0.04	0.63	0.52	-0.04
Slope		-0.40	-0.49	-0.45	-0.23	-0.20	-0.43
Aspect		-0.52	-0.65	-0.63	-0.23	-0.23	-0.58
Heat Load Index		-0.40	-0.49	-0.46	-0.23	-0.21	-0.44
Litter % cover	2003	-0.34	0.43	0.31	-0.49	-0.42	0.10
	2004	0.01	-0.32	-0.22	-0.03	-0.10	-0.68^{e}
Cheatgrass % cover	2003	-0.40	-0.49	-0.44	-0.50	-0.51	-0.42
	2004	-0.19	0.15	0.15	-0.27	-0.20	0.25
Cheatgrass density	2003	-0.33	-0.61	-0.54	-0.33	-0.35	-0.56
	2004	-0.29	-0.26	-0.24	-0.34	-0.30	0.00
Alien forbs % cover	2003	0.85^{c}	-0.11	0.23	0.84^{c}	0.79^{e}	-0.05
	2004	-0.22	0.40	0.18	-0.47	-0.44	0.57
Native perennial grass	2003	-0.34	-0.71^{e}	-0.66	-0.11	-0.16	-0.60
% cover	2004	-0.05	-0.68^{e}	-0.62	0.02	-0.09	-0.61
All native perennials	2003	-0.36	-0.72^{e}	-0.71^{e}	-0.16	-0.22	-0.72^{e}
% cover	2004	-0.23	-0.74^{e}	-0.73^{e}	-0.12	-0.21	-0.66
Total % cover	2003	0.18	-0.66	-0.39	0.23	0.17	-0.60
	2004	-0.15	0.07	-0.04	-0.34	-0.32	0.41
Sandberg's bluegrass	2003	-0.22	-0.60	-0.56	-0.11	-0.14	-0.54
% cover	2004	0.06	-0.60	-0.54	0.06	-0.05	-0.53
Wheatgrass % cover	2003	0.55	0.20	0.44	0.38	0.44	0.59
	2004	-0.01	0.94^{b}	0.73^{b}	-0.11	-0.06	0.40

Table 2.23. Mean percent cover (a) and frequency (b) of seedlings in the drill seed treatment, ALE Reserve 2003 – 2004.

a.	n	**Percent Cover (± SD)** 2003	2004
Bluebunch wheatgrass (All)	9	0.7 (0.5)	0.2 (0.3)
Anatone	3	0.9 (0.3)	0.1 (0.1)
Hells Canyon	3	0.5 (0.4)	0.4 (0.4)
Secar	3	0.8 (0.8)	0.2 (0.2)
Sandberg's bluegrass	9	0.1 (0.2)	0.4 (0.4)
Yarrow	9	0.4 (0.5)	0.03 (0.0)
All	9	1.4 (0.9)	0.6 (0.5)

b.		**Percent Frequency (± SD)** 2003	2004
Bluebunch wheatgrass (All)	9	47.8 (43.1)	9.4 (8.8)
Anatone	3	65.0 (22.9)	5.0 (8.7)
Hells Canyon	3	38.3 (25.7)	18.3 (16.1)
Secar	3	40.0 (22.9)	5.0 (5.0)
Sandberg's bluegrass	9	13.9 (12.7)	36.7 (30.7)
Yarrow	9	30.0 (23.3)	3.9 (3.8)
All	9	66.7 (63.0)	51.1 (46.8)

Table 2.24. Native plant species found on sandy soils in the Cold Creek Valley drill seed strip that are uncommon or absent on other soil types on the ALE Reserve. These species were noted primarily in areas of native plant dominance or where native species codominate with alien annuals. Scientific names follow Hitchcock and Cronquist (1973).

Annuals	**Scientific name**
Bailey's buckwheat	*Eriogonum baileyi*
shy gilia	*Gilia sinuata*
low lupine	*Lupinus pusillus*
whitestem stickleaf	*Mentzelia albicaulis*
Perennials	**Scientific name**
white sand verbena	*Abronia mellifera*
stalked-pod milkvetch	*Astragalus sclerocarpus*
turpentine springparsley	*Cymopterus terebinthinus*
pale evening primrose	*Oenothera pallida*
Indian ricegrass	*Oryzopsis hymenoides*
sand beardtongue	*Penstemon acuminatus*
dune scurfpea	*Psoralea lanceolata*

Cheatgrass abundance and effects of herbicide treatments. Mean percent cover of cheatgrass in 17 long-term permanent plots within the BAER project area increased by just under 1.0% in the year following herbicide applications made during the fall and winter of 2002-2003. This increase was not statistically significant ($P = 0.492$; Table 2.25a). In contrast, mean percent cover of cheatgrass in 53 untreated long-term plots throughout the ALE Reserve exhibited a significant increase of more than 2.0% during the same period ($P = 0.009$). Mean density of cheatgrass in 20 long-term permanent density plots (vegetation plots plus transition density plots) within the project area was unchanged between 2002 and 2003 ($P = 0.966$; Table 2.25b).

Plots located at elevations of 800' (244 m) and above exhibited considerably lower values for both cover and density compared to plots located below 800'. These differences became even more pronounced as cheatgrass abundance at lower elevations increased disproportionately between 2003 and 2004 (Table 2.25).

Both cover and density of cheatgrass increased within the project area between 2003 and 2004, with the highest values of both parameters in both years occurring in Polygon 1. Mean percent cover of cheatgrass increased from 14.8% (± 12.0 SD) in 2003 to 16.7% (± 12.7 SD) in 2004 ($P = 0.068$; Table 2.26a). Cover in Polygon 1 was significantly greater than both cover in Polygon 3 and overall mean cover in 2003, and was significantly greater than values from all polygons except Polygon 4 in 2004. Mean overall density of cheatgrass increased from 349.8 stems/ m^2 (± 373.2 SD) in 2003 to 653.0 stems/ m^2 (± 669.7 SD; $P < 0.0001$; Table 2.26b). Density in Polygon 1 was significantly greater than density in all other polygons or polygon groups except Polygon 4 in 2003, and was significantly greater than density in all other polygons or polygon groups in 2004.

Cover and density of cheatgrass within the project area increased between 2002 and 2003 in both long-term permanent plots and in recently established rehabilitation plots (Table 2.27). Percnt cover of cheatgrass in permanent plots increased from 8.7% (± 9.7SD) to 11.6% (± 12.6 SD; $P = 0.040$) while cover in rehabilitation plots increased from 19.4% (± 13.0 SD) to 21.8% (± 12.7 SD; $P = 0.147$).

Both cover and density were substantially greater in the randomly-located rehabilitation plots than in the long-term permanent plots located based on habitat quality criteria (Table 2.27). Cheatgrass cover was more than 10.0% greater in rehabilitation plots during both 2003 and 2004, and cheatgrass density in rehabilitation plots exceeded values in long-term plots by more than 350 stems/ m^2 during the same two years.

Percent cover of Sandberg's bluegrass (*Poa sandbergii*) in permanent plots within the project area was statistically unchanged following herbicide treatments in the fall and winter of 2002-2003 (Table 2.28). Cover of Sandberg's bluegrass decreased between 2002 and 2003 in untreated plots throughout the ALE Reserve. Percent frequency of Sandberg's bluegrass increased within the project area in 2003 ($P < 0.001$) as a result of seedling emergence following aerial seeding operations (see below).

Table 2.25. Percent cover (a) and density (b) of cheatgrass (*Bromus tectorum*) in treated ('project area') and untreated areas before and after glyphosate applications made in fall and winter 2002-2003. Data from long-term permanent plots. ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are the results of two-tailed paired-sample *t* tests between 2002 and 2003 values only.

a. Percent cover†	n	2002 (± SD)	2003 (± SD)	2004 (± SD)	ANOVA *P* value	*t* test *P* value
Project area						
All Plots	17	9.2 (11.7)	10.1 (9.9)	13.5 (12.8)	0.029	0.492
> 800 ft.	10	3.4 (4.8)	4.6 (5.5)	5.0 (4.6)	0.154	---
< 800 ft.	7	17.5 (13.9)	18.0 (9.6)	25.6 (10.7)	0.068	0.867
Untreated areas	53†	8.6 (10.9)	10.8 (11.4)	9.3 (10.8)	0.016	0.009

b. Density (stems/ m^2)‡	n	2002 (± SD)	2003 (± SD)	2004 (± SD)	ANOVA *P* value	*t* test *P* value
Project area						
All Plots	20	200.7 (329.0)	203.8 (341.1)	432.5 (580.5)	0.010	0.966
> 800 ft.	12	86.9 (163.4)	60.4 (69.7)	99.6 (128.4)	0.321	---
< 800 ft.	8	371.4 (443.2)	418.8 (469.2)	932.0 (643.1)	0.010	0.795
Untreated areas	67‡	307.0 (411.8)	267.3 (304.3)	301.4 (390.5)	0.586	---

† – Vegetation plots only.
‡ – Vegetation plots + transition density plots.

Table 2.26. Mean percent cover (a) and density (b) of cheatgrass (*Bromus tectorum*) within the BAER rehabilitation project area, ALE Reserve, 2003-2004. ANOVA *P* values are results of one-way repeated measures ANOVA; values with the same superscript within columns are significantly different (Tukey, $\alpha < 0.05$). *t* test *P* values are the results of two-tailed paired-sample *t* tests.

a. Percent cover	n	2003	(± SD)	2004	(± SD)	*t* test *P* value
All Plots	44	14.8^{a}	(12.0)	16.7^{ab}	(12.7)	0.068
Polygon 1	5	32.3^{ab}	(13.2)	41.1^{acde}	(2.7)	0.147
Polygon 2	8	16.4	(11.2)	19.0^{c}	(8.2)	0.195
Polygon 3	19	6.6^{bc}	(6.7)	7.6^{bd}	(7.7)	0.234
Polygon 4	3	15.8	(1.8)	18.8	(3.3)	0.206
Drill seed	9	20.7^{c}	(20.7)	19.4^{e}	(8.8)	0.635
ANOVA *P* value		< 0.001		< 0.0001		

b. Density (stems/ m^2)	n	2003	(± SD)	2004	(± SD)	*t* test *P* value
All Plots	44	349.8^{a}	(373.2)	653.0^{ab}	(669.7)	< 0.0001
Polygon 1	5	1083.1^{abcd}	(480.9)	2104.8^{acdef}	(453.8)	0.053
Polygon 2	8	310.6^{b}	(293.8)	645.3^{c}	(337.9)	0.003
Polygon 3	19	163.8^{c}	(169.7)	210.6^{bd}	(217.1)	0.125
Polygon 4	3	457.9	(122.7)	558.8^{e}	(77.4)	0.139
Drill seed	9	362.8^{d}	(214.7)	887.8^{f}	(467.1)	0.002
ANOVA *P* value		< 0.0001		< 0.0001		

Table 2.27. Comparison of (a) percent cover and (b) density of cheatgrass (*Bromus tectorum*) between non-randomly selected long-term permanent plots within the BAER project area and randomly selected rehabilitation plots (polygons 1-4 only) established in 2003, ALE Reserve. Between-year comparisons were made using two-tailed paired-sample *t* tests; within-year comparisons (between plot types) were made using independent two-sample *t* tests.

a. Percent cover	n	2003 (± SD)	2004 (± SD)	*P* value
Permanent plots	20	8.7 (9.7)	11.6 (12.6)	0.040
New plots	15	19.4 (13.0)	21.8 (12.7)	0.147
P value		0.008	0.024	

b. Density (stems/ m^2)	n	2003 (± SD)	2004 (± SD)	*P* value
Permanent plots	20	190.8 (346.4)	418.8 (588.1)	0.005
New plots	15	554.1 (397.1)	853.6 (791.7)	0.082
P value		0.007	0.071	

Table 2.28. Percent cover and frequency of Sandberg's bluegrass (*Poa sandbergii*) in treated and untreated areas before and after glyphosate applications made in fall and winter 2002-2003. Data from long-term permanent plots, ANOVA *P* values are results of one-way repeated measures ANOVA; *t* test *P* values are the results of two-tailed paired-sample *t* tests between 2002 and 2003 values only.

	n	2002 (± SD)	2003 (± SD)	2004 (± SD)	ANOVA *P* value	*t* test *P* value
Percent Cover						
Treated	17	9.0 (9.1)	8.2 (8.2)	8.9 (8.9)	0.546	---
Untreated	53	17.6 (7.8)	16.3 (6.6)	16.0 (6.8)	0.057	0.055
Percent Frequency						
Treated	12	39.6 (24.9)	64.2 (23.4)	51.7 (30.6)	< 0.001	< 0.001
Untreated	25	86.0 (14.1)	87.8 (14.4)	87.4 (87.4)	0.706	---

Recovery of bulldozer fireline. Significant differences in both alien and native vegetation were clearly evident between the fire suppression swath in upper Snively Basin and the surrounding grassland. In both 2003 and 2004 all measures of cheatgrass abundance were significantly higher within the suppression swath compared to the less disturbed (burned but not bulldozed) adjacent grassland. Percent cover of cheatgrass in 2004 (22.8% ± 15.1 SD) was nearly 10-fold that of the surrounding area (2.4% ± 4.2 SD; $P < 0.0001$; Table 2.29). Cheatgrass density within the suppression swath in 2004 (691.7 stems/m^2 ± 495.7 SD) was more than 10 times that of the adjacent area (59.7 stems/m^2 ± 148.6 SD; $P < 0.0001$), and percent frequency (100% ± 0.0 SD) was more than double that of the adjacent area (44.4% ± 34.7 SD; $P = 0.109$) in 2004.

Percent cover of native perennial bunchgrasses in 2004 was significantly less in the suppression swath (17.3% ± 12.9 SD) compared to the surrounding area (31.8% ± 17.4 SD; $P = 0.010$). Percent cover of Sandberg's bluegrass, which was seeded into the suppression swath as part of rehabilitation efforts in 2001, was still lower in the suppression swath in 2004 (7.7% ± 6.0 SD) than in surrounding vegetation (10.8% ± 5.4 SD; $P = 0.139$; Table 2.29).

While only small amounts of microbiotic crust were found in the surrounding grassland, microbiotic crust was entirely absent within the suppression swath (Table 2.29).

In contrast to the results from 2003 surveys where cover of bare ground was statistically greater in the suppression swath than in surrounding area ($P = 0.047$), in 2004 bare ground was not statistically different between the two sets of plots (Table 2.29). Percent cover of bare ground in the suppression swath had not changed significantly between 2003 and 2004, but the surrounding area showed an increase in bare ground from 8.8% (± 6.5 SD) to 24.8% (± 20.3 SD) between years ($P = 0.005$; Table 2.30).

Owing to the high abundance of alien annuals, total vegetation (i.e. total cover of all vascular plants) was higher in the suppression swath (68.7% ± 13.7 SD) than in the surrounding area (59.3% ± 20.9 SD; $P = 0.097$; Table 2.29). However, total native perennial vegetation was significantly lower in the suppression swath (43.4% ± 16.0 SD) compared to the surrounding vegetation (55.6% ± 23.0 SD; $P = 0.035$).

Comment: I made this 2 sentences so that we would not have a one sentence paragraph.

Within the suppression swath, the only community variable that changed significantly between years was percent cover of litter, which increased from 14.3% (± 7.4 SD) in 2003 to 27.1% (± 11.3 SD; $P < 0.001$) in 2004 (Table 2.30a). Estimated litter also increased in the surrounding area between years. Within the surrounding area, significantly less total vegetation was recorded in 2004 (59.3% ± 20.9 SD) compared to 2003 (74.1% ± 10.1 SD; $P = 0.012$). Native components of total vegetation, including percent cover of native bunchgrasses and total percent cover of native perennials also were reduced compared to the previous year, coincident with a significant increase in bare ground between years in the surrounding grassland (Table 2.30b).

Table 2.29. Comparison of community variables between the fire-suppression swath and surrounding vegetation in upper Snively Basin, ALE Reserve, 2004. *P* values are results of paired-sample *t* tests; n = 18 for each variable except percent frequency of cheatgrass (n=3). MBC = microbiotic crust.

		Mean (± SD)				
		Suppression Swath		Surrounding Vegetation		*P* value
Cheatgrass	% Cover	22.8	(15.1)	2.4	(4.2)	<0.0001
	% Frequency	100.0	(0.0)	44.4	(34.7)	0.109
	Density (stems/m^2)	691.7	(495.7)	59.7	(148.6)	<0.0001
Percent Cover	MBC	0.0	(0.0)	3.3	(6.5)	0.047
	Native Bunchgrasses	17.3	(12.9)	31.8	(17.4)	0.010
	Sandberg's Bluegrass	7.7	(6.0)	10.8	(5.4)	0.139
	Total Native Perennial Vegetation	43.4	(16.0)	55.6	(23.0)	0.035
	Total vegetation	68.7	(13.7)	59.3	(20.9)	0.097
	Litter	27.1	(11.3)	23.2	(12.6)	0.384
	Bare Ground	20.3	(16.7)	24.8	(20.3)	0.544

Table 2.30. Between-year comparisons of community variables between (a) the fire-suppression swath and (b) surrounding vegetation in Upper Snively Basin, ALE Reserve, 2003 - 2004. *P* values are results of independent sample *t* tests; n = 18 for each variable except percent frequency of cheatgrass (n=3). MBC = microbiotic crust.

a. Suppression Swath		Mean (± SD) 2003		2004		*P* value
Cheatgrass	% Cover	23.3	(21.9)	22.8	(15.1)	0.944
	% Frequency	94.4	(9.6)	100.0	(0.0)	0.423
	Density (stems/m^2)	483.3	(438.0)	691.7	(495.7)	0.190
Percent Cover	MBC	0.0	(0.0)	0.0	(0.0)	---
	Native Bunchgrasses	12.6	(14.7)	17.3	(12.9)	0.307
	Sandberg's Bluegrass	6.3	(6.1)	7.7	(6.0)	0.516
	Total Native Perennial Vegetation	43.1	(21.7)	43.4	(16.0)	0.952
	Total vegetation	76.7	(25.8)	68.7	(13.7)	0.255
	Litter	14.3	(7.4)	27.1	(11.3)	<0.001
	Bare Ground	19.3	(21.8)	20.3	(16.7)	0.878

b. Surrounding Vegetation		2003		2004		*P* value
Cheatgrass	% Cover	1.4	(2.1)	2.4	(4.2)	0.375
	% Frequency	50.0	(33.3)	44.4	(34.7)	0.851
	Density (stems/m^2)	30.6	(57.2)	59.7	(148.6)	0.445
Percent Cover	MBC	6.2	(7.3)	3.3	(6.5)	0.211
	Native Bunchgrasses	40.2	(11.3)	31.8	(17.4)	0.095
	Sandberg's Bluegrass	11.4	(8.6)	10.8	(5.4)	0.783
	Total Native Perennial Vegetation	70.4	(8.6)	55.6	(23.0)	0.018
	Total vegetation	74.1	(10.1)	59.3	(20.9)	0.012
	Litter	13.4	(2.9)	23.2	(12.6)	0.005
	Bare Ground	8.8	(6.5)	24.8	(20.3)	0.005

DISCUSSION

The outcome of rehabilitation and restoration efforts frequently depends to a large extent upon climatic factors beyond the control of restoration practitioners (Stevens and Monsen 2004, Monsen 1994, Jordan 1983). Both 2003 and 2004 experienced growing seasons with normal temperatures and above-normal precipitation at Hanford. Differences in the form and distribution of precipitation between the two years may have important implications for the outcome of the BAER rehabilitation project as well as for the overall development of affected plant communities.

The first growing season is critical for the establishment of new seedlings. Whether plants are installed as outplantings or by direct seeding, moisture in the upper layers of the soil is critical during the period before roots establish and penetrate to depths where the availability of moisture is more reliable and lasting. Precipitation at Hanford for the 12-month period (December 2002 – November 2003) during and following planting and seeding on the ALE Reserve was 8.54" or 122% of normal (HMS 2004). Precipitation during winter and early spring was especially favorable for plant establishment and early growth; precipitation from December 2002 through April 2003 (7.54") was more than 200% of normal. However, precipitation from May through November 2003 (1.0") was only 30% of normal, and an extended drought period that year was not substantially relieved until early December 2003. The early onset of dry weather coupled with the late arrival of fall rains may have contributed to increased stress and possibly mortality of seedlings that might have survived with more normally distributed precipitation.

Plants that survived their first year in 2003 may have experienced more favorable conditions in 2004. Precipitation during the period December 2003 through November 2004 was again well above normal (9.55", more than 130% above normal; HMS 2004). A full 5.0 inches fell between December 1st 2003 and February 29th 2004, and snow covered the ground at low elevations for nearly the entire month of January. Although rainfall was somewhat below normal in March and April 2004, above-normal precipitation resumed in May and June. Following the summer dry period, substantial rainfall returned, in timely fashion, by mid-October. Although elevated soil moisture may be favorable to new plantings, above-normal winter and spring precipitation also favors increased abundance of cheatgrass and increased competitive stress upon newly emerged seedlings.

Cheatgrass abundance and effects of herbicide treatments. Effects of herbicide treatments and native seedings on cheatgrass abundance within the rehabilitation project area were slight, if any. Herbicide treatments may have dampened increases in cheatgrass abundance somewhat, as percent cover of cheatgrass within the project area in 2003 remained statistically similar to pre-treatment levels while cover in untreated areas increased significantly by more than 2.0% ($P = 0.009$). Nevertheless, cheatgrass cover and density over most of the project area (with the exception of elevations over 800 ft. in Polygon 3) were still sufficiently high to interfere with native seedlings (Rafferty and Young 2002, Evans et al. 1970) and likely contributed to reduced emergence and recruitment at lower elevations.

Glyphosate has been used effectively to relieve restoration plantings from competition with cheatgrass and other weedy species (D. Larsen pers. comm., J. Benson pers. comm., Org 1994). Soil moisture, air temperature, precipitation and biological activity in the target species at the time of application can influence the effectiveness of glyphosate in field applications (Monsanto Inc. 2002). Roundup™ is a post-emergence herbicide and has no effect on plants that germinate after treatment is applied. Cheatgrass germinates opportunistically when environmental conditions, especially soil moisture, are favorable (Young and Evans 1985). Periods during the winter and spring of 2003 were extremely moist (HMS 2004), favoring multiple germination events for cheatgrass that very likely swamped potential treatment effects. Record precipitation during April 2003, for example, likely promoted germination of this invasive annual well after the final glyphosate application in February of that year.

Big sagebrush outplantings and seeding. Based on our samples, initial sagebrush outplanting densities failed to meet specifications rehabilitation plans. Although average planting densities of Wyoming big sagebrush were reasonably close to project specifications, it is surprising that so few of the samples matched or exceeded specifications or that so few came close. This is especially puzzling in light of planting crews' compliance with USFWS instructions to avoid planting in weed-infested draws. While the actual proportion of the planting polygons occupied by draws has not been quantified, it is not insignificant. Removing these areas from planting could be expected to move average planting densities somewhat above the original specifications, but this was not apparent in most of the sample plots.

Overall survival of outplanted bare root and tube-grown seedlings of Wyoming big sagebrush in this study differed from the results of other big sagebrush outplantings on ALE over a comparable period. Overall survival in this study (36.3% ± 26.0 SD) was slightly higher than the 28.9% (± 15.7 SD) reported by Newsome (2004) for two-year old seedlings planted in habitats similar to those in this study. The survival rate for bare root stock (19.0% ± 12.3 SD) in Newsome's surveys was slightly better than what we observed in our surveys, but tubling stock survived at a lower rate (32.7% ± 21.2 SD) than in our surveys. Durham and Sackschewsky (2004), however, reported overall survival of 55.5% from samples that included plantations at low elevations and on sandy soil habitats that were not represented in plantations monitored by our study. Durham and Sackschewsky (2004) reported just under 48.0% survival of tubling stock after the second year, which is comparable to the rate of survival of tubling stock that we observed. The primary difference between the two studies is that Durham and Sackschewsky (2004) reported nearly 60.0% survival of bare root seedlings after two years, while bare root survival was under 35.0% in our study. The underlying reason behind this difference was the performance of bare root plants treated with the mycorrhizal hydrogel, which exhibited less than 15.0% survival over two years. In contrast, bare root plants treated with a non-mycorrhizal hydrogel exhibited greater than 70.0% survival, outperforming the bare root plants surveyed by both Durham and Sackschewsky (2004) and Newsome (2004).

None of the environmental and community variables measured by this study offer a ready explanation for the poor performance of sagebrush outplantings in the mycorrhizal treatment polygons. Competition from either alien or native species may reduce survival of big sagebrush seedlings (Stevens and Monsen 2004, Schuman et al. 1998, Downs et al. 1995, Owens and Norton 1989). In our study, however, even though total cover of alien and native grasses was greater in the mycorrhizal polygons, this factor was only weakly correlated with sagebrush survival.

Wyoming big sagebrush forms mycorrhizal associations and may be obligately mycorrhizal (Allen 1982 cited in Schuman et al. 1998). Mycorrhizal inoculation has been shown to enhance shoot growth and increase the tolerance of this species to soil moisture stress in field tests (Stahl et al. 1998, Call and McKell 1981). Durham and Sackschewsky (2004) found that mycorrhizal inoculation enhanced sagebrush survival over that of controls on some sites, and did not adversely affect survival at other sites. In this study, however, big sagebrush outplantings exhibited greater survival in the absence of mycorrhizal inoculation in 2003 (Table 2.9). Based on what is known regarding mycorrhizal associations with this species it seems unlikely that the addition of mycorrhizae would be deleterious to seedling establishment and survival on the ALE Reserve. Laboratory analyses of a small number of samples (n = 18) collected from mycorrhizal and nonmycorrhizal plots indicated a significantly greater occurrence of endomycorrhizal colonization of sagebrush roots in the treated plots (11.6% ± 5.4 SD) compared to untreated plots (1.1% ± 2.2 SD; $P < 0.0001$), indicating that the treatment was effective in promoting mycorrhizal colonization of plant roots (Table 2.31). However, no difference was observed between live, stressed, and dead plants' roots within the same plot or treatment, suggesting that mortality was independent of the level of mycorrhizal colonization. Laboratory observations revealed an abundance of unidentified nonmycorrhizal brown fungus in abundance on the exterior of all of the sagebrush roots examined, regardless of plot or condition. The origin of this nonmycorrhizal growth is unknown, but its presence and abundance may have implications for the health of affected plants.

Differences unrelated to mycorrhizae in the two hydrogels used, differences in application rate and other contractor practices, and other uncontrolled factors confound efforts to explain the discrepancy in survival between mycorrhizal and non-mycorrhizal bare root seedlings in this study. However, available evidence does provide some grounds for an hypothesis.

The roots of Wyoming big sagebrush require high levels of oxygen for normal growth. Oxygen restriction in the rooting zone for as little as a few days following precipitation events can result in injury and eventual death, perhaps because affected roots become vulnerable to invasion by pathogens (Lunt et al. 1973). High moisture levels may also promote the proliferation of parasitic fungi (Allen et al. 1987). The brown fungi observed on big sagebrush roots may be an indication of this. A combination of factors including the high moisture-holding capacity of the fine-textured silt loam soils of the planting sites, much greater-than-normal precipitation during the winter and early spring of 2002-2003, and the application of hydrogel to plant roots prior to planting may plausibly have contributed to low rates of survival in the affected polygons (M. Amaranthus pers. comm.). Differences in characteristics between the Plant Success™ and Terra Sorb™ hydrogels, or differences in the rates of hydrogel application by different contractors may account for the discrepancy in survival between mycorrhizal and non-mycorrhizal treatments. While available evidence does not allow for a definitive explanation of this phenomenon, improved experimental control during future plantings may help provide information that addresses this question and enhances the performance of restoration plantings on the ALE Reserve and elsewhere (see Conclusions and Recommendations).

Table 2.31. Mycorrhizal colonization of roots of Wyoming big sagebrush: treated (M +) vs. untreated (M -) plants. Data courtesy of M. Amaranthus, Mycorrhizal Applications, Grants Pass, OR.

Polygon	**Treatment**	**Condition**	**Percent Endomycorrhizal Colonization**
B	M -	Live	0
B	M -	Live	4
B	M -	Live	6
B	M -	Stressed	0
B	M -	Stressed	0
B	M -	Stressed	2
B	M -	Dead	0
B	M -	Dead	0
B	M -	Dead	0
J	M +	Live	24
J	M +	Live	8
J	M +	Live	8
J	M +	Stressed	12
J	M +	Stressed	10
J	M +	Stressed	14
J	M +	Dead	8
J	M +	Dead	6
J	M +	Dead	14

The high rates of survival of outplantings that were not inoculated with mycorrhizae suggest that mycorrhizae are not necessary for shrub establishment, at least during periods of above-normal winter and spring precipitation. Whether mycorrhizae enhance survival of Wyoming big sagebrush during periods of average or below average precipitation is a question that should be addressed experimentally in the future.

Most surviving sagebrush outplantings surveyed during fall 2004 appeared to be vigorous and well-established and site trajectories towards the development of mature shrub canopies in these areas appear promising. Despite low rates of survival in some areas, most polygons (except Polygon C) can be expected to develop a shrub component over the coming years far in excess of that which would have developed in the absence of active enhancement. Polygons with the highest survival (the non-mycorrhizal polygons B and K, along with the tubling polygons A and LM) hold the promise of developing into relatively dense stands within the next decade.

A small percentage of outplantings flowered in 2004, primarily in the untreated bare root polygons. As the majority of surviving plants become reproductive over the next several years, natural establishment should begin to augment plantation densities and sagebrush may begin to move, albeit slowly, into adjacent areas.

The apparent failure of Wyoming big sagebrush to establish via aerial seeding in the shrub seed polygon is not surprising. Broadcast seeding has been as successful as other methods of establishing big sagebrushfrom seed (Meyer 1994) and is recommended over drill seeding. Nevertheless. attempts to establish big sagebrush from seed have exhibited variable success, and low rates of establishment are not uncommon (Monsen and Richardson 1984). Above normal precipitation during the winter and early spring of 2003 favored the emergence of broadcast-seeded big sagebrush. The prolonged drought of late spring and fall 2003, along with competition from established native perennial grasses and invasive annuals (Schuman 1998, Eliason and Allen 1997, Evans and Young 1978), likely exceeded the stress tolerance of vulnerable new seedlings.

Most emerging big sagebrush seedlings do not survive beyond the first year (Meyer 1994). Natural establishment of big sagebrush seedlings in arid lands may be episodic, limited to occasional years of above-normal precipitation and favorable temperatures (Meyer 1994, Young and Evans 1989, Daubenmire 1975). In extreme environments such as in the lower elevations of the Columbia Basin there may be many years between the coincidence of conditions favorable to large-scale seed production, seedling emergence, and recruitment.

Most seed of Wyoming big sagebrush germinates or loses viability within the first season, although a small percentage of seeds may retain viability for longer (Schuman et al. 1998, Meyer 1994, Young and Evans 1989). Successful germination of big sagebrush seed depends upon adequate levels of moisture, favorable temperatures (Meyer 1994, Boltz 1994), and seedbed characteristics such as the availability of cracks or other depressions for safe sites within the soil surface (Eckert et al. 1986). Germination occurs across a broad range of temperatures, and although germination tends to increase with temperature, Wyoming big sagebrush is capable of germination within the range of temperatures that occur during winter months within the project area (Meyer and Monsen 1992, McDonough and Harniss 1974, Weldon et al 1959). Germination rates ranging from 1 – 100 % have been reported for Wyoming big sagebrush in laboratory studies (Meyer and Monsen 1992, Harniss and McDonough 1976, McDonough and Harniss 1974). Maximum rates may be lower in the field. Germination may vary considerably between sites, years, and parent plants (Harniss and McDonough 1976).

In a field experiment conducted on the Hanford site, Downs and others (1995) found that big sagebrush seedings into a dense native grassland habitat responded positively to herbicide application but not to soil disturbance. Application of glyphosate herbicide (4.0 oz. product / gal. water) in late February in native grasslands reduced cover of Sandberg's bluegrass but had no apparent effect on other native perennial bunchgrasses or forbs which were still dormant at the time (Downs et al. 1995). Seedings into annual grasslands have responded positively to a combination of soil disturbance (harrowing) and light herbicide applications in late winter prior to germination of sagebrush seeds (D. Larsen pers. comm.). The use of soil disturbance as a seedbed preparation may conflict with other conservation values in areas with functional or recovering microbiotic crusts, however (Belnap 1999).

Grass seed density, emergence, and recruitment. Overall seed densities in sample plots were 2.6 to 6.6 times higher than specified densities. High seeding rates may be necessary in order to achieve adequate stocking densities for habitat stabilization and recovery (Launchbaugh and Owensby 1970), as rates of seedling recruitment observed in this and other studies suggest. Absolute seedling densities increase with increasing seed densities, but rates of emergence and recruitment decline as seed density is increased (Launchbaugh and Owensby 1970, McGinnies 1970).

The great variability in seeding densities of individual species observed in this study may have resulted during the mixing process or as a result of vibration-induced differential movement of seeds of different weights and sizes while in the airplane during applications. Seed rate specifications based on density may not be relevant when application of large quantities of seed by weight is involved. This may be especially true when seed mixes include large quantities of small, light seeds such as those of Sandberg's bluegrass.

Emergence rates of grass seedlings in aerial seeding treatments were low when compared to rates of germination observed in laboratory studies. Germination rates from 33.4 % to 95.0 % have been reported for bluebunch wheatgrass in laboratory studies (Hardegree 1996, Kitchen and Monsen 1994, Plummer 1943). Hardegree (1996) reported germination rates of 92.0% for thickspike wheatgrass and 70.0% for Sandberg's bluegrass, while Evans et al. (1977) reported germination rates of 38% - 95% for Sandberg's bluegrass following six months of afterripening. Germination test results for all species used in the 24 Command Fire rehabilitation were greater than 80**%** (Table 2.32). Actual germination in the field is likely to be lower than results of laboratory tests performed under optimal conditions.

Table 2.32. Germination test rates of species and cultivars used in rehabilitation seedings, ALE Reserve, December 2002-January 2003. From Smith et al. (2003).

Species (Cultivar)	**Germination Rate (%)**
bluebunch wheatgrass	91.0
Hells Canyon bluebunch wheatgrass	87.4
bluebunch wheatgrass ('Anatone')	87.0
bluebunch wheatgrass ('Secar')	92.0
thickspike wheatgrass ('Schwendimar')	92.3
Indian ricegrass ('Nezpar')	93.6
Sandberg's bluegrass	84.0 - 87.6
needle-and-thread	84.8
squirreltail ('Sandhollow')	84.3 - 86.0
yarrow	84.0 - 89.5

No target density for grass seedling establishment was made explicit as part of the BAER rehabilitation effort. Data on densities of native bunchgrasses on seeded range or in natural habitats are scarce in the ecological literature, and few models are available to which to compare results from this study. The Great Plains Agricultural Council (1966, cited in Vallentine 1989) developed rating systems for seeded stands in different climatic zones. Based upon the system most closely applicable to conditions on the ALE Reserve (Table 2.33), recruitment stocking rates achieved by the BAER project on ALE (16.3 seedlings/ m^2 ± 28.5 SD) may be rated as excellent, primarily due to high densities of Sandberg's bluegrass.

Overall potential recruitment densities reported in this study (9.8 to 33.2 seedlings/m^2) were higher than those observed in post-fire bunchgrass mortality plots in comparable sites[4] on the ALE Reserve (3.9 tussocks/m^2; Section I, this volume). These survival plot densities are not directly comparable to our overall recruitment densities, since the latter include seedlings of Sandberg's bluegrass, which species is largely excluded from the former study by a minimum size requirement (Section I, this volume). Moreover, the mortality surveys recorded densities of large, mature bunchgrasses, which also may not be directly comparable to densities of recently established seedlings.

Potential recruitment densities for wheatgrass seedlings in three of four rehabilitation treatment types were very similar to density values for bluebunch wheatgrass in mortality plots (2.1 tussocks/ m^2 ± 1.8 SD). Only the low elevation seed mix over silt loam soils, the treatment which covered the largest area, exhibited densities that were substantially lower than these values. After two years, seedling plants are still quite small. In the rehabilitation areas where recruitment densities are comparable to natural habitats, it remains to be seen whether these, recently established seedlings will continue to survive and expand to the point where the desired habitat stabilization has been achieved.

Potential recruitment densities of seeded species in treatment plots did not differ statistically from seedling densities in control plots, suggesting that aerial seeding had no effect on seedling density. However, control plots were outside the project area and had higher densities of established perennials as a source of seed than did the treatment plots. This factor could have been especially significant with regard to wheatgrass species, which were entirely absent below approximately 800 ft.(244m) elevation within the project area prior to seeding.

For a number of reasons, actual emergence on ALE may have been greater than our data reflect. Seedling emergence of Sandberg's bluegrass was observed in a non-systematic check of polygons 1, 2, 3, and a portion of the drill seed strip on February 13th, 2003 (pers. obs.), just prior to followup herbicide treatments in mid-February. The impact of this treatment on newly-emerging perennial seedlings is not known. Bluebunch wheatgrass may also begin to germinate in late winter and continue until late spring (Humphrey and Schupp 1999). Emergence of bluebunch wheatgrass cultivars may have continued after sampling was completed in plots surveyed near the beginning (i.e., during April) of the 2003 field season. However, given the absence of spring

[4] This comparison uses mid-elevation bluebunch wheatgrass and needle-and-thread grasslands as a reasonable measure of target densites for habitat stabilization measures.

Table 2.33. Rating system for evaluation of seeded stands on foothill ranges in the Intermountain Region (11 – 13 in. precipitation zone). Adapted from Great Plains Agricultural Council (1966, cited in Vallentine 1989).

Rating	Stocking Rate (Seedlings/ft.^{2})	Stocking Rate (Seedlings/m^2)
Excellent	$\geq$ 0.75	$\geq$ 8.1
Good	0.5 - 0.75	5.4 – 8.1
Fair	0.25 – 0.5	2.7 – 5.4
Poor	< 0.25	< 2.7

precipitation on the Hanford Site after April 30^{th} 2003 (HMS 2004) it seems unlikely that any substantial germination would have taken place after that date.

The rates of grass seedling emergence and recruitment fromaerial seeding efforts observed in this study are probably typical of broadcast seeding efforts in semiarid environments (Nelson et al. 1970). Performance rates for direct seeding efforts are frequently less than 10% (Vallentine 1989, Launchbaugh and Owensby 1970). Under natural conditions, soil moisture availability limits seed germination, seedling emergence, and early seedling growth (Abbott and Roundy 2003, Chambers 2000, Abbott et al. 1995). Native perennial grasses and forbs favor soil coverage or cracks and depressions in the soil surface as safe sites where soil moisture is more favorable for germination and development than at the surface (Chambers 2000, Winkel et al. 1991, Eckert et al. 1986, Evans et al.). Surface-sown seed are exposed to high temperatures and desiccation during a time when they are extremely vulnerable (Nelson et al. 1970).

The higher emergence rates observed at upper elevations in the project area may reflect somewhat more favorable moisture conditions for seedling emergence and early survival in these habitats.

In addition to soil moisture, biological factors may also affect the outcome of wildland seeding efforts. Seed predation and grazing by small mammals and birds may influence rates of emergence and establishment in rehabilitation plantings (Archer and Pyke 1991, McAdoo et al. 1987, Rogers and Gano 1980. Johnson 1977). Nelson et al. (1970) report seed predation by rodents and birds of over 90% of broadcast seeds in some trials (although predation is unlikely to be so high where seeds are broadcast into existing vegetation as was the case in much of the ALE site). Pyke (1987, 1986) recorded significant effects of timing and frequency of grazing by small rodents on survival of seedlings of bluebunch wheatgrass. Seedlings were most vulnerable when they were very young (less than 30 days old) or when grazed during late winter. Grazing habits of black-tailed jackrabbits at Hanford showed strong preferences for two species, needle-and-thread and yarrow, included in BAER project seed mixes (Uresk 1978). Competition both with established vegetation and among new seedlings also contributes to mortality and influences restoration outcomes (Pyke and Archer 1991, Nelson et al. 1970).

Seed dormancy may contribute to lower observable emergence (Chambers 2000). Bluebunch wheatgrass and squirreltail exhibit little seed dormancy (Kitchen and Monsen

1994, Young and Evans 1977). Needle-and-thread exhibits some delayed germination, although this trait may vary with seed source (Monsen et al. 2004). Optimum germination for Sandberg's bluegrass occurs only following a six-month afterripening period (Evans et al. 1977). High seed dormancy is a widely recognized characteristic of Indian ricegrass (*Oryzopsis hymenoides*). This species was found in some abundance in seed traps in areas seeded with the low elevation seed mix; however no seedlings of Indian ricegrass were recorded in the field in 2003, and only a handful of ricegrass seedlings were identified (from a single plot) in 2004. Seeds of Indian ricegrass may require scarification, stratification, or rodent activity such as harvesting, processing, and caching to induce germination (Jones 1990, McAdoo et al. 1983, Young et al. 1983). Longland (1995) found field germination of ricegrass seeds in rodent caches to exceed 90%, while laboratory germination of hand-collected seeds was only 2%. Dormant seeds of ricegrass may remain viable for many years, however (Hull 1973), and ungerminated seeds from this project may become germinable in years to come.

Drill seed treatment. While drill seeding is generally accepted as a much more reliable method of achieving seeding success (Vallentine 1989, Nelson et al. 1970), there were no significant differences in either emergence or recruitment in the drill seed treatment compared to aerially seeded areas on ALE. The large-seeded bluebunch wheatgrass cultivars exhibited higher emergence within drill seed plots in 2003 compared to emergence in plots in comparable sandy soils at low elevation in the aerial seeding treatments, although these differences were not significant (Table 2.17).

Strong negative correlations with percent cover of native perennials suggests that potential recruitment of wheatgrass seedlings in the drill seed strip was related to competition. Interestingly, this correlation was much stronger with established perennials than with cover or density of cheatgrass.

Numerical differences in performance between the three wheatgrass stocks installed by drill seeding were not significant. Anatone bluebunch wheatgrass erxhibited nearly four times greater emergence in 2003 than either of the other stocks, but numerically greater emergence did not translate into greater recruitment in 2004. The highest recruitment, both in absolute terms and as a percentage of emergence dxensity, belonged to the Hells Canyon stock. Low recruitment rates for all three wheatgrass stocks probably reflect the serious obstacles to seedling establishment in the extreme environment of this portion of the Columbia Basin. Drill seeding was carried out in the warmest, most arid portion of the ALE Reserve where annual precipitation averages only 6.25" (160 mm)/ year. Anatone bluebunch wheatgrass is recommended for sites receiving at least 10" (250 mm) of annual precipitation (Monsen et al. 2003, cited in Monsen et al. 2004), and is reported to be more competitive against cheatgrass at lower elevations in the Intermountain West than other bluebunch wheatgrass cultivars (NSN 2004a). Secar is reported to be more drought tolerant than Anatone, being recommended for sites receiving 8"-12" (200 – 300 mm) of annual precipitation (Alderson and Sharp 1994 cited in Monsen et al. 2004). Although originally considered to be a bluebunch wheatgrass cultivar, Secar has more recently been described as Snake River wheatgrass (*Elymus wawawaiensis*). The precipitation regime of the Cold Creek planting area lies below the recommended limits for all three wheatgrass stocks (NSN 2004a, b, Monsen et al. 2004, R. Herman pers. comm.).

Sandberg's bluegrass density increased dramatically in 2004 in drill seed plots north of Milepost 8. This increase could be related to delayed germination of seed from 2003 (Evans et al. 1977). Although we do not know how long this species retains its viability in the field, there is evidence that Sandberg's bluegrass remains viable in storage for at least a decade (J. Downs, pers. comm.). Emergence in 2004 could also have resulted from *in situ* seed production by established perennials, which occurred nearby.

The Cold Creek Valley corridor that borders SR 240 along the eastern margins of the ALE Reserve is a mosaic of native and non-native vegetation. Certain portions of this corridor, such as the area between mileposts 6 and 9, and the area surrounding mileposts 18 and 19, support significant concentrations of native perennial vegetation. Other areas, such as the area between gates 118 and 120, and long stretches south of milepost 9, are dominated by alien plant communities with no conspicuous native component. Areas where native perennial plants do dominate or codominate, while neither pristine nor weed-free, are reservoirs of low elevation native biodiversity and probably function at least as well as the target greenstrip vegetation in terms of firebreak effectiveness.

Recovery of bulldozer fireline. While the 24 Command Fire exerted a broad and profound influence on the shrub-steppe ecosystem of the ALE Reserve, the bulldozed fireline in upper Snively Basin introduced yet another level of disturbance to an area of high-quality native vegetation. Measures directed at enhancing the recovery of the fireline swath have not overcome the huge disturbance caused by its construction. Three years after rehabilitation efforts, the swath is still visible from across Snively Basin because of significant differences in vegetation between it and the surrounding high-quality bluebunch wheatgrass–Sandberg's bluegrass grassland.

Results of 2004 monitoring of fireline recovery were similar to findings from 2003 field surveys. Cheatgrass abundance by all measures (percent cover, frequency, and density) continued to be significantly higher in the suppression swath than in the surrounding area. Whether Sandberg's bluegrass is more abundant within the suppression swath than it would have been in the absence of the 2001 broadcast seeding cannot be determined; however, percent cover of Sandberg's bluegrass and other key native elements such as bluebunch wheatgrass and microbiotic crust remained significantly reduced within the affected area.

Native vegetation in the affected area is unlikely to recover fully so long as cheatgrass and other invasive species are competing in their present abundance. In some environments a narrow swath through a high-quality native plant community might reasonably be expected to reseed naturally from surrounding sources. However, cheatgrass aggressively colonizes disturbed areas **(**Mack 1981, Stewart and Hull 1949) and outcompetes the seedlings of most native bunchgrass species (Rafferty and Young 2002, Aquirre and Johnson 1991, Harris 1977, 1967). In the absence of substantial competition from established perennials, cheatgrass will likely preempt the recovery of native vegetation and can be expected to increase within the suppression area over time. If cheatgrass succeeds in dominating the suppression area, the swath may begin to function as an inoculum of invasive species and facilitate the spread of invasives out into the surrounding high quality habitat.

Conclusions and recommendations. No project can hope to restore the ecological structure and function of a complex ecosystem within a few years. The practical hope of the restorationist is to set the landscape on a successional trajectory that will lead to the recovery of ecological processes and habitat quality within a reasonable period of time. Complete recovery of the structure and function of ALE shrublands impacted by the 24 Command Fire is still decades away. It will take many years of continued planting and monitoring, persistent efforts at weed and fire management, and years of patience as restored stands slowly develop.

Disturbed sites in the Wyoming big sagebrush steppe of the Columbia Basin present extreme challenges to restoration efforts due to the warm, semi-arid climate, the presence of aggressive invasive species, and the increasingly frequent occurrence of wildfires within the region. The probability of achieving success in restoration projects within this region has been rated as no better than moderate, and restoration outcomes depend upon effective fire management and control of invasive species in addition to seeding and planting success (Stevens and Monsen 2004, Bunting et al. 2003, Hemstrom et al 2002, Monsen 1994, Nelson et al. 1970).

Despite low survival rates in some plots, percent survival in sample plots indicates that more than 254,000 of 700,000 outplanted sagebrush seedlings may have established successfully on ALE. The general success of sagebrush outplantings reported by this and other local studies (Newsome 2004, Durham and Sackschewsky 2004) provides an encouraging sign that this component of the ALE ecosystem may be restorable. Outplanting of nursery-grown stock is more reliable than direct seeding as a method for restoring shrubs to the landscape (Monsen and Richardson 1984) and it is highly recommended that this practice be continued on the ALE Reserve beyond the limits of the BAER program. There are tens of thousands of acres on the ALE Reserve that legitimate candidates for the reintroduction of Wyoming big sagebrush, along with threetip sagebrush and spiny hopsage in appropriate habitats. The establishment of shrub islands is desirable at all elevations. Plantations at higher elevations will enjoy the benefits of higher precipitation and more moderate growing season temperatures that should facilitate establishment of outplantings during average years.

One additional year of monitoring the existing array of sagebrush survival plots in 2005 is strongly recommended in order to document that the trends described in this study are firmly established. Annual survival monitoring can be completed by a crew of three people in no more than four to five days. Biologists may elect to forego monitoring plots with less than 10.0% survival (plots D2, D3, F1, and F3), since little additional information will be learned from the survival of the remaining small fraction. Survival surveys are well-suited to utilizing assistance from volunteers working under experienced supervision. After 2005, periodic monitoring at five-year intervals will provide valuable information with which to calibrate performance objectives for future shrub introduction projects. In order to assess the impacts of restoration on habitat quality, monitoring of invertebrate populations and wildlife use of these developing sagebrush stands is also highly recommended.

It is strongly recommended that at least some portion of all future shrub plantings on the Hanford Reach National Monument be designed as scientific experiments. Experimental design should control for complicating factors such as hydrogel characteristics, potential subtle differences in similar soil types, differences in plant competition, and other factors that contributed to the difficulty in interpreting current monitoring results. Final experimental designs can be reviewed by the USFWS Land Management and Research Demonstration Biologist, by Arid Lands Ecologist and the Director of Research Programs at The Nature Conservancy of Washington, and/ or by other interested and qualified parties. Important questions that remain to be answered with precision include the following:

1. Does inoculation with mycorrhizal fungi enhance the establishment, growth rates, and long-term survival of bare root seedlings of Wyoming big sagebrush? Does the importance of mycorrhizae differ between years of above-normal precipitation compared to years of normal or below-normal precipitation?
2. How do different methods of inoculation with mycorrhizal fungi compare regarding resulting rates of shrub performance and survival?
3. Do hydrogel treatments enhance the establishment and long-term survival of bare root seedlings of Wyoming big sagebrush on silt loam soils? Are there differences in the performance of commonly available hydrogel products? On what soil types and under what soil moisture conditions might hydrogel treatments be either beneficial or deleterious to survival of big sagebrush plantings on the Hanford Reach National Monument?

The surprising results of our surveys suggest that Plant Success™ mycorrhizal hydrogel may not be appropriate for use in Wyoming big sagebrush plantings on silt loam soils on the ALE Reserve. Until controlled experiments provide clearer answers regarding the questions posed above, prudence demands that we recommend against the use of this method of mycorrhizal inoculation in sagebrush plantings on silt loam soils on the Hanford Reach National Monument. Mycorrhizal inoculation via alternative methods is still recommended. The most effective method of delivering mycorrhizae to seedling roots is through inoculation of the growing medium at the nursery (M. Amaranthus pers. comm.). Dry tablets containing mycorrhizal inocula may be dropped into holes with seedlings at time of planting if nursery inoculation is not possible.

Additional management intervention will be required within the BAER rehabilitation project area in order to stabilize soils, suppress invasive species, and promote recovery of these critical wildlife habitats. Continued monitoring of selected areas, such as the former sagebrush stands between the Gate 117 and Gate 118 roads and elsewhere, will assist resource managers in developing plans for further treatment of areas at risk.

Firebreaks along SR 240 and other highway corridors should fully incorporate the highway and its margins. The performance of firebreaks is a function of total width (Wilson 1988). A greenstrip immediately adjacent to the highway corridor (with no intervening vegetation) would take advantage of this additional fuel break and likely double the effectiveness of the installation.

Future efforts to establish and maintain a greenstrip along the SR 240 corridor should carefully assess the condition of perennial vegetation along the corridor. Portions of this corridor, such as the areas mileposts 6 and 9, and around mileposts 18 and 19, currently support fair to good quality native perennial vegetation. While neither pristine nor weed-free, these areas are reservoirs of low elevation native plant biodiversity and probably function at least as well as the target greenstrip vegetation in terms of firebreak effectiveness. These areas should not be disturbed unnecessarily. Greenstripping activities along these stretches may serve no fire management purpose and may disrupt functioning communities. It is recommended that these areas be monitored and maintained as diverse natural plant communities. Excluding them from disruptive interventions will allow management resources to be focused on establishing perennials along portions of this corridor where they are presently largely absent.

Middle to higher elevation grasslands such as those in upper Snively Basin are among the more intact native plant communities of the Arid Lands Ecology Reserve. While not immune to perturbation and the proliferation of invasive plant species, environmental factors along with the biological characteristics of established native plant communities at these elevations may confer upon the herbaceous components of these communities a resiliency that is lacking in stands at lower elevations. Where restoration efforts are required, higher precipitation and reduced evaporative demand associated with lower growing season temperatures increase the likelihood of favorable outcomes for restoration projects at such sites (see Section I, this volume).

One to two years of effective control of cheatgrass will be necessary within the Snively Basin suppression area to allow native vegetation to recover, whether or not the area is enhanced with additional native seedings or plantings. Chemical treatments can be applied during late winter or very early spring, when cheatgrass is active, but most native perennial herbaceous species are not (Downs et al. 1995, D. Larsen pers. comm.). One of the desired native matrix species, Sandberg's bluegrass, however, is active during the time of year when herbicide treatments are most often recommended. Insofar as possible within this narrow strip, herbicides should be applied as spot applications carefully targeted to avoid contact with Sandberg's bluegrass and other cool season native perennials.

In concert with invasive species control, reintroduction of important structural native grasses will greatly increase the likelihood of recovery of the disturbed area. Seeds of bluebunch wheatgrass and Sandberg's bluegrass may be applied preferably in a fall dormant seeding and lightly raked into the soil. Herbicide treatments should be timed carefully to precede emergence of native seedlings or spot applied carefully to avoid impacts on native cohorts.

Successional approaches to restoration recognize that post-disturbance conditions may not be well-suited to the environmental requirements of late-successional species that dominated the site prior to disturbance (Whisenant 1999). Squirreltail (*Sitanion hystrix*) is an early seral native bunchgrass that has shown competitive ability when grown with invasive annual grasses (Arredondo et al. 1998, Hironaka and Sindelar 1973). Although this species is not abundant in less disturbed areas surrounding the fireline swath, introduction of quantities of squirreltail may help to secure the site from invasive annuals while climax species such as bluebunch wheatgrass recolonize more slowly.

Over an area this small it is feasible to augment broadcast seeding of bluebunch wheatgrass and squirreltail with nursery-grown bare root or container seedlings. Regardless of the density at which they are planted, installation of nursery-grown seedlings will hasten the recovery of large bunchgrasses within the site, and the more mature plants will compete more effectively against invasive annuals than will newly-germinated seedlings.

Within this high quality native grassland it is especially important to use locally derived native plant stock. If appropriate seed or plant material is not available it may be better to postpone seeding or planting for a year than to proceed with cultivars or with collections from distant sites. Seed may be collected from the surrounding vegetation during the next growing season if no other appropriate sources are available. Capable volunteers are available in the Tri-Cities area willing to assist in seed collection and broadcast efforts in this area. Recovery of native vegetation within the suppression swath should reduce the threat of cheatgrass increasing in abundance throughout this high-quality habitat.

Sources of seeds for rehabilitation or restoration projects should be carefully considered. Existing native vegetation on the Hanford Site represents a poorly understood but potentially irreplaceable genetic resource, a pool of genetic variability uniquely adapted over millennia to the unique range of environmental variability that characterizes this portion of the Columbia Plateau. Seeds collected from distant sites may introduce genotypes that are poorly adapted to conditions in the Columbia Basin, may fail to establish vigorous populations or, if successful, may have unpredictable impacts on the genetic integrity of local stocks (Belnap 1995, Linhart 1995, Knapp and Rice 1994, Millar and Libby 1989).

The lack of commercial availability of sufficient quantities of locally derived native seed was a limiting factor in this project as it often is in large-scale rehabilitation and restoration projects (Stevens and Monsen 2004, Richards et al 1998). Wildfire will remain a part of the shrub-steppe landscape and resource management agencies will do well to plan responses to future wildfires before they occur, rather than forcing managers to scramble to respond after the disturbance has occurred. Restoration opportunities are often greatest within the first one to several years following wildfire, before invasive species populations recover or newly colonize a site (Monsen and Stevens 2004, Evans and Young 1978, Section I this volume). For the Hanford Reach National Monument to take advantage of this window of opportunity it must develop, either on its own or in concert with partners and contractors, the capacity to stockpile native seeds that will be available when needed soon after disturbances occur. Many of the key structural species of native shrub-steppe plant communities of the Columbia Basin maintain viability for many years when properly stored (Table 2.34). The USFWS and DOE, as co-managers of the Hanford Reach National Monument, are urged to take the lead in investigating the feasibility of instituting a Columbia Basin native seed storage bank. If willing partners are found, a seed storage bank of this kind could serve a consortium of federal and state agencies such as the U.S. Bureau of Land Management, Washington Department of Fish and Wildlife, Washington Department of Natural Resources, and Washington State Parks.

This recommendation is easy to make from a biological standpoint given the wildfire and community dynamics of the Columbia Basin. Implementation may be much more difficult. Implementing this strategy may require substantial investments of time and other resources from public land stewards already making do with shrinking budgets, and may require fundamental changes in institutional approaches to funding restoration (Richards et al. 1998). However, with the long-term integrity of the shrub-steppe ecosystem in doubt both locally and regionally (Knick 1999b, Knick et al. 2003), bold initiatives are warranted.

Table 2.34. Longevity of seed viability in storage for common structural species of shrub-steppe plant communities in the Columbia Basin. Longevity estimates from Jorgensen and Stevens (2004). Scientific names are from Hitchcock and Cronquist (1973); see Appendix C for updated nomenclature.

Common name	**Scientific name**	**Estimated longevity in storage (years)**
Shrubs		
spiny hopsage	*Atriplex spinosa*	0 - 3
Wyoming big sagebrush	*Artemisia tridentata* ssp. *wyomingensis*	4 - 6
threetip sagebrush	*Artemisia tripartita*	4 - 6
winterfat	*Eurotia lanata*	0 - 3
Grasses		
bluebunch wheatgrass	*Agropyron spicatum*	11-15
Idaho fescue	*Festuca idahoensis*	4 - 6
Indian ricegrass	*Oryzopsis hymenoides*	11-15
Sandberg's bluegrass	*Poa sandbergii*	4 - 6
squirreltail	*Sitanion hystrix*	11-15
needle-and-thread	*Stipa comata*	11-15

LITERATURE CITED

Abbot, L.B., and B.A. Roundy. 2003. Available water influences field germination and recruitment of seeded grasses. Journal of Range Management 56: 56-64.

Formatted: Line spacing: single

Abbott, L.A., B.A. Roundy, and S.H. Biedenbender. 1995. Seed fate of warm season perennial grasses. In B.A. Roundy, E.D. McArthur, J.S. Haley, and D.K. Mann (compilers), Proceedings: Wildland Shrub and Arid Land Restoration Symposium. General Technical Report INT-315. U.S. Forest Service, Intermountain Research Station, Ogden, UT. pp. 37-43.

Aguirre, L., and D.A. Johnson. 1991. Root morphological development in relation to shoot growth in seedlings of four range grasses. Journal of Range Management 44: 341-346

Alderson, J., and W.C. Sharp. 1994. Grass varieties in the United States. Agricultural Handbook 170, U.S. Department of Agriculture, Soil Conservation Service, Washington, D.C. 296 pp.

Formatted: Line spacing: single

Allen, M.F., E.B. Allen, and N.E. West. 1987. Influence of parasitic and mutualistic fungi on *Artemisia tridentata* during high precipitation years. Bulletin of the Torrey Botanical Club 114: 272-279.

Allen, M.F. 1982. Influence of vescicular arbuscular mycorrhizae on water movement through *Bouteloua gracilis* (H.B.K.) lag ex Stued. New Phytologist 91:191-196.

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.

Arredondo, J.T., T.A. Jones, and D.A. Johnson. 1998. Seedling growth of intermountain perennial and weedy annual grasses. Journal of Range Management 51: 584-589.

BAER (Northern States Burned Area Emergency Rehabilitation) Team. 2000. 24 Command Fire Burned Area Emergency Rehabilitation (BAER) Plan. United States Department of Interior. 152 pp. plus appendices

Belnap, J. 1999. Structure and function of biological soil crusts. In P.G. Entwhistle, A.M. DeBolt, J.H. Kaltenecker, and J. Steenhof (compilers), Proceedings: Sagebrush Steppe Ecosystems Symposium. Bureau of Land Management Publication No. BLM/ID/PT-001001+1150. Boise, ID. Pp. 55-64. Available online at: http://www.id.blm.gov/publications/data/SSSymp.pdf

Belnap, J. 1995. Genetic integrity: Why do we care? An overview of the issues. In B.A. Roundy, E.D. McArthur, J.S. Haley, and D.K. Mann (compilers), Proceedings: Wildland Shrub and Arid Land Restoration Symposium. General Technical Report INT-315. U.S. Forest Service, Intermountain Research Station, Ogden, UT. pp. 265-266

Booth, D.T. 2002. Seed longevity and seeding strategies affect sagebrush revegetation. Journal of Range Management 55: 188-193.

Boltz, M. 1994. Factors influencing post-fire sagebrush regeneration in south-central Idaho. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service, Intermountain Research Station. Pp. 281-290.

Bunting, S.C., J.L. Kingery, and M.A. Schroeder. 2003. Assessing the restoration potential of altered rangeland ecosystems in the Interior Columbia Basin. Ecological Restoration 21: 77-86.

Call, C.A., and C.M. McKell. 1981. Vescicular-arbuscular mycorrhizae: A natural revegetation strategy for disposed processed oil shale. Reclamation and Revegetation research 1: 337-347.

Chambers, J.C. 2000. Seed movements and seedling fates in disturbed sagebrush steppe ecosystems: Implications for restoration. Ecological Applications 10: 1400-1413.

Daubenmire, R. 1975. Ecology of *Artemisia tridentata* in the state of Washington. Northwest Science 49: 24-35.

Downs, J.L., W.H. Rickard, and L.L. Cadwell. 1995. Restoration of big sagebrush habitat in southeastern Washington. In B.A. Roundy, E.D. McArthur, J.S. Haley, and D.K. Mann (compilers), Proceedings: Wildland Shrub and Arid Land Restoration Symposium. General Technical Report INT-315. U.S. Forest Service, Intermountain Research Station . pp. 74-77.

Durham, R.E., J.S. Lewinsohn, and M.R. Sackschewsky. 2004. W-519 Sagebrush Mitigation Project, FY-2004 Final Review and Status. Report to the US Department of Energy. PNNL-14901. Pacific Northwest National Laboratory, Richland, WA.

Eckert, R.E., F.F. Peterson, M.S. Meurisse, and J.L. Stephens. 1986. Effects of soil-surface morphology on emergence and survival of seedlings in big sagebrush communities. Journal of Range Management 39: 414-420

Eliason, S.A., and E.B. Allen. 1997. Exotic grass competition in suppressing native shrubland re-establishment. Restoration Ecology 5: 245-255.

Elzinga, C.L., D.W. Salzer, and J.W. Willoughby. n.d. Measuring and Monitoring Plant Populations. Bureau of Land Management Technical Reference 1730-1. 477 pp.

Evans, J.R. 2004. Recovery and Rehabilitation of Vegetation on the Fitzner-Eberhardt Arid Lands Ecology Reserve, Hanford Reach National Monument, Following the 24 Command Fire. Annual report of findings: 2003 Field Season. Report to the US Fish and Wildlife Service, Hanford Reach National Monument. The Nature Conservancy of Washington, Seattle, WA.

Evans, J.R. 2001. Vegetation monitoring on the Fitzner-Eberhardt Arid Lands Ecology Reserve Following the 24 Command Fire of June-July, 2000: Annual Report of Findings. Report to the US Fish and Wildlife Service, Hanford Reach National Monument. The Nature Conservancy of Washington, Seattle, WA. 54 pp. plus appendices

Evans, J.R., J.J. Nugent, and D.E. Ekblaw. 2002. Short-term Impacts of the 24 Command Fire on Vegetation of the Fitzner-Eberhardt Arid Lands Ecology Reserve, Hanford Reach National Monument. Synthesis of Findings 2001-2002. Report to the US Fish and Wildlife Service, Hanford Reach National Monument. The Nature Conservancy of Washington, Seattle, WA. 88 pp. plus appendices.

Evans, R.A., J.A. Young, and B. A. Roundy. 1977. Seedbed requirements for germination of Sandberg bluegrass. Agronomy Journal 69:817-820.

Evans, R.A., and J.A. Young. 1978. Effectiveness of rehabilitation practices following wildfire in a degraded big-sagebrush-downy brome community. Journal of Range Management 31: 184-188.

Evans, R.A., H.R. Holbo, R.E. Eckert, and J. A. Young. 1970. Functional environment of downy brome communities in relation to weed control and revegetation. Weed Science 18: 154 – 162.

Great Plains Agricultural Council. 1966. A Stand Establishment Survey of Grass Plantings in the Great Plains. Great Plains Council Report 23. Nebraska Agricultural Experiment Station, Lincoln, NB. 60 pp.

Hajek, B.F. 1966. Soil Survey of the Hanford Project in Benton County Washington. BNWL-243. Battelle Northwest Laboratory, Richland, WA.

Hardegree, S.T. 1996. Optimization of seed priming treatments to increase low-temperature germination rate. Journal of Range Management 49: 87-92.

Harniss, R.O., and W.T. McDonough. 1976. Yearly variation in germination in three subspecies of big sagebrush. Journal of Range Management 29: 167-168.

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

Harris, G.A. 1967. Some competitive relationships between *Agropyron spicatum* and *Bromus tectorum*. Ecological Monographs 37: 89-111.

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

Hilty, J.H., D.J. Eldridge, R. Rosentreter, M.C. Wicklow-Howard. and M. Pellant. 2004. Recovery of biological soil crusts following wildfire in Idaho. Journal of Range Management 57: 89-96.

Hironaka, M., and B.W. Sindelar. 1973. Reproductive success of squirreltail in medusahead-infested ranges. Journal of Range Management 26: 219-221.

Hitchcock, C. L., and A. Cronquist. 1973. Flora of the Pacific Northwest. University of Washington Press, Seattle, WA. 730 pp.

HMS (Hanford Meteorological Services). 2004. Pacific Northwest National Laboratory, Richland, WA. Available online at http://hms.pnl.gov

Hooge P.N., and B. Eichenlaub. 2000. Animal movement extension to Arcview. ver. 2.0. Alaska Science Center, Biological Science Office, U.S. Geological Survey, Anchorage, AK.

Hull, A.C. 1973. Germination of range plant seeds after long periods of uncontrolled storage. Journal of Range Management 26: 198-200.

Humphrey, L.D., and E.W. Schupp. 1999. Temporal patterns of seedling emergence and early survival of Great Basin perennial plant species. Great Basin Naturalist 59: 35-49.

Johnson, T.K., and C.D. Jorgensen. 1981. Ability of desert rodents to find buried seeds. Journal of Range management 34: 312 – 314.

Johnson, M.K. 1977. Food of Townsend ground squirrels on the Arid Lands Ecology Reserve (Washington). Great Basin Naturalist 37: 128.

Jones, T.A. A viewpoint on Indian ricegrass research: Its present status and future prospects. Journal of Range Management 43: 416-420.

Jordan, G.L. 1983. Planting limitations for arid, semiarid, and salt-desert shrublands. In S.B. Monsen and N. Shaw, compilers, Managing Intermountain Rangelands: Improvement of Range and Wildlife Habitats. General Technical Report GTR-INT-157. US Forest Service, Intermountain Research Station, Ogden, UT. pp. 11-16.

Jorgensen, K.R., and R. Stevens. 2004. Seed collection, cleaning, and storage. Chapter 24, in S.B. Monsen, R. Stevens, and N.L. Shaw (compilers), Restoring Western Ranges and Wildlands, v. 3. General Technical Report RMRS-GTR-136. U.S. Forest Service, Intermountain Research Station., Ft. Collins, CO. Pp. 699-716.

Formatted: Font: 12 pt

Formatted: Normal

Formatted: Font: 12 pt

Formatted: Font: 12 pt

Formatted: Font: 12 pt

Formatted: Line spacing: single

Knapp, E.E., and K.J. Rice. 1994. Starting from seed: Genetic issues in using native grasses for restoration. Restoration and Management Notes 12: 40-45.

Kartesz, J.T.. 1999. A Synonymized Checklist and Atlas with Biological Attributes for the Vascular Flora of the United States, Canada, and Greenland. First edition. In J.T. Kartesz and C.A. Meacham, Synthesis of the North American Flora. CD-ROM version 1.0. North Carolina Botanical Garden, Chapel Hill, NC.

Kitchen, S.G., and S.B. Monsen. 1994. Germination rate and emergence success in bluebunch wheatgrass. Journal of Range Management 47: 145-150.

Knick, S.T. 1999. Requiem for a sagebrush ecosystem? Northwest Science 73: 53-57.

Formatted: Line spacing: single

Knick, S.T. , D.S. Dobkin, J.T. Rotenberry, Michael A. Schroeder, W. Mathew Vander Hagen, and C.van Riper. 2003. Teetering on the edge or too late? Conservation research issues for avifauna of sagebrush habitats. The Condor 105: 611-634.

Launchbaugh, J.L., and C.E. Owensby. 1970. Seeding rate and first-year stand relationships for six native grasses. Journal of Range Management 23: 414-417.

Linhart, Y.B. 1995. Restoration, revegetation, and the importancw of genetic and evolutionary perspectives. In B.A. Roundy, E.D. McArthur, J.S. Haley, and D.K. Mann (compilers), Proceedings: Wildland Shrub and Arid Land Restoration Symposium. General Technical Report INT-315. U.S. Forest Service, Intermountain Research Station. pp. 271-287.

Luke, F., and S.B. Monsen. 1984. Methods and costs for establishing shrubs on mined lands in southwestern Wyoming. In A.R. Tiedemann, M.E. Durant, H.C. Stutz, R. Stevens, and K.L. Johnson (compilers), Proceedings: Symposium on the Biology of *Atriplex* and Related Chenopods. General Technical Report GTR-INT-172. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 286-292.

Lunt, O.R., J. Letey, and S.B. Clark. 1973. Oxygen requirements for root growth in three species of desert shrubs. Ecology 54: 1356 – 1362.

Longland, W.S. 1995. Desert rodents in disturbed shrub communities and their effects on plant recruitment. In B.A. Roundy, E.D. McArthur, J.S. Haley, and D.K. Mann (compilers), Proceedings: Wildland Shrub and Arid Land Restoration Symposium. General Technical Report INT-315. U.S. Forest Service, Intermountain Research Station. pp. 209-215.

Mack, R.N. 1981. Invasion of *Bromus tectorum* L. into western North America: an ecological chronicle. Agro-Ecosystems 7: 145-165.

McAdoo, J.K., W.S. Longland, G.J. Cluff, and D.A. Klebenow. 1987. Use of new rangeland seedings by black-tailed jackrabbits. Journal of Range Management 40: 520-524.

McAdoo, C.C. Evans, B.A. Roundy, J.A. Young, and R.A. Evans. 1983. Influence of Heteromyd rodents on *Oryzopsis hymenoides* germination. Journal of Range Management 36: 61-64.

McCune, B., and D. Keon. 2002. Equations for potential annual direct incident radiation and heat load. Journal of Vegetation Science 13: 603-606.

McDonough, W.T., and R.O. Harniss. 1974. Effects of temperature on germination in three subspecies of big sagebrush. Journal of Range Management 27: 204-205.

McGinnies, W.J. 1970. Effects of seeding rate and row spacing on establishment and yield of crested wheatgrass. Agronomy Journal 62: 417-421.

Meyer, S.E. 1994. Germination and establishment ecology of big sagebrush: implications for community restoration. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 244-251.

Meyer, S.E., and S.B. Monsen. 1992. Big sagebrush germination patterns: Subspecies and population differences. Journal of Range Management 45: 87-93.

Millar, C.I., and W.J. Libby. 1989. Disneyland or native ecosystems: Genetics and the restorationist. Restoration and Management Notes 7: 18-24.

Monsanto, Inc. 2002. Roundup Pro Herbicide product label. Monsanto, Inc., St. Louis, MO. Available online at: http://www.cdms.net/ldat/ldo7A010.pdf

Monsen, S.B. 1994. The competitive influences of cheatgrass (*Bromus tectorum*) on site restoration. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service, Intermountain Research Station. Pp. 43-50.

Monsen, S.B., S.G. Kitchen, K. Memmott, N. Shaw, M. Pellant, S. Young, D. Ogle, L. St. John. 2003. Notice to release Anatone germplasm bluebunch wheatgrass. Unpublished paper on file at U.S. Forest Service, Rocky Mountain Research Station., Ft. Collins, CO.

Monsen, S.B., and B.Z. Richardson. 1984. Seeding shrubs with herbs on a semiarid mine site with and without topsoil. In A.R. Tiedemann, M.E. Durant, H.C. Stutz, R. Stevens, and K.L. Johnson (compilers), Proceedings: Symposium on the Biology of *Atriplex* and Related Chenopods. General Technical Report GTR-INT-172. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 298-305.

Monsen, S.B., R. Stevens, and N. Shaw. 2004. Grasses. Chapter 18, in S.B. Monsen, R. Stevens, and N.L. Shaw (compilers), Restoring Western Ranges and Wildlands, v. 2. . General Technical Report RMRS-GTR-136. U.S. Forest Service, Rocky Mountain Research Station., Ft. Collins, CO. Pp. 295 – 424.

Nelson, J.R., A.M. Wilson, and C.J. Goebel. 1970. Factors influencing broadcast seeding in bunchgrass range. Journal of Range Management 23: 163-169.

Newsome, H. 2004. Update of sagebrush restoration effort, Burned Area Emergency Rehabilitation (BAER) for the 24 Command Fire. Unpublished report. U.S. Fish and Wildlife Service, Richland, WA.

NSN (Native Seed Network). 2004a. Anatone bluebunch wheatgrass. Native Seed Network, Plant Material Profiles. Available online at http://www.nativeseednetwork.org

NSN (Native Seed Network). 2004b. Secar Snake River wheatgrass. Native Seed Network, Plant Material Profiles. Available online at http://www.nativeseednetwork.org

Ogg, A. G. 1994. A review of chemical control of downy brome. In Monsen, S.B., and S.G. Kitchen (eds.), Proceedings: Ecology and Management of Annual Rangelands. General Technical Report INT-GTR-313. U.S. Forest Service Intermountain Research Station, Ogden, UT. pp. 194-196.

Owens, M.K., and B.E. Norton. 1989. The impact of 'available area' on *Artemisia tridentata* seedling dynamics. Vegetatio 82: 155-162.

Plummer, A. P. 1943. The germination and early seedling development of twelve range grasses. Journal of the American Society of Agronomy 35: 19-34.

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. 1987. Demographic responses of *Bromus tectorum* and seedlings of *Agropyron spicatum* to grazing by small mammals: The influence of grazing frequency and plant age. Journal of Ecology. 75: 825-835.

Pyke, D. A. 1986. Demographic responses of *Bromus tectorum* and seedlings of *Agropyron spicatum* to grazing by small mammals: Occurrence and severity of grazing. Journal of Ecology. 74: 739-754.

Formatted: Font: 12 pt
Formatted: Font: 12 pt, Italic
Formatted: Font: 12 pt
Formatted: Font: 12 pt, Italic
Formatted: Font: 12 pt
Formatted: Font: 12 pt
Formatted: Font: 12 pt
Formatted: Font: 12 pt, Italic
Formatted: Font: 12 pt
Formatted: Font: 12 pt, Italic
Formatted: Font: 12 pt
Formatted: Font: 12 pt
Formatted: Font: 12 pt

Rafferty, D.L., and J. A. Young. 2002. Cheatgrass competition and establishment of desert needlegrass seedlings. Journal of Range Management 55: 70-72.

Rasmussen, J.J. 1971. Soil Survey of Benton County Area, Washington. US Department of Agriculture, Soil Conservation Service, Washington, D.C.

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

Rogers, L.E., and K.A. Gano. 1980. Townsend ground squirrel diets in the shrub-steppe of Southcentral Washington. Journal of Range Management 33: 463-465.

Schuman, G.E., D.T. Booth, and J.R. Cockrell. 1998. Cultural methods for establishing Wyoming big sagebrush on mined lands. Journal of Range Management 51: 223-230.

Smith, , D.N., H.L. Newsome, J. K. Meisel, J.R. Evans, J. Gaston, and E. Hagen. 2003. 24 Command Fire Burned Area Emergency Stabilization and Rehabilitation (BAER) Plan Final Accomplishment Report for 2000-2003 Treatments. U.S. Fish and Wildlife Service, Hanford Reach National Monument, Richland, WA. 49 pp. plus appendices.

Stahl, P.D., G.E. Schuman, S.M. Frost, and S.E. Williams. 1998. Arbuscular mycorrhizae and water stress tolerance of *Artemisia tridentata* ssp. *wyomingensis* seedlings. Soil Science Society of America Journal 62: 1309-1313

Stevens, R., and S.B. Monsen. 2004. Guidelines for restoration and rehabilitation of principal plant communities. Chapter 17, in S.B. Monsen, R. Stevens, and N.L. Shaw (compilers), Restoring Western Ranges and Wildlands, v. 1. General Technical Report RMRS-GTR-136. U.S. Forest Service Rocky Mountain Research Station., Ft. Collins, CO. Pp. 199-294.

Stewart, G. and A,C, Hull, 1949. Cheatgrass (*Bromus tectorum* L.): An ecological intruder in southern Idaho. Ecology 30: 58-74.

Uresk, D.W. 1978. Diets of the black-tailed hare in steppe vegetation. Journal of Range Management 31: 439-442.

USFWS (United States Fish and Wildlife Service). 2001. 24 Command Fire Burned Area Emergency Rehabilitation Plan Amendment. U.S. Fish and Wildlife Service, Hanford Reach National Monument, Richland, WA. 54 pp. plus appendices.

Vallentine, J.F. 1989. Range Development and Improvements, 3rd Edition. Academic Press, New York, NY. 524 pp.

Weldon, L.W., D.W. Bohmont, and H.P. Alley. 1959. The interrelation of three environmental factors affecting germination of sagebrush seed. Journal of Range Management 12: 236-238.

Formatted: Line spacing: single

Whisenant, S.G., 1999. Repairing Damaged Wildlands: A Process-Oriented, Landscape-Scale Approach. Cambridge University Press, Cambridge, UK. 312 pp.

Wilson, A.A.G. 1988. Width of firebreak that is necessary to stop grass fires: Some field experiments. Canadian Journal of Forest Research 18: 682-687.

Winkel, V. K., B.A. Roundy, and J.R. Cox. 1991. Influence of seedbed microsite charcteristics on grass seedling emergence. Journal of Range Management 44: 210-214.

Young, J.A., and R.A. Evans. 1989. Dispersal and germination of big sagebrush (*Artemisia tridentata*) seeds. Weed Science 37: 201:206.

Young, J.A., and R.A. Evans. 1985. Demography of *Bromus tectorum* in *Artemisia* communities. In J. White (ed.), The Population Structure of Vegetation. Der Junk, Dordrecht. Pp. 489-502.

Young, J.A., and R.A. Evans. 1977. Squirreltail seed germination. Journal of Range Management 30: 33-36.

Young, J.A., R.A. Evans, and G.J. Cluff. 1987. Seeding on or near the surface of seedbeds in semiarid environments. In G.W. Frasier and R. A. Evans (eds.), Proceedings of a Symposium on Seeds and Seedbed Ecology of Rangeland Plants. USDA Agricultural Research Service, Washington, D.C. Pp. 57-62.

Formatted: Line spacing: single

Young, J.A., R.A. Evans, and B.A. Roundy. 1983. Quantity and germinability of *Oryzopsis hymenoides* seeds in Lahontan sands. Journal of Range Management 36: 82-86.

Zar, J.H. 1996. Biostatistical Analysis. Prentice Hall. New Jersey. 718 pp.

PERSONAL COMMUNICATIONS

Amaranthus, M. Mycologist, President, Mycorrhizal Applications, Inc. Grants Pass, OR

Benson, J. Restoration ecologist and consultant. Moses Lake, WA.

Downs, J.L. Senior Research Scientist, Plant Ecology, Pacific Northwest National Laboratory, Richland, WA.

Herman, R. L & H Seeds Inc., Connell, WA.

Larsen, D. Regional Biologist, Washington Department of Fish and Wildlife, Kennewick, WA.

Meisel, J. Biological Technician, Hanford Reach National Monument, Richland, WA.

Appendices

APPENDIX A. Study plot descriptions and locations

This appendix contains detailed information regarding permanent vegetation and density plots used by The Nature Conservancy on the Fitzner-Eberhardt Arid Lands Ecology (ALE) Reserve, Hanford Reach National Monument from 2001-2004. Information provided includes general location and directions to plots, plot identification markings, and details of plot layout. Plots are treated in the following order:

I. Biodiversity Plots
II. Steppe-in-Time Plots
III. Hanford Biological Resources Management Plan (BRMaP) Plots
IV. Transition Density Plots
V. Rehabilitation Plots
VI. Sagebrush survival plots

UTM coordinates (UTM Zone 11, NAD27)for all plots are listed at the end of each section. Locations of permanent plot types I through V are indicated on a 1:30,000 scale map (accompanied by supporting GIS data) addendum to this report.

I. Biodiversity Plots

A. Plot Identification & Setup

In 2001, 5 m x 20 m plots were marked with ½" x 30" rebar at each corner. Eight-12" inches of rebar was left exposed above the ground's surface. A circular aluminum tag inscribed with the letters "DW" and the plot number was attached to a rebar at one corner.

Cheatgrass density was recorded in 20cm x 20cm microplots arranged along parallel 50m transects laid out beginning along the long (20m) axis of each 5m x 20m plot. Three microplots were located at random whole-meter points within each 10m segment of the parallel transects, for a total of 30 microplots /plot (Fig A1). Microplot locations are presented in Table A2.

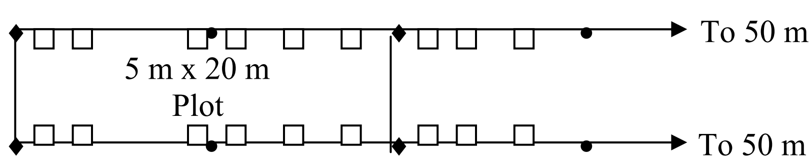


Fig. A1. Example of placement of 20 x 20 cm microplots (☐) for cheatgrass (*Bromus tectorum*) density measurements in Biodiversity Plots. Tick marks represent 5 m intervals.

B. Directions to Individual Plots. GPS coordinates are presented in Table A2. Vegetation descriptions are from 2001, unless otherwise noted.

Plot 33

- Just 3.5 m west of road at base of Yakima Ridge. Plot begins on west edge of shallow draw (which road crosses). If you cross an old road, you've gone too far. 1.84 km N along road from junction at Rattlesnake Springs. Area burned in 1998 as well as in 2000.

Plot 34

- Just east (50-60 m) of road running along base of Yakima Ridge, 3.8 km from road junction at Rattlesnake Springs. In shallow draw. Area burned in 1998 as well as in 2000.
- Park where firm, graveled portion of road ends. From this point plot is in first (very) shallow draw encountered, just walking west a few feet down the sandy part of the road.

Plot 35

- In ARTR refugia upslope from plot 34. Not burned in 2000.
- Refugia is visible on the left (south) and uphill from end of gravel portion of Rattlesnake Springs-to-Gate 120 powerline road, close to plot DW34.
- Park in same location as for plot DW34, 3.8 km from road junction at Rattlesnake Springs and end of gravel portion of road
- Proceed south from road. In summer it's easier going through the sagebrush than through the cheatgrass-choked and hummocky wash. Plot is about 520 m south of road approaching west end of refugia.

Plot 36

- Located ~ 700m south of SR 24 in the NW portion of ALE. From Richland follow SR 240 to Yakima Barricade/Junction with SR 24. Turn west on 24. Find safe place to park along left (ALE) side of road before passing mile post 7. Shoulder is high, steep, and soft so recommend not do this in vehicle other than high clearance 4WD.
- Cross fence and go south about 700m from SR 24, passing sandy "barrens [in 2001]' and ARTR refugia, down into one little drainage and up and over a low E<->W convexity, finally dropping down to the Cold Creek flats. Plot DW36 is on the flats just beyond the edge of the last low ridge.

Plot 37

- Heavily burned old ARTR stand on flat bench above upper Cold Creek.
- Follow Gate 120 BPA powerline rd. to vicinity of 2 large elk exclosures. Plot is about 1.35 km north of rd.

Plot 38

- 250 m SE of plot DW36, roughly out in the middle of broad flats of Cold Creek drainage.
- A highly impacted site dominated by BRTE. Many ARTR skeletons and scantier remains of ARTR stands.
- Crust damaged, degraded, for the most part missing.

Plot 39

- Formerly an old, open ARTR stand before the fire. In mosaic burn area. A single old ARTR in plot survived the fire.
- Follow Gate 120 BPA powerline rd. west from Cold Creek Valley up low slopes of Yakima Ridge. Park about 400m past elk exclosures (see plot DW37) on south side of rd.
- Plot is 725m south and slightly upslope of road.

Plot 42

- Directions: Plot is an easy 10 minute walk 250m SE of gate 111 Rd across convexity to the NW facing slope (i.e., the far slope) at the 1st draw SE at the road. Park about 1.0 km down slope from the intersection at the Gate 111 Rd and Benton REA wooden powerline rd. DW 51 and 53 may also be accessed from this parking spot.

Plot 43

- Closest access is from intersection at Gate 111 Rd and Power Line Rd or nearby. From intersection head south on Power Line Rd about 100 m and cross 1st draw. Follow the far slope at the draw (NW-facing) down the draw about 300 m to vicinity of plot.
- Alternatively can park a little farther down Gate 111 Rd and travel about 250m SE from rd., over low ridge separating road from 1st draw, down into draw, and up onto facing slope.

Plot 44

- On broad ridge top in middle of gentle slopes between 1200' Road and Cold Creek Valley. Just east of old dozer line or track.
- An area of moderate to severe fire (2000) esp. around EULA clumps. EULA resprouting where scattered, appears to have largely died back in this area where it was somewhat dense.
- Directions: From Gate 111 Rd. just upslope of intersection w/ wooden powerline rd., go 1.1 km NE, crossing two large draws. Plot is on ridge top 90-100m SE of old dozer track (if you cross the dozer track you've gone too far) and about 80m NW of a charred orange fiberglass BWIP marker.

Plot 45

- ~200 feet up steep slope near lower Snively Springs.
- Park at road closure at spring and ascend hill directly in front of you (to NW). A red fencepost with bright red painted top marks the origin of a transition plot (T-10) at the base of this hill.
- Plot is ½ to 2/3 way up this slope, and just above an old road or dozer track which contours along the slope ~ 1/2 way up.

Plot 46

- About 1.0 km East of 1200' Road, on gentle slopes of lower Rattlesnake Mt., on same convexity as transition plots T5 and T6.
- Plot is ~60 m from T5/10 (1000m) stake of transition plot T5.
- General area is excellent AGSP-POSA, but little crust.

Plot 47

- On low gentle slopes ~2.5 km down hill from 1200' road, elevation ~650'. Best access is from Gate 112 (wooden powerline) Rd.
- Plot is one draw and < 500m away from DW50 to the SE. T5-T6 complex is to the NW.

Plot 50

- On low gentle slopes approaching Cold Creek Valley, ~3 km downslope from 1200' Rd. On lower 1/3 of a broad bench running down to Cold Creek. Best access is from Gate 112 (wooden powerline) Rd.
- Plot is one draw and < 500m away from DW47.

Plot 51

- About 500 m NW from Gate 111 Rd, downslope from where road crosses wooden power line, elev ~750' ASL. Parking same as for DW42, 53.

Plot 52

- On low, broad, gentle slopes just above Cold Creek Valley. Access is overland from Gate 109 quarry.
- Plot is approx. 1.4 km NW of the quarry (where vehicle can be parked), and beyond the first major draw.
- Elev. ~650'

Plot 53

- Near foot of gentle slopes, between 1200' Road and Cold Creek Valley.
- About 40' uphill from large draw near mouth of draw as it empties out into Cold Creek Valley. On NNE side of bench about 1.5 km SE of Gate 111 Rd. Can also be accessed via little used but passable track from gate in Cold Creek valley

Plot 54

- On low gentle slopes just above Cold Creek Valley, elev. ~650'
- 2.1 km (approx.) from gate 109 gravel pit where vehicle can be parked.
- On gentle convexity above 2nd major draw to the NW of gravel pit.
- Directions: From DW55 head NW across a wide, deep wash to the crest of the next ridge (~ 550m distance).

Plot 55

- On low gentle slopes, just above Cold Creek Valley, elev ~650'.
- About 1.7 km across these slopes NW of Gate 109 gravel pit, on gentle ridge above and before crossing 2nd major draw NW of gravel pit.
- 340-350m NW of and across a shallow wash from DW52.

Plot 56

- 0.57 km from Gate 117 Road about ¼ of the way down from 1200" Road junction towards Benson Ranch.
- Cross 2 deep and narrow washes and another shallow one, then cross a broad, major wash (full of cheat).
- Plot is on bench just west of this broad wash.
- Vehicle parked on 117 Rd is easily seen from plot.

Plot 58

- ~1 km north of road in upper Snively Basin
- Degraded plant community with lots of BRTE and SIAL.
- Natural community type was ARTP/STCO. Both severely impacted bu recovering.

Plot 59

- On high steep slope above flat where plot DW58 is located
- Beautiful native plant community, BRTE not quite absent
- FEID scattered, plentiful, but AGSP dominant. Nearly all bunchgrass recovering. Patch of *Senecio integerrimus* slightly upslope of plot
- Slope so steep and soils so fragile surveyors cause damage, best not to visit too often

Plot 61

- Located in saddle in hills 660m west of lower Snively Springs. Park at or near road closure at spring.

DW 71

- From 117 rd ~ ½ mile below jct. w/ 1200' Rd., route travels SE 750-800m crossing 3 major draws, a forth smaller draw and 1-2 shallow depressions.

Plot 73

- Sand dune area along SR 240, Located along low slope of first long, high dune crossed by SR 240 heading North from Richland. Located between milepost 9 and ALE Gate 114. Park along roadside near milepost 9 and ascend dune to plot. From here locate 1st white fiberglass highway marker towards Richland side at dune and proceed SW (220 ° magnetic) to plot.
- Note: mileposts are numbered beginning at junction with SR 24 at Yakima Barricade and increase towards Richland.

Plot 74

- ~300' down slope from unpaved end of Rattlesnake Mtn. Rd
- Park where road "circles" last set of radio towers at north (NW) end of summit area. Road begins to turn eastward and head down hill about 3400'
- Drop down from road, heading N or NE, steeply at first, then onto a gently sloping bench. Cross bench. Plot is located just after you cross the lip at the bench and drop onto steeper slopes below.

Plot 75

- Vegetationally diverse area along upper slopes of Rattlesnake Mt, NW of main summit area
- From "curve" of road described in plot 74 (but before road begins to lose elevation) traverse flat area NW, then drop down along side of steep slope to rocky area.
- Lots of diversity in all classes of vegetation, esp. forbs.

Plot 78

- Lower end of high basin on upper slopes @ SE end of high ridge of Rattlesnake Mtn.
- From Rattlesnake Summit Rd drop down through basin where FEID is prominent on North and NW slopes of draws. ARTP occasional.
- Plot is near lower lip of this basin, just east of a rock-rib outcrop, which practically forms the south boundary of the plot.
- Plot is on W or NW-facing slope of a draw.
- Look directly up draw to radio tower on east end of Rattlesnake Summit Ridge.

Plot 79

- On gentle slope ~1.0 km along lower slopes from Rattlesnake Mt. Rd, on very gentle slope just before steeper terrain.
- Park ~ where road begins to climb more steeply. Cross old ditch as you approach plot.
- Extensive AGSP-POSA community in very good condition. Community continues up slope to paved road (where it is now heading north) and down slope about ½ way (?) to E-W power line where dense BRTE-SIAL is encountered.

Plot 88

- On moderate slope 200' below Rattlesnake Mt Road in area approaching summit.
- Park about where powerline, heading up from the East or SE, ends along the high ridge, after it passes the summit facilities.
- A stand of shrubby (*Prunus sp.*) cherries, prominent even from the 1200' road far below, lies about 100' below and somewhat NE of plot.

Plot 89

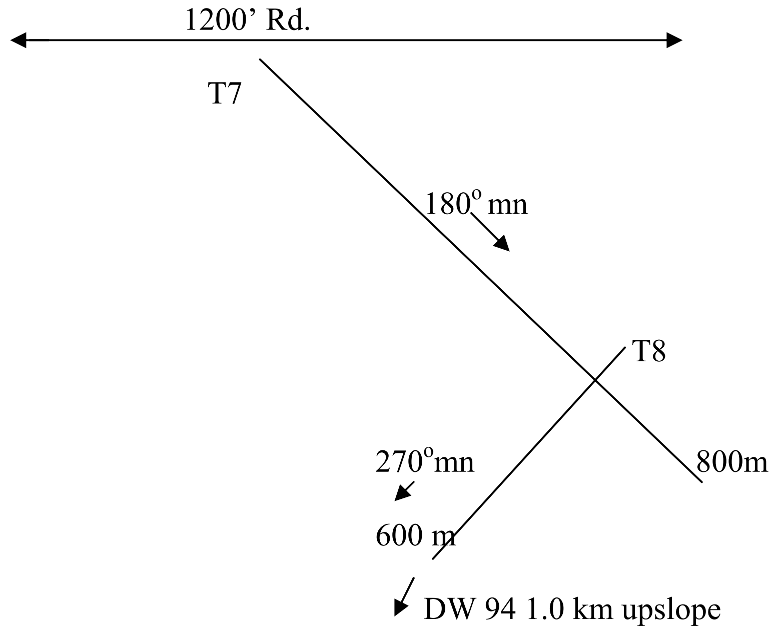
- NW from plot DW88 along upper slopes at Rattlesnake Mt. and ~300 below plot DW88.
- Steep east-facing slope with many large rocks exposed. Forb-rich community.
- From plot 88, proceed NW past *Prunus* patch, dropping down as you traverse.

Plot 93

- Plot is about 0.5 km upslope from 1200' Rd on gradual, middle slopes along foot of Rattlesnake Mtn. AGSP-POSA community, in good condition. Forb-rich community. BRTE scarce in area around plot, but common-dense in surrounding washes and downslope.

Plot 94

- ARTP-grassland on intermediate slope at foot of first steep slope at Rattlesnake Mtn.
- Plot is about 1.0 km upslope from the west end of transition plot T-8 (Stake marked T8/6 or 600 meters from T8 origin). Can also be approached from BRMaP 8-PC5.
- *Artemesia tripartita* resprouting abundantly, with some TECA on slope 100 m to 150 m east of plot.
- Pattern of alien dominance in draws and washes very evident from this vantage point looking down to lower gentler slopes between here and 1200' road.



Plot 95

- Only ~100 m north of the 1200' road in "Sagebrush Triangle," area of severe fire between Rds 117 and 118
- Just west of a major draw, perhaps the biggest draw the road traverses in this area, and near the NW end of "ARTR Triangle," shortly before the road drops down towards Snively flats.
- Park in old "borrow" pit at the head of the draw, uphill side of road
- Also in this area: Steppe-in-Time plot #14 ('Sage Flats'), a pair of N-S transects (200m each) about 0.5 km downslope.
- This area of plot burned very severely in 2000 fire-ARTR burned right to ground. Bunchgrasses (AGSP) recovering very well, however.
- Plot is within area planted w/ bare root ARTR in Dec. 2001. 6 ARTR BRs were in plot in 2002 but no live ARTR persisted in plot into 2003.

Plot 96

- Located in area of intense fire in "Sagebrush Triangle" between Rds 117 and 118. 0.5 km downhill from 1200' Rd and NE of plot 95 and steppe-in-time #14.
- Plot is located just 50 m north of the terminus at BRMP 24/ PC2 vegetation transect.
- This area VERY heavily burned. ARTR burned to ground and even below in some cases. Even POSA reduced by fire here.

Plot 102

- In Iowa Flats area, area of greatly rolling terrain north of Gate 106 Road between high voltage power line and sharp bend in the road.
- Plot is ~300 m from road.

Plot 105

- AGSP site at 2100', 1.75 km overland from plot 109 (see Route, next page).
- Area is heavily infested with BRTE, with dense strands of SIAL nearby.
- DW mentions BRTE and SIAL in area, but his plot was fairly pristine; invasives have increased greatly since 1994.

Plot 106

- STCO stand in beautiful basin above (1780') Benson/Bobcat Springs. 1.3 km downhill from plot 105. Lots of *Stipa,* fair amount of ARTP, even a *Carex*
- Unfortunately, basin is home to dense BRTE and SIAL, and all native vegetation heavily infested. Upper part of basin is +/- solid aliens
- Accessed from DW105 (see Route, next page). Route from here to 1200' road heads up and over large summit NE of this basin, then drops down to skirt upper part of Bobcat Canyon (elk trails are helpful here) and heads for Bobcat/ Rd., visible on hills to SE. From Bobcat Rd. is easy walk downhill to 1200' road

Plot 107

- ARTP/AGSP –FEID community in good condition in steep slopes just below crest of Rattlesnake Ridge
- Most of plot surroundings are convexities burned moderately in 2000 fire and are nearly free of alien species. But areas where ARTP were clustered burned very hot—charred ground surrounds these areas and crust mostly destroyed. BRTE and SIAL are common in these spots.
- Many native grasses and forbs (and resprouting ARTP, CHNA, CHVI) are abundant in these patches and in the draws, which are in much better condition than low elevation draws—though BRTE and SIAL are usually present.
- Access: Accessed easily via the access road for microwave towers at west end of Rattlesnake Ridge. This road is on private land and permission must be granted by the following gentlemen: Steve Harrison (SH) 973-3455 and Bud Hamilton (BH) 973-2327 or cell 781-0500.
- Access road begins on SH's property, but closest approach to plot 107 (and BRMaP 13, and Transition-Density plot T16) is on BH's property.
- From Benton City, take Hazel Rd/Old Inland Empire Hwy to District Line Rd to Hanks Rd. to Case Rd to Pearl Rd. SH lives at junction with Pearl Road.
- Alternatively, this cluster of plots can be accessed from far end of Rattlesbnake Mt. summit rd. (followed all the way to its end, see directions to DW 108) and walking BH's road along the fenceline to T16-6. Permission should still be sought from Mr. Hamilton.

Plot 108

- NE slope of Rattlesnake Mountain, circa 3400'.
- Drive Rattlesnake Mt. Rd. to near its end (along outside of fence, but Greg Hughes says this is still ALE) almost to property line and wheatfields just below microwave towers. Travel up moderate slope to ridge crest, over and down ~ 200'.
- In 2001 a lone wooden "telephone" pole-with no utilities attached- stood on the ridge crest ~150' east of where we crossed the ridge to drop down to plot 108.
- Plot runs across a slope convexity, with a more mesic community in the wash.
- Broad bands of talus with *Huechera, Agastache* and even some *Woodsia* occur upslope of plot. Slope convexity to the north has several basalt outcrops.

Plot 109

- Plot located about 1.0 km from grove of planted trees at Snively Ranch site. On slope overlooking Snively basin and old ag fields.
- Area is good bunchgrass (AGSP) along top of ridge, with few or no shrubs. North-facing slope of ridge has considerable ARTP and FEID
- Burned in '81 or '84 according to DW (prior to 2000).
- Ridge in good condition around plot, but within 0.5 km or less dense BRTE and SIAL are encountered.

<u>Route</u> (from DW109 to DW105 to DW106):

- We visited plots 109,105 and 106 consecutively on a long day, parking at the Snively homestead grove and traversing to the Bobcat Rd. just east of Bobcat Springs, then following this road down to 1200' road where another vehicle was spotted.
- Much of the way from 0.4 to 0.5 km from 109 was <u>choked</u> by BRTE and SIAL—both species knee high and higher in dense stands with few or no native components. SIAL stands so tall and dense, difficult to walk through. Evidence of BRTE mats in these SIAL stands, from pre-2000 fire. Route from 105 to 106 is more of same
- Occasionally very small patches in good condition, but those discontinuous and at limited extent-usually contain some BRTE too.

Plot 111

Access same as for #108. 111 is on south side of ridge, just ~100' upslope from fence and 'road.' 24 command fire burned up to crest of ridge on the north side, leaving this side of ridge unburned in this area.

Table A1. GPS coordinates for Biodiversity Plots used in vegetation monitoring, 2001 – 2004. All coordinates recorded using a Garmin *etrex* portable GPS unit, datum NAD27.

Plot #	UTM N	UTM E	Corners	ID	Plot #	UTM N	UTM E	Corners	ID
33	5155592	292464	Rebar	Tag	59	5146568	288805	Rebar	no tag
34	5157525	292030	Rebar	Tag	61	5149064	290360	Rebar	Tag
35	5157364	291491	Rebar	Tag	71	5149143	297148	Rebar	Tag
36	5160599	288825	Rebar	Tag	73	5151586	300955	Rebar	Tag
37	5159245	288526	Rebar	Tag	74	5142716	299395	Rebar	Tag
38	5160404	288986	Rebar	Tag	75	5142860	298955	Rebar	Tag
39	5157138	288780	Rebar	Tag	78	5139752	302104	Rebar	Tag
42	5144506	306012	Rebar	Tag	79	5138717	304291	Rebar	Tag
43	5143992	305438	Rebar	Tag	88	5141238	300452	Rebar	Tag
44	5144911	304429	Rebar	Tag	89	5141603	300399	Rebar	Tag
45	5148324	291072	Rebar	Tag	93	5143592	301724	Rebar	Tag
46	5145571	301903	Rebar	Tag	94	5143565	299803	Rebar	Tag
47	5146292	303584	Rebar	Tag	95	5149612	293490	Rebar	Tag
50	5146239	304156	Rebar	Tag	96	5149944	294204	Rebar	Tag
51	5145072	305740	Rebar	Tag	102	5138834	310186	Rebar	Tag
52	5140944	310495	Rebar	Tag	105	5147114	294037	Rebar	Tag
53	5144574	307527	Rebar	Tag	106	5146431	295063	Rebar	Tag
54	5141650	309989	Rebar	Tag	107	5144336	295852	Rebar	Tag
55	5141177	310265	Rebar	Tag	108	5143620	297890	Rebar	Tag
56	5150006	295880	Rebar	Tag	109	5147357	292273	Rebar	Tag
58	5146500	289550	Rebar	Tag	111	5143446	297671	Rebar	Tag

Table A2. Microplot locations (meters from origin) used for measurements of cheatgrass density (and, in 2004, percent cover of plant litter and rodent mound and burrow presence-absence) along paired 50 m transects, Biodiversity Plots.

Plot	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15
36	1	6	8	15	17	19	21	26	28	34	36	39	41	43	47
38	1	4	6	11	17	19	21	25	28	32	34	36	41	45	48
42	3	7	9	11	15	18	22	26	29	33	35	38	41	44	46
43	4	7	9	11	15	19	22	26	28	33	35	39	41	44	47
44	1	5	9	12	14	16	21	24	28	32	36	39	43	45	48
45	2	6	9	12	15	17	22	24	28	34	36	40	42	45	47
46	2	6	9	14	16	18	21	24	30	35	37	39	41	43	49
47	5	7	9	12	14	19	23	25	27	31	33	39	44	46	48
50	2	4	7	11	13	17	26	28	30	32	34	39	44	47	49
51	3	6	/8	12	14	19	23	26	30	33	36	38	43	47	49
52	2	7	9	13	15	20	25	27	29	31	33	39	41	46	49
53	3	5	8	11	16	19	22	26	28	33	35	39	44	47	50
54	1	4	9	12	14	19	22	26	28	31	35	37	43	45	49
55	1	6	8	13	16	18	22	27	29	32	35	37	42	48	50
56	3	5	9	15	17	20	24	26	29	31	35	37	43	46	49
58	4	6	8	11	14	18	25	27	29	31	34	38	42	44	49
61	2	5	7	12	14	17	21	24	29	31	33	37	43	47	49
71	1	3	6	11	13	20	22	24	28	31	37	39	43	45	50
73	3	6	8	12	17	19	23	25	28	33	35	37	42	45	47
74	1	5	7	13	18	20	22	24	26	32	36	39	43	45	50
75	1	3	8	14	16	18	21	25	28	32	35	40	42	45	49
78	2	5	7	13	15	17	21	23	30	34	38	40	45	47	49
79	1	6	8	11	15	17	22	24	26	33	36	40	44	46	49
88	5	7	10	13	16	19	21	23	27	32	34	39	43	46	50
89	3	5	8	13	15	20	25	27	29	31	34	36	46	48	50
93	2	4	9	14	16	19	24	27	29	33	35	38	42	45	47
94	3	6	8	11	15	19	21	23	26	34	37	39	41	43	45
95	1	7	9	11	13	17	22	26	28	32	34	38	43	48	50
96	2	5	9	11	17	19	21	24	26	31	34	37	42	45	48
102	3	5	9	14	16	18	22	27	30	33	36	39	42	45	47
107	2	4	7	11	15	18	21	28	30	33	35	38	43	45	49
108	5	7	10	13	16	19	21	23	27	32	34	39	43	46	50
109	2	4	10	12	16	19	21	23	26	34	36	38	43	45	48

II. Steppe-in-Time Plots

A. Transect and microplot locations

Transects are marked with a steel fencepost or pipe at the origin, i.e., the south end of the transect. There are a few exceptions to this. The fenceposts for sites 66E and 66W are on the north ends of the transects – the ends closest to the access road. The site number and name of the transect, (and E or W if it is one of a pair) is written on the fencepost. Transect distances are measured from south to north regardless of which end the fencepost is on. Yellow-painted wooden stakes (original and renewed periodically through 2001) are located at 50m intervals and marked with their position in meters along the transect. ½" Rebar stakes are also installed at 50, 100, 150, and 200 m points. Rebar were driven in close to the ground and capped with orange plastic safety caps. Caps were frequently found removed (by wildlife?) and found lying some distance away.

Microplots are read via 0.1m 'Daubenmire' frames along the transect at 10m intervals. The frames are always placed along the right-hand side of the transect (as viewed from the origin) with the long (50cm) axis of the microplot oriented perpendicular to the transect, the short axis directly on the line, and the centerpoint of the short axis centered on the meter point sample location (Fig. A2). Cheatgrass density was recorded only within the 20 cm x 20 cm segment of the microplot closest to the transect line. When plots were first installed, 2" galvanized roofing nails were placed in the ground at opposite corners of microplot locations. In addition to the roofing nails,L-shaped ¼" pencil rod were inserted at the 80m and 180m microplot locations (fixed photo locations). Where these markers can be relocated microplots can be placed relatively precisely in their original locations. Because of changing vegetation (especially the disappearance of large shrubs from several of the transects) some of the original locations do not line up precisely with the appropriate meter mark along the tape. The heavier photopoint markers are nearly always relocatable. However, many of the original roofing nails have been ejected by frost heaving or animal activity, or obscured by vegetation, so that finding both nails in place is no longer common. Sometimes a single nail will be found. Where no usable placement markers are available, the default microplot location used by this study is the meter point itself, located along the straightest transect line possible.

B. Directions to Individual Plots. GPS coordinates (UTM NAD27) are presented in Table A3. Directions were compiled in the pre-GPS era; navigation with GPS instruments is much simpler. All compass bearings are MAGNETIC bearings. The South end of the transect is the reference point for geographic coordinates. Vegetation descriptions are from Dr. Michael Marsh and are based on observations made prior to the 24 Command Fire. Site names were created by Dr. Marsh.

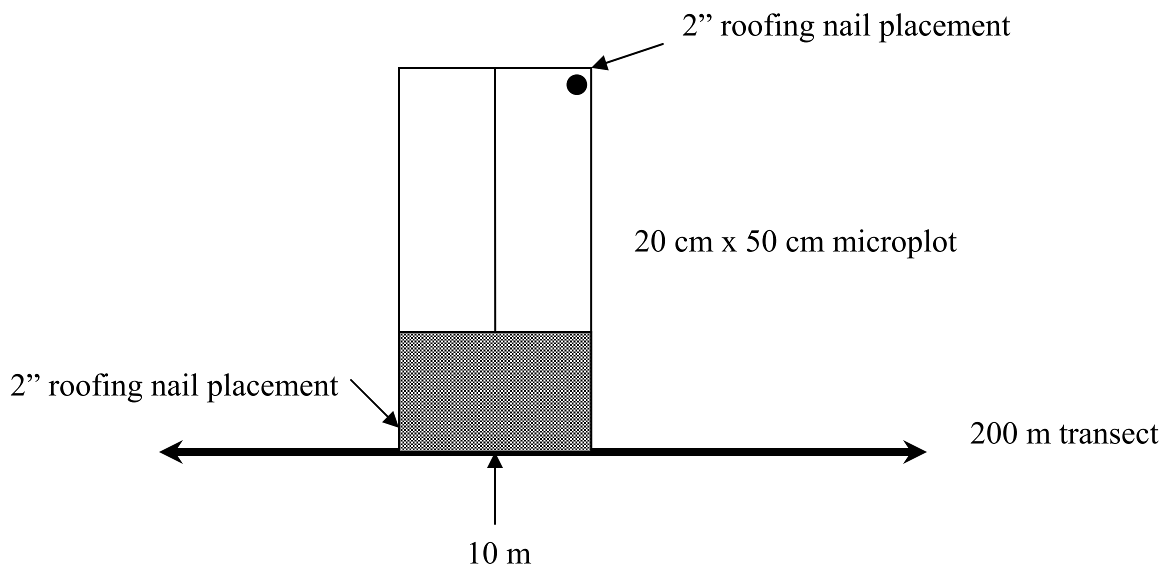


Fig. A2. Microplot placement along Steppe-in-Time transects. Cheatgrass density was recorded within the 20cm x 20cm (shaded) segment of the microplot nearest to the transect line.

SIT 14 ("Sage Flats"): T 12, R 25, Sec. 33 SE,
bordering T 11, R 25, Sec. 4 NE.
Lat 46 28.542', Long. 119 41.33, Elev. 800'
Established 1992.

Plots 14E and W are in an area of old growth Wyoming big sagebrush prior to the 2000 wildfire. Enter ALE at Gate 117 and proceed up 117 Rd. to its intersection w/ the 1200' Rd. Turn right onto the 1200' Rd. and continue another 2 miles with the former sagebrush area on your right. Off road parking is available in a borrow pit where the road crosses a wash. On your right, NE of the road, you will see an unusual east-west oriented ridge. Take one of the valleys that go down to this ridge and around its east end. Navigate with GPS, OR follow this valley 850m Northwest (stay in the valley - don't cross a ridge), to a fallen barbed wire fence crossing the valley. One 6"x 6" fencepost stands on the ridge to the east. Follow the fence west to where deep old wheel tracks cross it on top of the ridge on your west. The South Stake of the West Transect (a steel fencepost) is 72m from the wheel tracks on a bearing of 280^{o}. It is on near the top of a low ridge near its north end, and is marked with a tall stake. The West Transect crosses a gully, goes over a low ridge, then crosses a second gully to end on a higher ridge to the east of a high, steep South-facing slope covered with cheatgrass. The south stake (a steel fencepost) of the East Transect is 101 m from the 6"x 6" fencepost on the ridge east of the valley you followed, on a compass heading from the post of 107^{o}. The East Transect runs along the West slope of a valley, then down into and across its bottom at its north end.

Pre-fire description: Pre-fire herbaceous cover mostly POSA, with cryptogams and bare ground in rather dense big sagebrush on the South end of the East Transect while it is brome grass and tarweed on the northern end. The West Transect, similarly, has less disturbed ground near its south end.

SIT 31 ("Elk Slope"): T 12, R 24, Sec 35, SE
Lat 46 28.857', Long 119 46.663', Elev. 1000'. Established 1992.

On ALE. Follow directions to SIT 14 and then to SIT 66 and SIT 67. The East Transect is about 700 m bearing 330^{o} from the roadside marker stake for SIT 67 (which is approximately due south of the West Transect of SIT 67). Its south end is marked by a steel fencepost in the fork of two draws. The East Transect crosses the east one of these gullies and climbs a low hill. The South Stake of West Transect is 300 m West across two draws, on a broad plain or slope. Its south end is marked with a steel fencepost.

Pre-fire description: Sagebrush burned off (probably in 1984) West Transect prior to 2000 wildfire, but charred wood still present. Some large sagebrush still exists in south 100M of East Transect.

SIT 47 ("1200 ft. Road"): T 11, R 26, Sec 18 SE
Lat 46 26.281', Long 119 36.381', Elev. 1100'. Established 1992.

Accessed from 1200' Rd. via Gate 106 Rd. and through Gate 107 past the Nike site. Both transects actually cross the road and one microplot of the West Transect lies in the gravel of the road, while one microplot of the East Transect falls on the weedy roadside berm. The south end of each transect is marked by a steel fencepost south of the Road. The south stake of the West Transect is in the fork of a shallow draw, and the 50 m stake of this transect is also south of the road. Both transects cross the road, and they lie 300m apart from each other. Southwest of the West Transect there is a prominent pile of basalt rocks.

Pre-fire description: On the gentle slope north of Rattlesnake Mt., with stony outcrops, a fairly flat slope broken by 6 ft deep gullies. No sagebrush on this site. Cheatgrass is still fairly abundant because of early grazing history before WW II, but native forbs are moderately abundant.

SIT 48 ("Upper Bluebird"): T 11, R 25, Sec 14, SW,
46 deg. 26.10' Lat., 119 deg. 39.00' Long.;
2080'; Dir slope 6 (w); Slope 5 deg. Established 1997.

The single transect is ½ to ¾ mile SW of SIT 326. Located in a small basin open to the NW, overlooking Bobcat Canyon in the distance. Easiest access is from 1200' Rd. via Bobcat Rd. to 1950 – 2000' contour, then overland SE into shallow high basin. Can also be accessed from SIT #326 via a steep climb, crossing windswept lithosol-type communities at crest of basin. Best to do these two plots on the same overland trip.

0m stake is a steel pipe. Once painted white, it has been wind-scoured such that plot name & number are mostly illegible.

Pre-fire description: Abundant low shrub cover, both big- and three-tipped sagebrush; good meadow.

SIT 60 (“ALE Lab”): T 10, R 26, Sec 3, NE and 2, NW
Lat 23.025'. Long 119 32.145, Elev. 1300'
Established 1992.

The plots lie south of the ALE Laboratory/ Nike site buildings. Park at the ‘Garden Bldg.’ A wooden powerline runs south between the E and W transects and is a useful route for returning th a parked vehicle. The site lies between the 6th and 7th power poles (beginning with the first of a series running in a straight line South) south of the Lab. The south end of the West Transect is about 100M West of the Power Line, and is marked with a steel fencepost. The south end of the East Transect is also marked by a steel fencepost. The south stake (fencepost) and the 50 m stake of the East Transect are south of a shallow draw.

Pre-fire description: The West Transect appears to be the most heavily impacted (by past grazing) of our Arid Land Ecology Reserve sites, but SIT 14 is actually at least as poor, looking better only because it has sagebrush. Site slopes very gently northeast. There is no sagebrush here.

SIT 66 (“Shooting Star Hill”): T 11, R 24, Sec 1 SE
Lat 46 27.91', Long 119 45.319' (N end W tran), Elev. 1400'
Established 1992.

Approach lower Snively Spring via the Gate 118 Rd. and turn right where a road fords the spring (a muddy ford) and heads towards the northern boundary of ALE. After about a mile the road crosses a gully where a steeper slope is on your left and you face a gently rising ridge where the road turns right, out of the gully. The ridge is actually an isolated hill, and the north end of the West Transect, marked with a steel fencepost (not visible from the road), is near its south end. From where the road crosses the gully bottom, the steel fencepost, and perhaps the old tall North stake for the East Transect is visible with binoculars on a compass bearing of 200^o (SSW) on a steep slope at a distance of 1/4 to 1/2 mile. It is about 150m up from the gully bottom west of it. The West Transect is 300m west. It starts out by crossing a rocky creek bed (S end).

Description. Floristically poor near the creekbed at the south end of West Transect, the vegetation improves to the north, and the East Tranect is in very good condition. The only sagebrush here is *A. tripartita*, low-growing on the steepest north slopes.

SIT 67 ("Rattlesnake Ridge"): T 12, R 24, Sec 35 SE and T 11, R 24, Sec 2, NE
Lat 46 28.62', Long 119 46.318', Elev. 1200'.
Established 1992.

NOT on Rattlesnake Mountain but across road north of SIT 66 and slightly south of, and an easy walk from, SIT 31E. From the road the West Transect of this site is 450 paces north of a roadside stake, over the knoll. The East Transect is 300 m to the east of the West Transect. It crosses two shallow draws before climbing a hill with scattered sagebrush. The South ends of both transects are marked with steel fenceposts, and they are 300 m apart.

Pre-fire description: The transects are on gently north-facing slopes, with charred sagebrush stumps only on the West Transect, but some sagebrush bushes persist on the East Transect. The floristic composition is quite good.

SIT 188 ("Microwave Hill"): T 11, R 25, east margin of Sec 23 SE.
46 deg. 24.43 Lat, 119 deg, 38.35 Long. Dir slope 0 (N), slope 20 deg.
Established 1996.

The single transect is located just inside the ALE boundary fence next to the microwave towers at the west end of Rattlesnake Mt. Plot is unique in that it is arranged in the shape of an "L". The 1st leg begins at the fencepost origin and runs W (magnetic) to 50m stake. Transect then turns 90^{o} and runs to magnetic north the rest of the way. This 50m stake is 444cm from the corner fencepost near the west guy wire at the base of the upper microwave tower.

Access could be as for DW 107, from Bennett Road north, between Rothrock Rd., Crosby Rd., continuing on towards microwave towers. Less roundabout access is is via the Rattlesnake Mt. Rd. as for DW108 and DW111. Follow this road to where it ends at the fence and microwave towers. Follow the fence on the ALE side around the corner where SIT 188 fencepost will be obvious. Or walk along the ridgetop from DW108 to the fence corner. Best to do all plots in this area in one day.

Pre-fire description. Rich flora. Just north of ridge crest (ridge sloping northwest). Plot begins in lithosol, then turns downhill into deeper soil and a different, more lush flora including *Hieracium cynoglossoides*.

SIT 326 (“ Lower Bluebird”): T 11, R 25, Sec.14 NE.
46 deg., 26.54' Lat, 119 deg, 38.59' Long. 1,600' (top).
Direction of Slope 0 (north) Slope 30 deg.
Established 1996.

0m stake is a steel pipe on knoll just east of the head of a large draw. Once painted white, the stake has been wind-scoured such that plot name & number are mostly illegible. Knoll is the 3rd knoll east of the old Bobcat Rd. Road can be seen from the plot. A small low basin lies between these knolls and the next steep rise of the Rattelesnake Hills. Easiest access to here is via the Bobcat Rd. from the 1200’ Rd. Where the road bends sharply to the S to SW, cross low basin on approximately the 1600’ contour to the plot. Can also come down over the ridge from SIT #48 (see above); best to do these two plots on same day, one long walk. Transect begins in lithosols, steeply descends a north-facing slope to less exposed soils, crosses a rocky draw and ends on a weedy south-facing slope across the draw.

Pre-fire description. BACA, AGSP, HAST, Lupine, MACA, POSA, BRTE in swales. COGR, LOTR, FEID. [2004: Soil crust community appeared very well-developed over 1st 100-150m of transect, esp. on less exposed areas below the beginning of the transect.]

Table A3. GPS coordinates for Steppe-in-Time plots used in vegetation monitoring, 2001 – 2002. All coordinates recorded using a Garmin *etrex* portable GPS unit, datum NAD27.

Site #	UTM N	UTM E	Observation Point
14 E	5150186	293824	Fencepost - South end of East Transect
14 W	5150301	293548	Fencepost - South end of West Transect
31E	5150935	287156	Fencepost - South end of East Transect
31W	5151036	286871	Fencepost - South end of West Transect
47 E	5145705	300115	Fencepost - South end of East Transect
47 W	5145807	299827	Fencepost - South end of West Transect
48	5145575	296416	Steel pipe - South end of Transect
60 E	5139496	305370	Fencepost - South end of East Transect
60 W	5139608	305096	Fencepost - South end of West Transect
66 E	5148944	288742	Wooden stake - South (top) end of East Transect
66 E	5149121	288813	Fencepost - North end of East Transect
66 W	5149217	288526	Fencepost - North end of West Transect
66 W	5149030	288466	Wooden stake - South end of West Transect
67E	5150486	287574	Fencepost - South end of East Transect
67W	5150574	287290	Fencepost - South end of West Transect
188	5144137	297277	Fencepost - South end of Transect
326	5146417	297066	Steel pipe - South end of Transect

III. Hanford Biological Resources Management Plan (BRMaP) Plots.

<u>A. Plot Identification and Microplot Placement</u>

These 20 ha macroplots are marked with steel fenceposts at each corner (Fig A3). BRMaPs 7, 8, 12, and 13 have an additional fencepost (road stake) marking the macroplot central axis at the nearest road approach. Five point-count stations (which serve as vegetation transect origins) marked with 4' fiberglass wands run down the center of each macroplot at 200 m intervals. Point-count stations also typically are more or less encircled by four wire semicircles (exact purpose unknown) whose pointed ends are inserted into the ground about 6" from the point-count station. Point-count stations on BRMaP 7, 8, 20, and 24 are labeled with the plot name and station number (e.g., 'BRMaP 20 / PC 1') on cardboard-backed aluminum strip tags. Stations on other plots are not labeled.

Vegetation transects are marked with fiberglass wands at the origin (0 m, the point count station) and at 100 m, and by ½" rebar at 0, 50, and 100 m. Microplot locations (at five meter intervals along vegetation transects) are marked with wire flags. The 20cm x 50cm microplot is always placed with its long axis along the transect line, facing back towards the origin (Fig. A3). Cheatgrass density was recorded only within the 20cm x 20cm segment of the microplot closest to the meter mark. Microplots are read either to the right or left of the transect line, depending on the orientation of the particular transect within the plot. Whether microplots are to be read on the right or left side of the transect line is indicated on the original 1996 field data forms (supplied by PNNL). Microplot orientation is also indicated on the plot diagrams that accompany the plot descriptions that follow.

Some fiberglass wands were damaged or destroyed during the 24 Command Fire. Wands are also popular rubbing posts for elk and perhaps deer in velvet and may be displaced in the process. Most damaged wands associated with vegetation transects have been replaced by PNNL or TNC personnel, but others may or may not be in place. Wire semicircles described in paragraph 1 above are an aid in relocating stations where wands have been displaced or destroyed. Vinyl flags marking microplot locations were melted during the 24 command fire but many flag wires remained in place. New flags were inserted alongside the old wires in 2001 and subsequent years. Where no wire could be found, new flags were placed at the appropriate interval along the transect.

<u>B. Directions to individual plots</u>. The following section describes plot locations and transect orientations. GPS coordinates are presented in Table A4. BRMaP plots are all located in proximity to existing roads or tracks, and UTM coordinates are precise, so that field location should require only a functioning GPS unit. A map of plot locations is helpful.

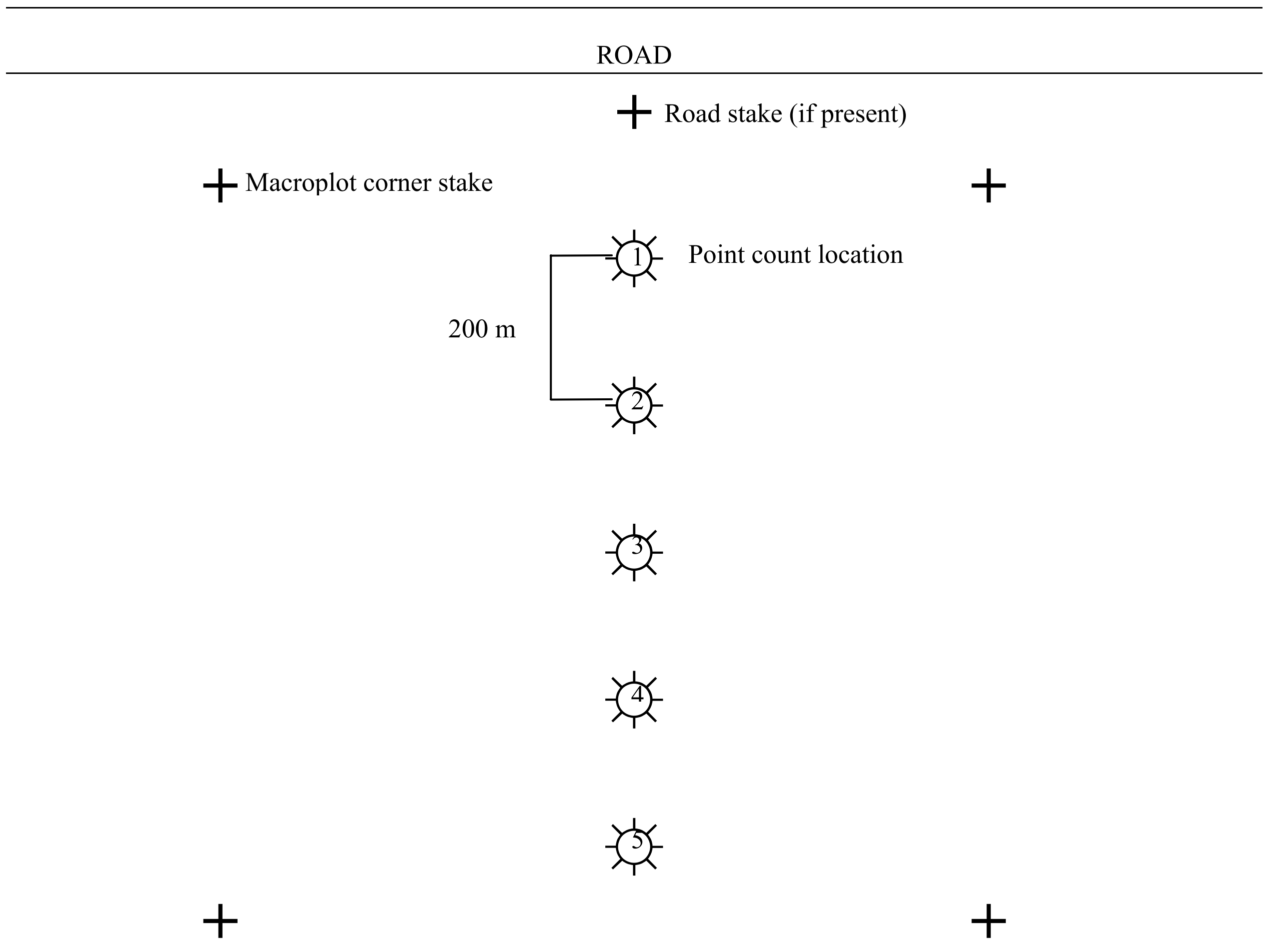


Fig. A3. General layout for BRMaP macroplots. Symbol + represents steel "T" fenceposts used for road marker (where present) and macroplot corner stakes. Symbol ☼ represents point count locations.

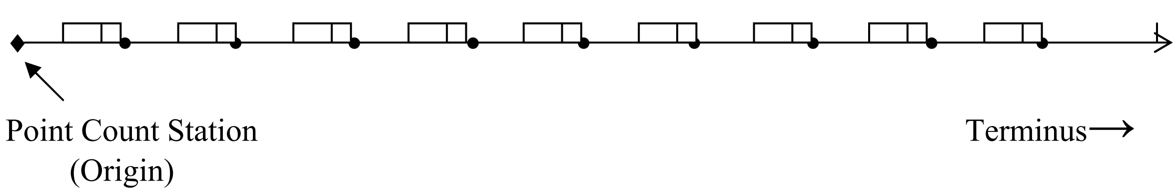


Fig. A4. Location of 20 x 20 cm cheatgrass density microplots (small squares) within 20 x 50 cm vegetation microplots (larger rectangles) along BRMaP vegetation transects. Tick marks represent 5 m intervals.

BRMaP 7

Plot is located on gentle slopes north of Iowa Flats and the Nike Site. The road stake is on the downslope side of the 1200 Foot Rd. 2.6 km west of Gate 107. Vegetation transects are associated with PCs 1, 3, and 5. PCs 3 and 5, farther downslope, are in better condition than PC1. All microplots are read on the uphill side of the transect.

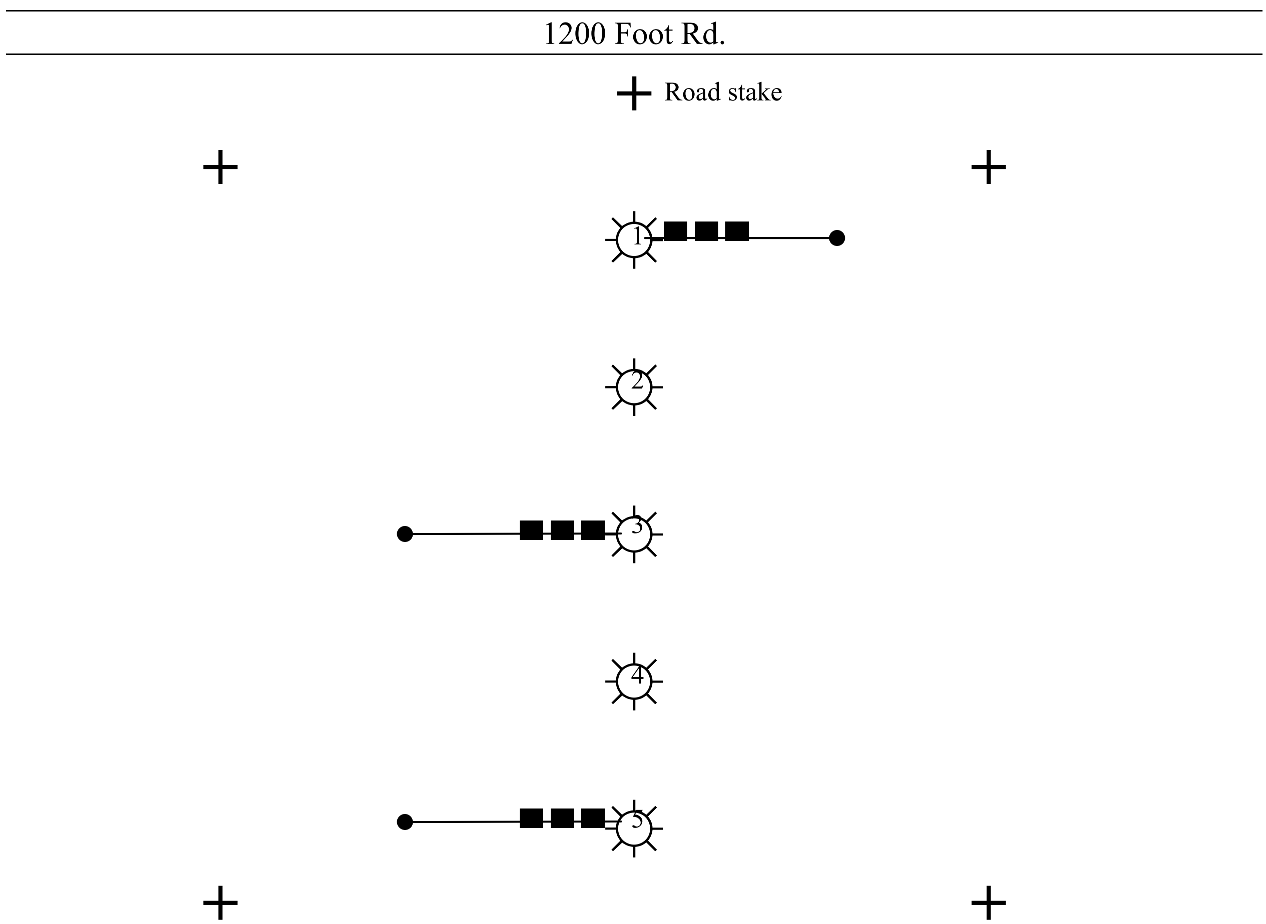


Fig. A5. General layout of BRMaP 7. Plus symbol (+) represents steel “T” fenceposts used for road marker (where present) and macroplot corner stakes. Sun symbol (☼) represents point count locations. Lines (–•) indicate direction of 100m vegetation transects. Blocks on lines (▀) indicate orientation of 20 cm x 50 cm microplots along transect lines.

BRMaP 8
Located on gentle mid-slopes of Rattlesnake Mt. Road stake is on the uphill side of 1200' Rd. 6.6 km west of Gate 107. Vegetation transects are associated with PCs 1, 3, and 5. All microplots are read on the <u>downhill</u> side of the transect.

The area just above the road (incl. PC1 transect) is the site of pre-fire cheatgrass mat. Very weedy, dominated by SIAL, BRTE, other aliens & very few natives. Dense alien community extends nearly to PC2 where SIAL disappears & BRTE diminishes. Both still dominate in washes. The area beyond PC 2, including PC3, PC5 and the surrounding area, is generally in very good condition, except for alien dominance in washes.

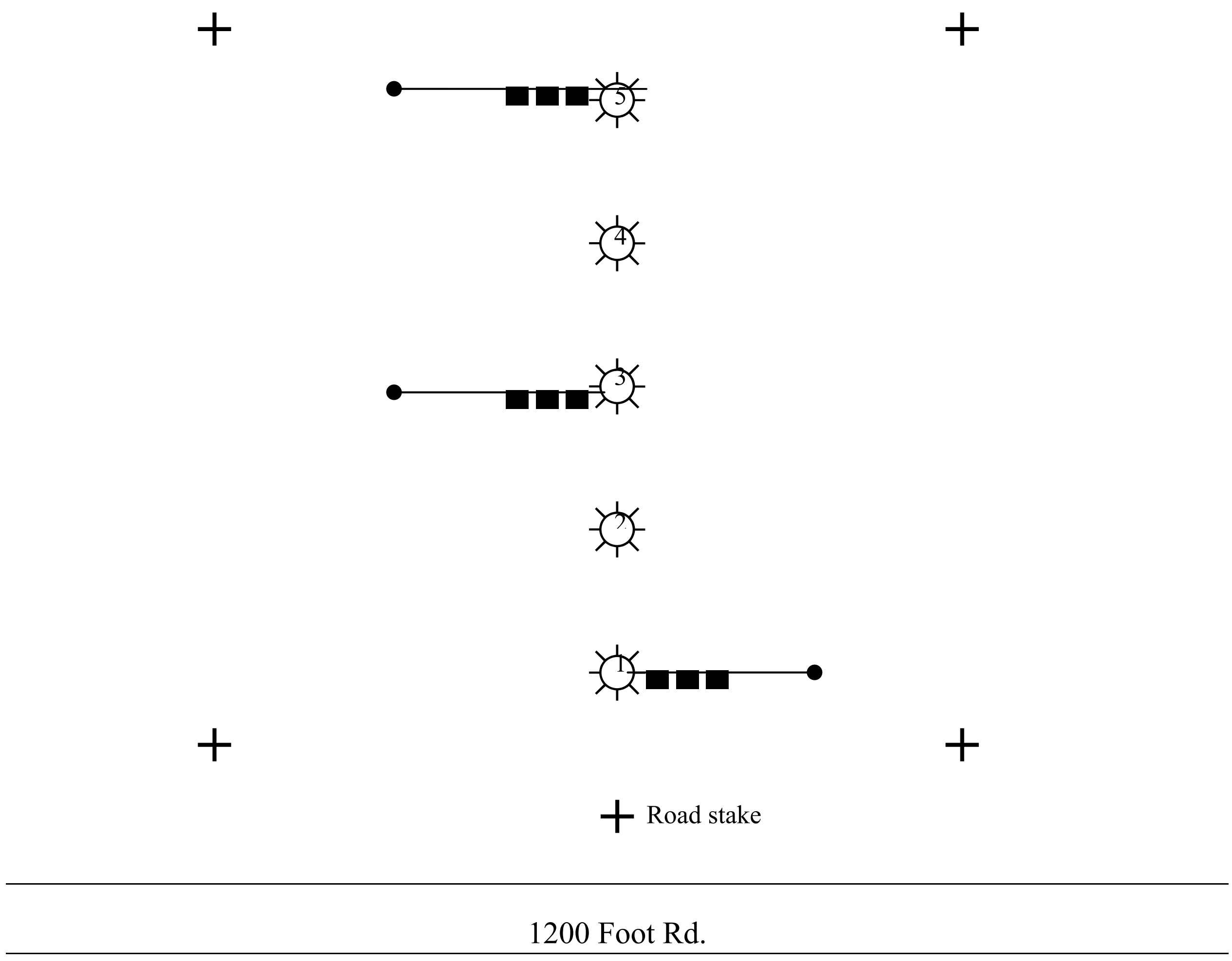


Fig. A6. General layout of BRMaP 8. Plus symbol (+) represents steel "T" fenceposts used for road marker (where present) and macroplot corner stakes. Sun symbol (☼) represents point count locations. Lines (–•) indicate direction of 100m vegetation transects. Blocks on lines (▀) indicate orientation of 20 cm x 50 cm microplots along transect lines.

BRMaP 12
In Rattlesnake Hills above upper Snively Basin, near ALE boundary. Access via Snively Basin Rd. Drive past Lower Snively Spring, Snively Ranch, and old rye fields. Park at base of steep switchback and continue up road on foot. Road stake is ¼ to ½ mile beyond top end of switchback, where road crests and turns west again. Macroplot is laid out on a westerly (~250^o mn) bearing from the road stake. PC1 is 200m west, uphill and across a draw that parallels the road. Vegetation transects are associated with PCs 1, 3, and 5. All microplots are read to the <u>west</u> side of the transects.

PC1 is ARTP/AGSP community type in good condition on deep soils, gentle slope. PC3 is located on a steep N-facing slope on shallow, rocky soil. ARTP/POSA w/ a number of lithosol-type plants. Excellent cond. Few aliens of any kind. Old dozer line approaches transect closely but even this is mostly weed free. PC5 lies across deep draw (w/ riparian vegetation) from PC3. Transect is on rolling terrain and intercepts a couple of shallow, weedy draws. ARTP/AGSP w/ some STCO; BRTE fairly abundant, dominant w/ SIAL in draws. Wheat fields visible up large draw & at crest of hill to west.

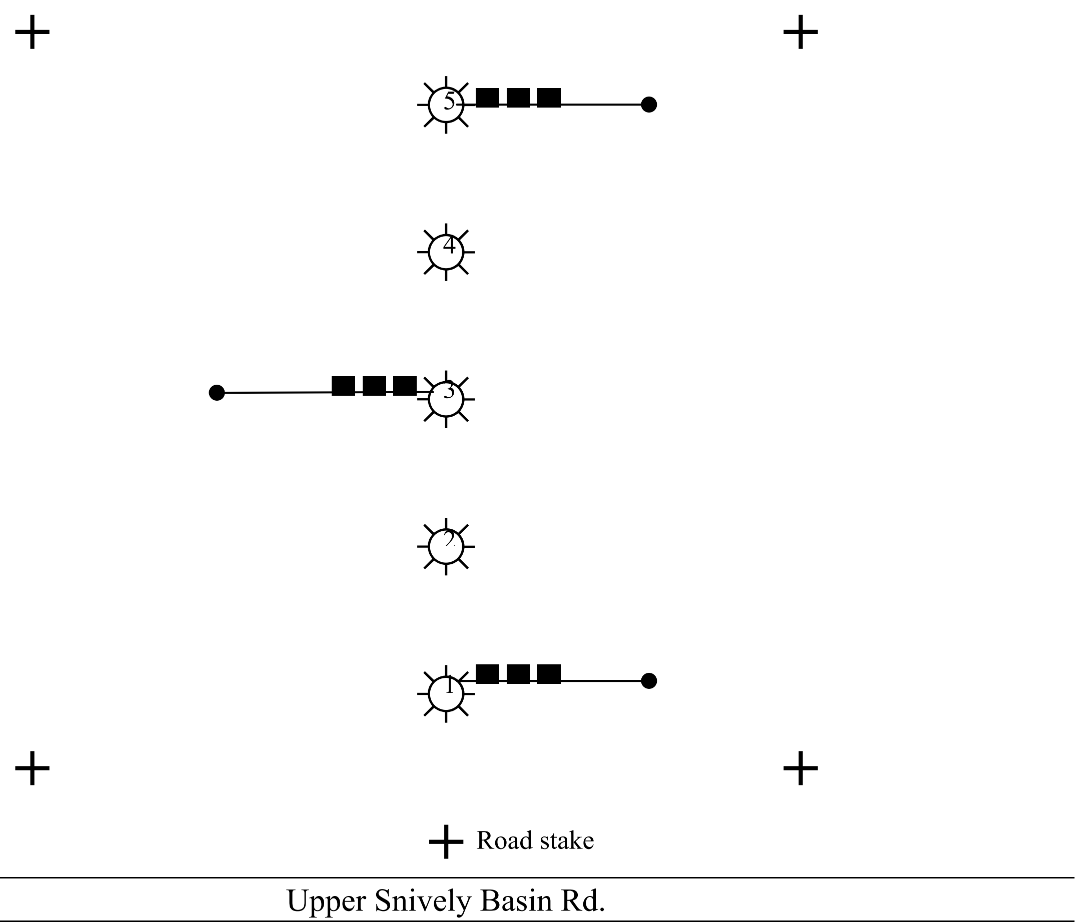


Fig. A7. General layout of BRMaP 12. Plus symbol (+) represents steel "T" fenceposts used for road marker (where present) and macroplot corner stakes. Sun symbol (☼) represents point count locations. Lines (–•) indicate direction of 100m vegetation transects. Blocks on lines (▀) indicate orientation of 20 cm x 50 cm microplots along transect lines.

BRMaP 13
In rolling Rattlesnake Hills far above Bobcat Spring. PC5 is only 100m downslope from access road to microwave towers and boundary fence. See DW 107 for driving directions. Access is also possible via upper end of Bobcat Rd, where there is a road stake. Do not attempt to DRIVE Bobcat Rd.! For greatest efficiency, survey BRMaP 13, DW 107, and T16 complex on same trip. All microplots are read to the east or southeast side of the transects.

PC-5. Along boundary fence a rebar stake (tagged T16/6) is just on the ALE side of the fence. From stake walk exactly 100m on a bearing 360° magnetic to PC5, 0m (fiberglass wand). Transect continues due N from here gently at first then steeply downhill into a little saddle.
PC-3. 400m west of PC5. A very heterogeneous transect, runs across little rocky ridge w/ lithosol spp., then through a lush swale & finally uphill through more typical ARTP/AGSP community.
PC-1. Less heterogeneous than PC3, but passes through lush terrain w/ regenerating ARTP and a stand of ELCI.

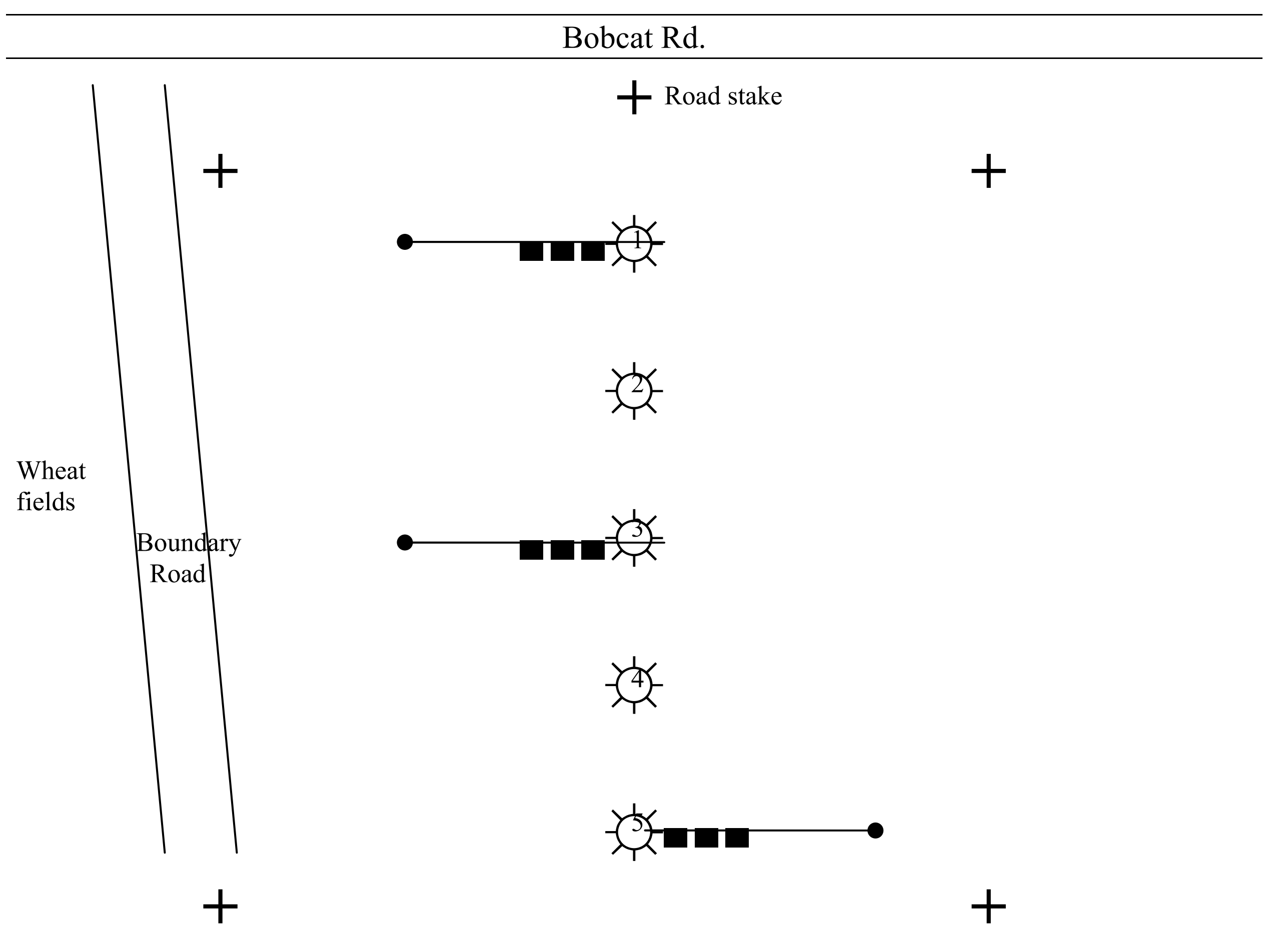


Fig. A8. General layout of BRMaP 13. Plus symbol (+) represents steel “T” fenceposts used for road marker (where present) and macroplot corner stakes. Sun symbol (☼) represents point count locations. Lines (–•) indicate direction of 100m vegetation transects. Blocks on lines (■) indicate orientation of 20 cm x 50 cm microplots along transect lines.

BRMaP 20

Macroplot is located roughly parallel to Gate 118 Rd in the Cold Creek Valley. Plot is on E side of Rd. PC5 is 100-150 m to N side of brushy cutoff rd. which jogs around Rattlesnake Springs – no rd. stake however. PC1 is at the N end of the macroplot, closest to SR 240. All microplots are read to the west or southwest side of the transects (facing away from SR 240).

This area of sandy soils supported a dense stand of ARTR prior to 2000 and burned severely in the fire. Vegetation is diverse. Area is full of old fenceposts & littered w/ irrigation equipment along w/ other old agricultural and scientific debris. Vegetation transects are associated with all 5 point count locations.

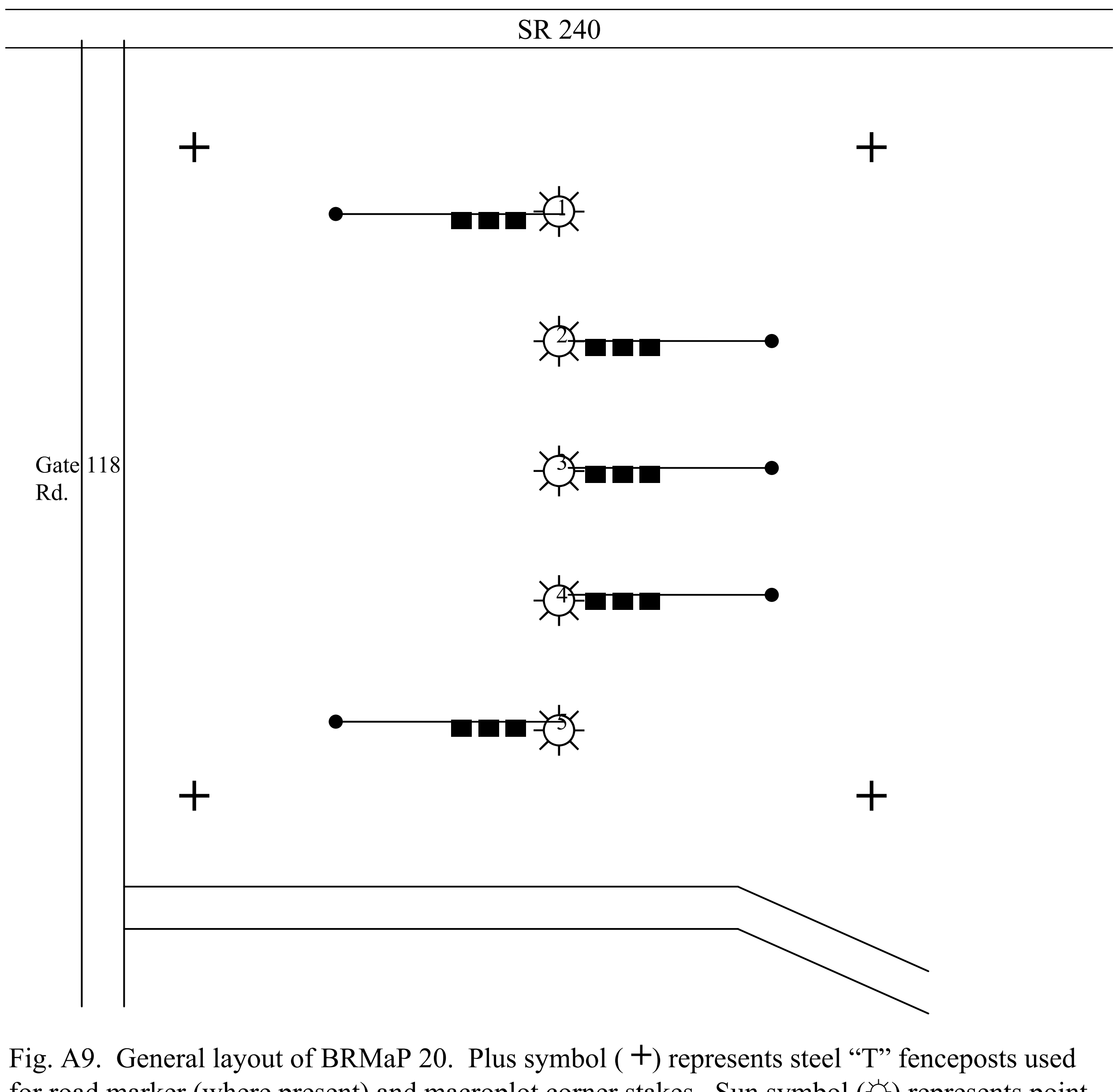


Fig. A9. General layout of BRMaP 20. Plus symbol (+) represents steel "T" fenceposts used for road marker (where present) and macroplot corner stakes. Sun symbol (☼) represents point count locations. Lines (–•) indicate direction of 100m vegetation transects. Blocks on lines (■) indicate orientation of 20 cm x 50 cm microplots along transect lines.

BRMaP 23
Macroplot is oriented roughly parallel to the Rattlesnake Springs-to-Gate 120 road, which runs along NE flank of Yakima Ridge. The macroplot area burned in 1998, burned only patchily in 2000. Originally (1996) vegetation transects were associated with all 5 point count stations; however, only PCs 1, 3, and 5 could be relocated in 2001. PC5 is the southern-most point count marker, located only about 13 m off west side of the road. The vegetation transect for PC5 runs approx. SW (220°). PC3 is in the center of the macroplot, with a somewhat large sagebrush refugia to the west. The terminus of the 100m PC3 vegetation transect is right on the edge of the road. PC1 is close to the 4th pair of high-tension pylons on the Gate 120 power line. All microplots are read to the south side of the transects.

Several ARTR refugia are in this general area. Only one is within the macroplot, and it is not intercepted by a vegetation transect. Area at the macroplot is intensely disturbed. Vegetation is dominated by BRTE and SAKA. Further to NE in Cold Creek veg is dominated by SIAL and *Descurainia* spp. With CADR (white top) in places.

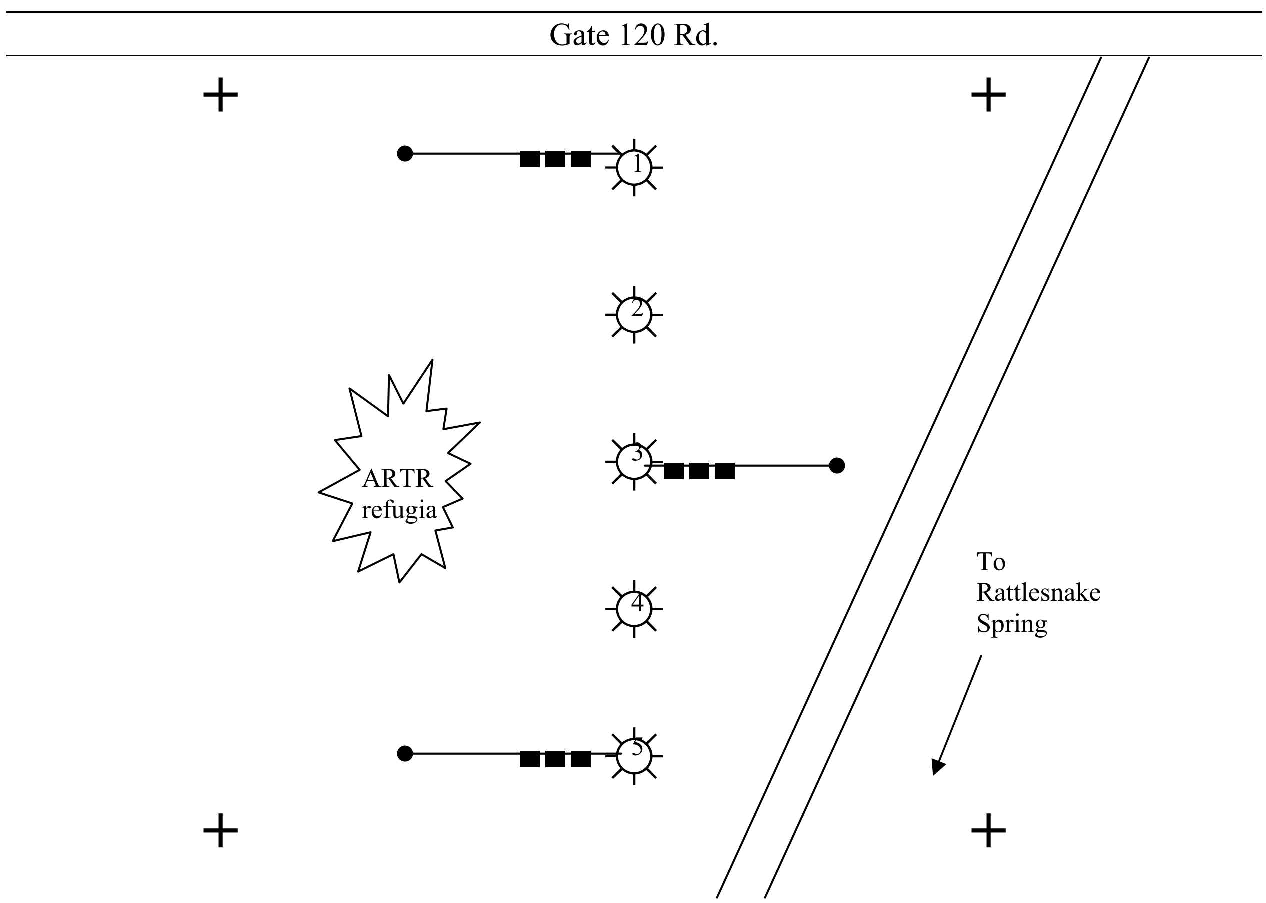


Fig. A10. General layout of BRMaP 23. Plus symbol (+) represents steel "T" fenceposts used for road marker (where present) and macroplot corner stakes. Sun symbol (☼) represents point count locations. Lines (–•) indicate direction of 100m vegetation transects. Blocks on lines (■) indicate orientation of 20 cm x 50 cm microplots along transect lines.

BRMaP 24
Plot is located downslope from the 1200 Foot Rd., 1 to 2 miles west of its junction with the Gate 117 Rd. The main axis of the plot (PC1 to PC5) runs roughly parallel to the 1200 Foot Rd, about 300 m downslope. Vegetation transects (associated with all 5 point count stations) run perpendicular to the main axis, i.e. towards or away from road. PC1 is at the NW end of the plot, while PC5 is at the SE end. All microplots are read to the east or southeast side of the transects.

The terrain of the area is composed of relatively narrow ridges dissected by deep draws. One draw in particular is very broad (near PC1). PC5, 4, and 2 stay up on the ridges, but PC3 dips into a shallow draw then rises again, while PC1 transect dips into and ends in the broad draw.

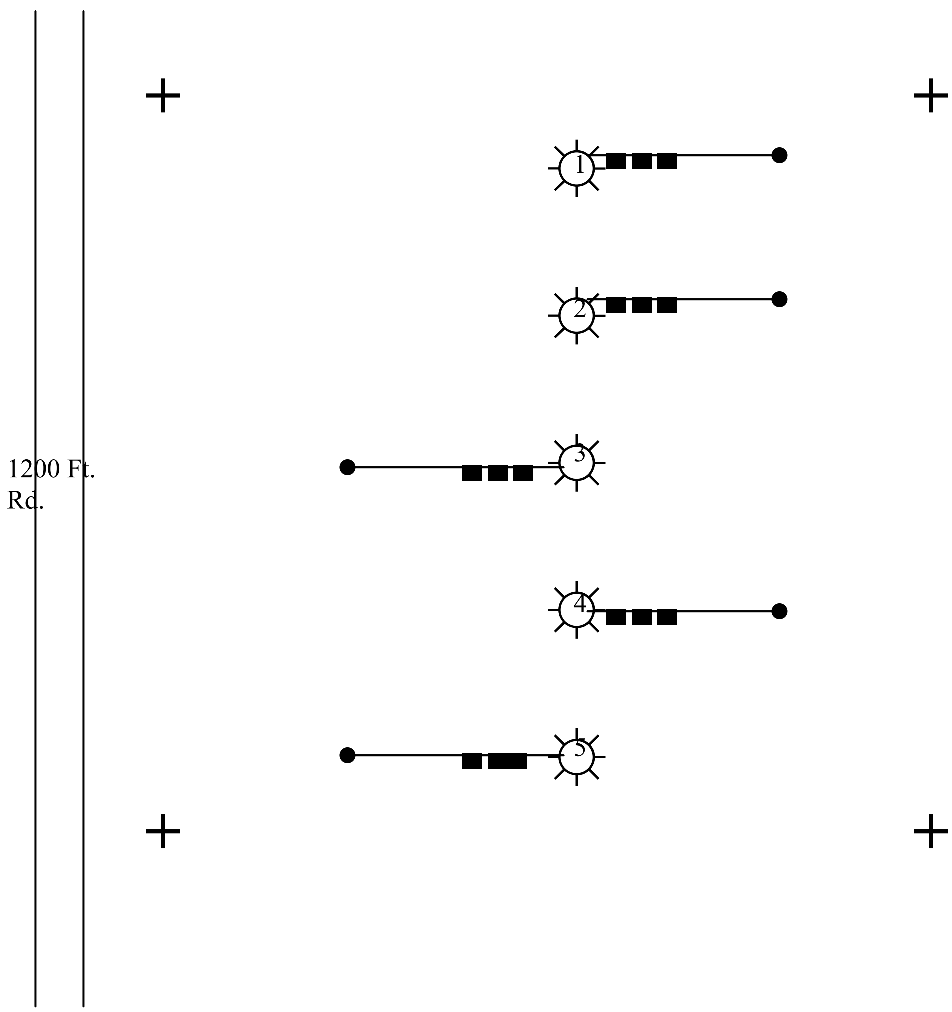


Fig. A11. General layout of BRMaP 24. Plus symbol (+) represents steel "T" fenceposts used for road marker (where present) and macroplot corner stakes. Sun symbol (☼) represents point count locations. Lines (–•) indicate direction of 100m vegetation transects. Blocks on lines (■) indicate orientation of 20 cm x 50 cm microplots along transect lines.

Table A4. GPS coordinates for BRMaP plots used in vegetation monitoring on the ALE Reserve, 2001 – 2004. Coordinates courtesy of Pacific Northwest National Laboratory, datum NAD27.

Plot	Point	UTM N	UTM E	Plot	Point	UTM N	UTM E
BRMaP7	ROAD	5142318.5	303660.53	**BRMaP8**	ROAD	5145173.5	300867.22
	NW	5142458	303669.47		SE	5145037.5	300869.47
	SW	5142309	303803.19		NE	5145172.5	300724.16
	PC1	5142449.5	303810.88		PC1	5145035	300722.06
	PC2	5142582	303961.53		PC2	5144900	300580.53
	PC3	5142721.5	304110.25		PC3	5144753.5	300445.09
	PC4	5142855	304255.69		PC4	5144621.5	300289.56
	PC5	5142993.5	304400.75		PC5	5144482.5	300150.56
	SE	5142991	304545.09		SW	5144342	300150.53
	NE	5143138.5	304410.88		NW	5144479.5	300005.06
BRMaP12	ROAD	5144868.5	289668.22	**BRMaP13**	ROAD	5144624	294991.31
	NE	5144976.5	289586.78		SW	5144500	295052.56
	SE	5144783	289571		NW	5144689.5	295118.22
	PC1	5144886.5	289474.28		PC1	5144564	295179
	PC2	5144900	289278.03		PC2	5144507.5	295371.72
	PC3	5144921.5	289076.59		PC3	5144435	295553.47
	PC4	5144926.5	288878.34		PC4	5144365.5	295737.94
	PC5	5144959	288684.28		PC5	5144309.5	295926.22
	SW	5144864	288576		SE	5144196.5	295988.13
	NW	5145063.5	288592.25		NE	5144372	296047.84
BRMaP20	SE	5154329.5	295009.25	**BRMaP23**	NW	5158992	291277.25
	NE	5154497	294906.38		NE	5159100.5	291445.38
	PC1	5154359.5	294869.91		PC1	5158968	291404.16
	PC2	5154266.5	294700.53		PC2	5158784.5	291488.34
	PC3	5154158.5	294524.03		PC3	5158605	291589.34
	PC4	5154058	294351.19		PC4	5158427	291678
	PC5	5153960.5	294178.47		PC5	5158245.5	291767.59
	SW	5153822.5	294148.38		SE	5158210	291908.31
	NW	5153995.5	294042.63		SW	5158108.5	291737.25
BRMaP24	NW	5150038.5	293912.06				
	SW	5149855	293829.28				
	PC1	5149894	293955.41				
	PC2	5149822.5	294144.84				
	PC3	5149735	294335.06				
	PC4	5149643.5	294503.78				
	PC5	5149554	294680.13				
	NE	5149596	294815.97				
	SE	5149413.5	294720.41				

Table A5. GPS coordinates for BRMaP plots on the Hanford Reach National Monument outside of the ALE Reserve. Coordinates courtesy of Pacific Northwest National Laboratory, datum NAD27.

Plot	Point	UTM N	UTM E	Management Unit
BRMaP 18	ROAD	289581.2	5164480.0	McGee - Riverlands
	NW	289457.9	5164410.0	
	NE	289639.4	5164339.0	
	PC1	289512.7	5164292.0	
	PC2	289430.1	5164104.0	
	PC3	289353.3	5163920.0	
	PC4	289286.4	5163724.5	
	PC5	289209.6	5163543.5	
	SW	289082.8	5163488.5	
	SE	289265.9	5163409.5	
BRMaP 29	NW	286083.0	5165645.0	McGee - Riverlands
	NE	286658.0	5165779.0	
	PC-A	286212.0	5165569.0	
	PC-B	286410.0	5165611.0	
	PC-C	286603.0	5165661.0	
	SW	286133.0	5165453.0	
	SE	286717.0	5165593.0	
BRMaP 21	ROAD	303190.1	5179065.0	Saddle Mountain
	NE	303309.1	5179006.5	
	NW	303131.4	5178922.0	
	PC1	303259.2	5178878.0	
	PC2	303363.2	5178708.5	
	PC3	303465.9	5178535.5	
	PC4	303564.4	5178362.5	
	PC5	303661.4	5178185.5	
	SW	303621.8	5178049.5	
	SE	303799.2	5178135.5	
BRMaP 14	ROAD	314899.9	5174032.0	Wahluke
	NE	314798.0	5174129.5	
	NW	313799.9	5174082.5	
	PC1	314699.1	5174024.5	
	PC2	314500.4	5174010.5	
	PC3	314297.4	5174003.0	
	PC4	314095.1	5173996.0	
	PC5	313896.3	5173984.0	
	SW	313805.2	5173889.0	
	SE	314804.5	5173928.0	

Continued

Table A5 (Continued). GPS coordinates for BRMaP plots on the Hanford Reach National Monument outside of the ALE Reserve. Coordinates courtesy of Pacific Northwest National Laboratory, datum NAD27.

Plot	Point	UTM N	UTM E	Management Unit
BRMaP 16	ROAD	315157.3	5177633.0	Wahluke
	NE	316130.9	5177121.5	
	NW	315288.2	5177664.5	
	PC1	315323.6	5177520.5	
	PC2	315491.4	5177417.0	
	PC3	315657.7	5177305.5	
	PC4	315827.0	5177197.0	
	PC5	315995.1	5177090.5	
	SW	315187.5	5177492.5	
	SE	316027.5	5176952.0	
BRMaP 17	NE	317756.9	5165893.0	Wahluke
	NW	317130.1	5166676.5	
	PC1	317112.2	5166535.5	
	PC2	317234.5	5166370.5	
	PC3	317360.1	5166215.5	
	PC4	317488.0	5166059.5	
	PC5	317620.1	5165906.5	
	SW	316971.5	5166557.5	
	SE	317596.0	5165771.0	
BRMaP 11	ROAD	319587.1	5154863.0	River Corridor
	NE	320382.7	5155633.5	
	NW	319597.5	5155001.5	
	PC1	319745.2	5154984.0	
	PC2	319894.3	5155113.5	
	PC3	320057.2	5155234.0	
	PC4	320212.6	5155359.0	
	PC5	320370.3	5155487.0	
	SW	319726.4	5154849.5	
	SE	320503.4	5155475.0	
BRMaP 27	NE	322359.6	5153047.0	River Corridor
	NW	322183.9	5153144.0	
	PC1	321775.8	5152352.0	
	PC2	321891.2	5152516.0	
	PC3	322002.5	5152678.0	
	PC4	322111.4	5152847.0	
	PC5	322221.4	5153015.0	
	SW	321632.4	5152330.0	
	SE	321795.9	5152212.0	

IV. Transition Density Plots.

A. Plot Indentification

Transect origins (typically along a road or wash) were marked with fence posts. Transect lines were marked with ½” x 36” rebar at 100m intervals. All posts and rebar were marked with aluminum tags as follows: “T” (for transition plot) followed by a number (the transect number) above, with a second number, the stake number (identifying distance along the transect) below. For example, a disk marked as follows:

T1
2

indicates transition density transect T1 at the 200m point from the origin.

See main text for methods. Microplots are located at 25 m intervals and face backwards towards the origin. The sole exception to this convention is the origin microplot itself, which faces away from the origin. Microplots are ALWAYS read to the right of the line (facing from origin towards terminus).

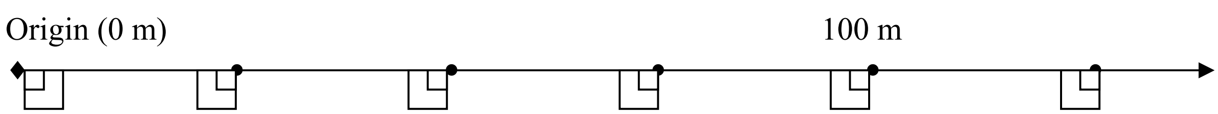


Fig. A12. Placement of 20 x 20 cm microplots (small squares) within 1 x 1 m microplots (larger squares) along Transition Density Transects. Tick marks represent 25 m intervals. Origin is exceptional in that microplots face forward along the transect: all other microplots face back towards the origin.

B. Directions to Individual Plots (all bearings are MAGNETIC). GPS coordinates are presented in Table A5.

T1.
Begins along Rattlesnake Mt. Rd. (~1400') just before Rd. begins to climb steeply and curve north. Transect runs at a bearing of 300^{o} for 900m across toe of gentle slopes at east end of Rattlesnake Mt. Road stake is a steel fencepost 15m off road. An unrelated old fencepost (rusty & blackened, next to an old wooden box) sits another 15m from rd., nearby but offline. The original (2001) fencepost origin for this plot was missing in 2004 and replaced with a new one.

T2.
Begins @ ~1900' on Rattlesnake Mt. Rd. Runs NE (30^{o} bearing) across 'bench' then downhill through pristine AGSP-POSA to intersect w/ T1 at base of slope. Road stake is a steel fencepost 15m off road. SIAL, BRTE & SAKA abundant along roadside, but there's a quick transition into predominantly native vegetation.

T3.
Located approaching the crest of a broad ridge which rises beyond & to the SE of the 1st draw SE of the Gate 111 Rd. where it crosses the Benton REA wooden powerline road. Origin fencepost (1000') is located along powerline rd. ~ 480m uphill from where it intersects w/ Gate 111 Rd. Fencepost is on downhill side of rd., about 5m off road.

Initial bearing downslope is 60^{o} – towards a white reactor dome on central Hanford. At 300m stake, the bearing changes to 40^{o} – towards an isolated, square building out on central Hanford. Community is AGSP-POSA. Together, T3-T4 has probably the densest BRTE distribution of any of the transition density polygons.

T4.
Located on the 1st major draw to SE of Gate 111 Rd. just downhill of where it intersects w/ the Benton REA wooden powerline rd. Origin fencepost (900') is in draw just uphill from dense BRTE-SIAL infestation (in the draw) and very close to Biodiversity Plot DW 43. Transect follows a bearing of 170^{o} as it rises to it's terminus, 800m away.

Plot origin is 330m from intersection of Gate 111 Rd. & powerline rd. From where the powerline rd., heading uphill, crosses the draw it is about 300m down the draw to the origin fencepost. Or plot may be accessed by driving ~ 0.3 km down Gate 111 Rd. from intersection w/ powerline rd. & crossing over convexity between road & draw.

Together, T3-T4 has probably the densest BRTE distribution of any of the transition density polygons.

T5.
Origin fencepost is at ~1150', just off N side of 1200' Rd. in vicinity of nearest point to Biodiversity Plot 46. In the 1st bend of a shallow S-curve 6.0 km NW from Gate 107. Follow bearing of 20^{o} 1500m siting on a gravel pit in Cold Creek Valley. Plant community is very good AGSP-POSA. LULE is prominent in 1st 300-400m. Transect length 1500m.

T6.
Origin fencepost is on far side of 1st draw to NW of T5. Post 10 (1000 m) of T5 coincides w/ post 2 (200m) of T6. The draw, esp. the NW side (i.e., the SE-facing side) is dense BRTE-SIAL. From 200m stake (post 10 of T5) on crest of convexity, site on cluster of trees at Benson Ranch & head towards them on a bearing of ~ 300° to find origin. The terminus (400m in the opposite direction, bearing ~ 120°) is in the bottom of a steep-sided draw full of large BRTE & SIAL. Transect length 400m.

T6-5.
Continuation of T6 numbering on new, short transect. Origin fencepost is in shallow, BRTE-SIAL draw about 300m downslope from 1200' Rd. Bearing 300°, elevation ~ 1160'. Transect crosses T5 transect between stakes T5-2 and T5-3 (in area w/ prominent LULE). Transect length 200m.

T6-8.
Origin fencepost is in the same draw as T6B, ~ 150m downslope from the 1200' Rd., elev. 1120' This SE-facing side of draw is dense w/ SIAL & BRTE; opposite side not as bad. Transect length 100m.

T7.
Across 1200' Rd. and about 100m closer to Gate 107 than T5 – 5.9 km NW of Gate 107. This 800m transect begins at 1180' and heads due south (180° mag.) towards the middle cluster of towers on the summit of Rattlesnake Mt. Plant community is good AGSP-POSA w/ abundant LULE, good MBC almost to roadside. Invasive spp. (BRTE, SIAL, LASE, SAKA) confined to roadside.

T8.
Intersects T7 near 800m stake of T7. Origin fencepost is @ edge of shallow BRTE/SIAL draw just E of T& transect. Bearing is due west (270° mag) 625 m through Good AGSP-POSA w/ abundant ACMI, LULE, POCU. General aspect of slope T7-8 traverse is ~ 20° mag.

T9.
Origin fencepost located on the west side of Gate 117 Rd. (Elev. 620') in very heavily burned ARTR, w/ ARTR refugia on E side of rd. 75-100m N of jct. W/ Rattlesnake Springs cutoff rd. Area was formerly ARTR-POSA, probably w/ lots of cheat. Now forb-dominated: HECU, BACA, PEAC, CYTE, patches of ORHY, AMAC, DEPI, abundant CHLE at W end. Aliens include SAKA, BRTE, SIAL, w/ AGCR, POBU, and other aliens in vicinity of Benson Ranch. Bearing 270° mag. Transect length 450m.

T9-5.
Transect perpendicular to T9. Origin fencepost on Rattlesnake Springs cutoff Rd. Bearing 360^{o} mag. Transect length 250m.

T9-9.
Transect perpendicular to T9. Origin fencepost on Rattlesnake Springs cutoff Rd. Bearing 360^{o} mag. Transect length 250m.

T9-14.
Origin fencepost located on the west side of Gate 117 Rd. about 0.5 km (?) south of Benson Ranch. Crosses & runs roughly parallel, but divergent from old fenceline towards far end of Yakima Ridge. Area was heavily burned. Alien spp. include POBU & AGCR. Beautiful stand of ORHY w/ ASCA in burnt out sage ~ 50m south & between 9/16 & 9/18. Bearing 260^{o} mag. Transect length 400m.

T10 - 0.
Transect runs uphill from road closure at Lower Snively Spring. Passes W of Biodiversity Plot 45 between T10-1 and T10-2. Origin fencepost is about 8m above Rd. Bearing 170^{o} mag. Transect length 350m.

T10 – 5.
Transect contours around hillslope upslope of Biodiversity Plot 45. Bearing varies from 220-270^{o} mag. Transect length 350m.

T10-13.
Transect contours around hillslope just downslope of Biodiversity Plot 45. Bearing varies from 230-290^{o} mag. Transect length 225m.

T11.
Iowa Flats. Transect runs downslope roughly parallel to Gate 106 Rd., west of high voltage powerline. Origin fencepost is off rd. about 150m on north side of rd. Excellent AGSP-POSA. Original bearing 90^{o} mag. for 1st 300m. Bearing changes to 70^{o} at 300m stake. Transect length 1075m.

T12-0.
Origin fencepost is at edge of dozerline along Gate 106 Rd., where aliens of highway swath meet native community. Between high voltage powerline & 1st sharp curve on 106 Rd. Transect is perpendicular to T11 transect. T12-0 is located just ~ 50 m downslope of yellow curve warning side on Gate 106 Rd. Bearing 360^{o} mag. Transect length 500m.

T12-6.
Origin fencepost is at edge of dozerline along Gate 106 Rd., where aliens of highway swath meet native community. 300m closer to high voltage line than 12-0. Transect is perpendicular to T11 transect, crossing it between T11-6 and T11-7. T12-0 is located just ~ 50m downslope of yellow curve warning side on Gate 106 Rd. Bearing 360^{o} mag. Transect length 500m.

T13.
Located along 1200' Rd. about 900m west of jct. w/ 117Rd. 920' ASL. Origin fencepost is along roadside. Bearing 350^{o} mag. from post towards trees at Benson Ranch. Transect length 900m.

T14.
Origin in shallow side draw, branch off first major draw crossed by T13, 809' a.s.l. Climbs out of little draw & proceeds down gently sloping ridge. Draw has good AGSP w/ scattered dense patches of BRTE & SIAL. Bearing 310^{o} mag. Transect length 1100m.

T15.
Top of side draw ~300 m below 1200' Rd & T13 origin, 880'a.s.l. On east side of ridge, where ridge turns NW. Draw has BRTE & SIAL in moderate densities. Bearing 310^{o} mag. Transect length 400m.

T16-0.
Begins on NE-facing side of deep draw east of T16-6 in large, dense SIAL & BRTE. 2980' a.s.l. Transects runs across & down slope near ALE border. Bearing 270^{o} mag. Transect length 500m.

T16-6.
Origin fencepost 5m inside fence along road bordering ALE, about 0.5 mi. west of microwave towers -- Hamilton wheatfields to the south across rd. Access is from microwave tower rd. (private land) via Benton City as in Biodiversity plot 107 (nearby). Transect begins in dense roadside cheat, SIAL, even some wheat, mixed w/ natives. Transect joins BRMP 13/PC5 at 100m stake & coincides w/ that transect for last 100m. Bearing 360^{o} mag. Transect length 200m.

T17.
Located along 1200' Rd. NW of Gate 107 ~250m SE of BRMP 7 road stake. Origin fencepost is on dozer berm 3-4m off north side of road in weedy vegetation. Transect runs down gentle 5^{o} slope from 1240' a.s.l. towards wooden powerline. Bearing 25^{o} mag. Transect length 1100m.

T18-0.
Transect origin is in dense BRTE-SIAL ~400m downslope from 1200' Rd. 1160' a.s.l. SIAL-BRTE patch is on uphill edge of draw. Transect is perpendicular to T17. Bearing 112^{o} mag. Transect length 525m.

T18-B
Transect begins at BRMP 7/ PC3 100m mark. T18/6 is the SECOND (or 100m) stake along this transect. Bearing 120^{o} mag. Total transect length 350m.

T19.
Located on so. side of wooden powerline rd. uphill from the ALE labs/ Nike site, on gentle lower slopes of Rattlesnake Mt. 1300' a.s.l. Roadside vegetation is dense BRTE-SIAL w/ AMLY. Excellent AGSP-POSA plant community. MBC is in excellent condition in this area w/ small scale disturbances (animal mounds).

T19B.
Shares origin fencepost w/ T20. Begins in dense SIAL-BRTE at edge of powerline & buried telephone cable swath. Transect runs across slope perpendicular to T19-0 transect. 1400' a.s.l. Bearing 130^{o} mag. Transect length 375m.

T20.
Origin about 400m up wooden powerline rd. from T19-0. Shares origin w/ 19B. Transect runs upslope through VG AGSP-POSA. Bearing 220^{o} mag. Transect length 1200m.

T21.
In disturbed swath along wooden powerline rd. upslope from T19B/ T20. Transect runs across slope perpendicular to and intersecting w/ T19-0 and T20 transects. 1540' a.s.l. Bearing 130^{o} mag. Transect length 650m.

Table A6. GPS coordinates for Transition density plots on the ALE Reserve, established 2001. All coordinates recorded using a Garmin *etrex* portable GPS unit, UTM Zone 11, datum NAD27.

Plot	UTM N	UTM E	Marker Type	Length of Transect (m)	Transect Bearing (mn)
T1 - 0	5138151	304816	Fencepost	900	300
T2 - 0	5138209	303896	Fencepost	1000	30
T3 - 0	5143557	305131	Fencepost	900	60
T4 - 0	5144012	305448	Fencepost	800	170
T5 - 0	5144787	301318	Fencepost	1500	20
T6 - 0	5145702	301825	Fencepost	400	120
T6 - 5	5144956	301531	Fencepost	200	300
T6 - 8	5144843	301429	Fencepost	100	300
T7 - 0	5144706	301361	Fencepost	800	180
T8 - 0	5143962	301115	Fencepost	625	20
T9 - 0	5151857	296200	Fencepost	450	270
T9 - 5	5151806	296004	Fencepost	250	360
T9 - 9	5151883	295870	Fencepost	400	360
T9 - 14	5152145	296094	Fencepost	400	260
T10 - 0	5148478	291061	Fencepost	350	180
T10 - 5	5148460	290897	Fencepost	350	270
T10 - 13	5148313	291091	Fencepost	225	290
T11 - 0	5138558	308468	Fencepost	1075	90*
T12 - 0	5138375	308775	Fencepost	500	360
T12 - 6	5138362	309039	Fencepost	500	360
T13 - 0	5149035	295435	Fencepost	1000	350
T14 - 0	5149163	295718	Fencepost	1000	310
T15 - 0	5149251	295528	Fencepost	400	310
T16 - 0	5144253	295954	Rebar	500	270
T16 - 6	5144221	295890	Rebar	200	360
T17 - 0	5142161	303842	Fencepost	1100	25
T18 - 0	5142648	304016	Fencepost	525	112
T18 - 6	5142722	304110	Fiberglass wand	450	120
T19 - 0	5140412	304470	Fencepost	1150	220
T19b	5140454	304200	T20 fencepost	375	130
T20 - 0	5140454	304200	Fencepost	1200	220
T21 - 0	5140548	3033628	Fencepost	650	130

* At the 300m stake bearing changes to 70^{o}.

V. Rehabilitation Vegetation Plots.

Rehabilitation plots were established between November 2002 and May 2003. Plots are 100 m transects. All plots except drill seed plots are oriented towards magnetic north, marked at both ends with ½" rebar. Coordinates are for the south end (origin) of each transect. Drill seed plots are arrayed roughly parallel to State Route 240; transect origins are to the north or northwest.

Table A7. GPS coordinates for rehabilitation plots. All coordinates recorded using a Garmin *etrex* portable GPS unit, UTM Zone 11, datum NAD27.

Polygon	Plot	UTM East	UTM North
P1	14	290006	5160414
	19	290635	5160145
	37	287708	5160780
P2	1	295463	5153972
	3	294712	5153638
	8	295262	5153352
P3	1	294941	5150593
	2	292124	5152641
	3	295275	5150112
	4	292944	5152037
	6	292391	5150598
	14	293974	5152627
	T9-6	296038	5151887
	T13-2	295473	5149235
	T13-8	295585	5149815
P4	2	300213	5148386
	36	297884	5150137
	90	300732	5147108
Shrub Seed	7	303110	5142533
	13	297988	5147000
	T7-1	301338	5144625
Drill Seed	3	293845	5157925
	6	296219	5153921
	7	297906	5153480
	9	300588	5151982
	11	303235	5150059
	13	305829	5148163
	15	308387	5146256
	18	311441	5142427
	19	312169	5141162

VI. Sagebrush Refugia plots

Study plots were established along the margins of un burned sagebrush patches in 2001 to measure the effects of the 24 Command Fire on cheatgrass density in burned areas adjacent to unburned refugia. Plots were not permanently marked, but, with the aid of GPS, the areas sampled in 2002, 2003, and 2004 were very close to the location of the original plots sampled in 2001.

R1. ARTR/AGSP community type on south side of 117 Rd., just below intersection w/ 1200' Rd. Narrow band of sagebrush was protected from 24 Command Fire by retardant drops in failed attempt to create a firebreak. Rather weedy, with little microbiotic crust (MBC).

R2. ARTR/POSA community type at junction of old track on south side of 117 Rd., west of and uphill from intersection w/ Rattlesnake Spring cutoff rd. Refugia generally in excellent condition with fair-good MBC.

R3. ARTR/POSA community on south side of 117 Rd. between Benson Ranch & Rattlesnake Spring cutoff rd. Crusts poor-fair.

R4. ARTR/POSA community on south side of 117 Rd., overland a short distance from R3. Crusts poor.

R5. North side of 118 Rd., east of and above Rattlesnake Spring and the intersection w/ road that runs north along base of Yakima Ridge. Refugia and crusts in fair condition w/ many elk trails.

R6a. Excellent ARTR/POSA community at about 900’ elev., just off on south side of 118 Rd. MBC in very good condition.

R6b. Excellent ARTR/POSA community at about 900’ elev., just off on south side of 118 Rd. MBC in very good condition. West of 6A at opposite end of same refugia.

R7. ARTR-GRSP/ POSA community along Rattlesnake Springs Bypass Rd. On west side of that road. Rather weedy site with poor MBC.

R8. ARTR/AGSP community on north side of 118 Rd. Elev. about 950’. Split plot using two small patches of fair-good quality.

R9. ARTR/POSA community on north side of 118 Rd., across wide draw from road. Elev. about 850’. Large patches with some good MBC

R10. ARTR/POSA community. Split-plot on both sides of road that runs north along base of Yakima Ridge. Small patches with some good MBC in refugia.

R11. Excellent ARTR-GRSP/POSA community underneath Gate 120 BPA powerline and just north of Gate 120 powerline access rd. On west side of broad drainage with refugia on both sides of drainage. Outstanding and diverse MBC.

R12. ARTR-GRSP/POSA community, north of R11 along same side of broad draw, in ± continuous ARTR along slope above burned area

R13. ARTR-GRSP/POSA community on broad flat above crest of east slope of broad draw where R11 &R12 are located. One of the larger blocks of sagebrush remaining in this area. Very weedy area, but some good patches with intact MBC are present.

R14. ARTR/POSA community in mosaic burn area near upper Cold Creek Access is from SR 24, about 200m away. Area is fairly weedy and MBC sparse.

R15. ARTR/POSA community on sandy soil at intersection of 117 Rd and Rattlesnake Spring cutoff road. Community impacted by blowing sand from adjacent burned areas and only in fair condition with little MBC.

R16. South of Gate 120 Rd., bordering area burned in 1998 wildfire. Near BRMaP 23. Refugium is bisected by a very old jeep or wagon track. Condition variable, MBC fair.

Table A8. GPS coordinates for sagebrush refugia plots. All coordinates recorded using a Garmin *etrex* portable GPS unit, UTM Zone 11, datum NAD27.

Plot	UTM East	UTM North
1	296352	5148925
2	296412	5151177
3	296263	5151808
4	296481	5151726
5	293213	5153919
6A	291884	5152219
6B	291938	5152161
7	294134	5153898
8	291319	5150906
9	291566	5152550
10	292392	5156026
11	290917	5158919
12	290430	5159004
13	290922	5159176
14	288611	5160943
15	296302	5151588
16	291497	5158590

VII. Sagebrush Survival Plots.

Sagebrush survival plots were established during winter 2003 after planting in fall 2002. Plots are laid out as 100 m transects oriented towards magnetic north from randomly selected origins. ½" rebar stakes mark the origin and the terminus of each plot. No tags identify the rebar.

In 2003 sampling was stopped at a plot when a sample size of ≥ 100 plants was assured. Because of varying planting densities the length of transect over which data was collected varied from plot to plot (although the baseline was always 100 m. Two plots with lower densities were extended beyond 100 m, but this practice was not continued.

Table A9. GPS coordinates for sagebrush survival plots. All coordinates recorded using a Garmin *etrex* portable GPS unit, UTM Zone 11, datum NAD27.

Polygon	Plot #	Transect Length	UTM East	UTM North
A	4	100	305264	5142499
A	5	50	304899	5142022
A	9	100	304558	5141652
A	15	90	304789	5142298
A	17	70	305423	5141757
B	5	100	302229	5144766
B	6	100	302023	5144545
B	9	70	302209	5144640
C	1	100	288549	5150338
C	4	100	288669	5150995
C	11	100	288495	5150030
D	1	125	288519	5149713
D	2	100	288526	5149477
D	3	115	288363	5149654
F	1	100	298737	5147048
F	3	90	298570	5146929
F	6	100	298884	5147121
J	1	100	299060	5146069
J	2	100	299121	5145978
J	4	80	298955	5145878
K	4	80	302443	5143198
K	5	100	302472	5143418
K	24	100	302750	5143230
LM	2	100	306431	5140379
LM	3	100	306786	5140032
LM	5	100	306479	5139976

Appendix B. Environmental variables for permanent vegetation and density plots on the ALE Reserve.

Elevations are given in feet. Aspect values were recorded as magnetic values in the field. They are presented here corrected for magnetic declination. See main text for calculations of rescaled aspect and Heat Load Index, and an explanation of fire severity codes.

Table B1. Environmental variables for Biological Resources Management Plan (**BRMaP**) **Plots**.

Plot	Pre-fire Cover Type	Pre-fire Shrub Cover	Elevation	Fire Severity Code	Soil Type	Slope (^{o})	Aspect (^{o})	Rescaled Aspect	Heat Load Index
7-1	AGSP		1200	2	Silt loam	3	45	0	0.8622
7-3	AGSP		1140	2	Silt loam	3	40	0.0019	0.8623
7-5	AGSP		1080	2	Silt loam	3	20	0.0468	0.8646
8-1	AGSP		1250	3	Silt loam	5	45	0	0.8391
8-3	AGSP		1380	2	Silt loam	5	45	0	0.8391
8-5	AGSP		1520	2	Silt loam	5	45	0	0.8391
12-1	ARTP	24.1	2540	2	Silt loam	5	40	0.0019	0.8393
12-3	ARTP	2.4	2640	2	Silt loam	25	330	0.3706	0.7307
12-5	ARTP	8.3	2680	2	Stony loam	5	110	0.2887	0.8634
13-1	ARTP	19.4	2660	2	Silt loam	5	20	0.0468	0.8430
13-3	ARTP	8.4	2760	2	Stony loam	10	330	0.3706	0.8408
13-5	ARTP	2.9	2960	2	Silt loam	12	20	0.0468	0.7631
20-1	ARTR	28.7	640	4	Sand	0	0	0.1464	0.8956
20-2	ARTR	16.5	650	4	Sand	0	0	0.1464	0.8956
20-3	ARTR	32.0	660	4	Sand	0	0	0.1464	0.8956
20-4	ARTR	17.2	670	4	Sand	0	0	0.1464	0.8956
20-5	ARTR	14.8	675	4	Sand	0	0	0.1464	0.8956
23-1	ARTR	8.6	840	3	Silt loam	0	0	0.1464	0.8956
23-3	ARTR	34.5	850	3	Silt loam	3	75	0.0670	0.8656
23-5	ARTR	30.3	840	3	Silt loam	1	80	0.0904	0.8862
24-1	ARTR	33.9	840	4	Silt loam	2	30	0.0170	0.8741
24-2	ARTR	28.4	820	4	Silt loam	2	40	0.0019	0.8736
24-3	ARTR	23.8	1200	3	Silt loam	5	30	0.0170	0.8405
24-4	ARTR	22.8	1140	4	Silt loam	3	70	0.0468	0.8646
24-5	ARTR	29.2	1080	4	Silt loam	5	20	0.0468	0.8430

Table B2. Environmental variables for **Steppe-in-Time Plots**.

Plot	Pre-fire Cover Type	Pre-fire Shrub Cover	Elevation	Fire Severity Code	Soil Type	Slope (^{o})	Aspect (^{o})	Rescaled Aspect	Heat Load Index
14E	ARTR	8.3	780	4	Silt loam	7	60	0.0170	0.8174
14W	ARTR	7.0	780	4	Silt loam	5	60	0.0170	0.8405
31E	AGSP		1080	2	Silt loam	5	240	0.9830	0.9217
31W	AGSP		1040	2	Silt loam	2	10	0.0904	0.8765
47E	AGSP		1170	2	Silt loam	7	30	0.0170	0.8174
47W	AGSP		1180	2	Silt loam	7	30	0.0170	0.8174
48	ARTP	7.8	2110	2	Silt loam	3	290	0.7113	0.8981
60E	AGSP		1220	2	Silt loam	3	90	0.1464	0.8696
60W	AGSP		1280	2	Silt loam	3	90	0.1464	0.8696
66E	ARTP	1.0	1480	2	Silt loam	25	355	0.1786	0.6525
66W	AGSP		1240	2	Stoney loam	2	50	0.0019	0.8736
67E	AGSP	3.3	1160	2	Stoney loam	5	290	0.7113	0.8989
67W	AGSP		1160	2	Silt loam	3	360	0.1464	0.8696
188	AGSP	3.0	3360	2	Stoney loam	25	340	0.2887	0.6973
326	AGSP	0.1	1600	3	Silt loam	20	340	0.2887	0.7438

Table B3. Environmental variables for **Biodiversity Plots**. Plots indicated by an asterisk [*] did not burn in 2000.

Plot	Pre-fire Cover Type	Pre-fire Shrub Cover	Elevation	Fire Severity Code	Soil Type	Slope (^{o})	Aspect (^{o})	Rescaled Aspect	Heat Load Index
33	ARTR	20	680	2	Silt loam	1	80	0.0904	0.8862
34	ARTR-GRSP	20	660	2	Silt loam	3	360	0.1464	0.8696
35*	ARTR	30	700	0	Silt loam	3	50	0.0019	0.8623
36	ARTR-GRSP	30	830	4	Silt loam	1	130	0.4564	0.8923
37	ARTR	30	940	4	Silt loam	3	360	0.1464	0.8696
38	ARTR-GRSP	20	820	4	Silt loam	0	0	0.1464	0.8956
39	ARTR	10	1050	2	Silt loam	5	20	0.0468	0.8430
42	STCO	1	760	2	Silt loam	10	350	0.2132	0.8145
43	AGSP	0	860	2	Silt loam	10	340	0.2887	0.8271
44	EULA	20	900	3	Silt loam	1	90	0.1464	0.8871
45	AGSP	0	1400	1	Silt loam	15	40	0.0019	0.7156
46	AGSP	0	1020	2	Silt loam	3	30	0.0170	0.8631

(Continued….)

Table B3 (continued). Environmental variables for **Biodiversity Plots**. Plots indicated by an asterisk [*] did not burn in 2000 (Continued).

Plot	Pre-fire Cover Type	Pre-fire Shrub Cover	Elevation	Fire Severity Code	Soil Type	Slope (^{o})	Aspect (^{o})	Rescaled Aspect	Heat Load Index
47	EULA	10	760	3	Silt loam	5	20	0.0468	0.8430
50	EULA	10	740	3	Silt loam	3	60	0.0170	0.8631
51	STCO	1	720	3	Silt loam	5	332	0.3538	0.8688
52	AGSP	0	670	2	Silt loam	1	40	0.0019	0.8847
53	STCO	1	600	3	Silt loam	5	35	0.0076	0.8397
54	AGSP	0	670	2	Silt loam	1	80	0.0904	0.8862
55	AGSP	0	675	2	Silt loam	3	40	0.0019	0.8623
56	ARTR	20	800	3	Silt loam	3	350	0.2132	0.8729
58	ARTP	10	1880	3	Silt loam	1	40	0.0019	0.8847
59	ARTP	10	2200	3	Silt loam	40	370	0.0904	0.4203
61	AGSP	0	1410	2	Silt loam	10	40	0.0019	0.7791
71	AGSP	0	880	2	Silt loam	3	20	0.0468	0.8646
73	PUTR	3	560	2	Sand	1	180	0.8536	0.8990
74	ARTP	1	3200	2	Stoney loam	30	40	0.0019	0.5099
75	ARTP	3	3400	1	Stoney loam	25	350	0.2132	0.6666
78	ARTP	1	2620	2	Stoney loam	30	35	0.0076	0.5127
79	AGSP	0	1540	2	Silt loam	5	80	0.0904	0.8467
88	ARTP	1	3400	1	Stoney loam	20	10	0.0904	0.6784
89	ARTP	1	2920	2	Stoney loam	30	60	0.0170	0.5172
93	AGSP	1	1360	2	Silt loam	5	40	0.0019	0.8393
94	ARTP	10	1980	3	Silt loam	10	40	0.0019	0.7791
95	ARTR	20	920	4	Silt loam	1	360	0.1464	0.8871
96	ARTR	30	820	4	Silt loam	3	10	0.0904	0.8668
102	AGSP	0	760	2	Silt loam	2	50	0.0019	0.8736
105	AGSP	1	2080	2	Silt loam	3	80	0.0904	0.8668
106	ARTP	10	1750	2	Silt loam	3	20	0.0468	0.8646
107	ARTP	10	2920	3	Silt loam	20	320	0.4564	0.7991
108	ARTP	3	3400	2	Stoney loam	25	40	0.0019	0.5805
109	AGSP	0	1750	2	Silt loam	10	312	0.5262	0.8669
111*	AGSP	0	3410	0	Stoney loam	15	230	0.9981	0.9642

Table B4. Environmental variables for **Transition density plots**.

Plot	Elevation - start	Elevation - finish	Avg. Elevation	Slope	Aspect	Rescaled Aspect	Soil Type	Heat Load Index
1	1450	1470	1460	4	80	0.0904	Silt loam	0.8568
2	1890	1390	1640	12	50	0.0019	Silt loam	0.7541
3	1000	840	920	4	70	0.0468	Silt loam	0.8539
4	900	960	930	4	70	0.0468	Silt loam	0.8539
5	1180	900	1040	5	30	0.0170	Silt loam	0.8405
6	1060	1000	1030	5	30	0.0170	Silt loam	0.8405
7	1180	1280	1230	3	10	0.0904	Silt loam	0.8668
8	1280	1340	1310	3	30	0.0170	Silt loam	0.8631
9	620	620	620	0	20	0.0468	Sand	0.8956
10	1300	1530	1415	22	20	0.0468	Silt loam	0.6382
11	940	850	895	1	100	0.2132	Silt loam	0.8882
12	900	860	880	1	95	0.1786	Silt loam	0.8876
13	920	750	835	5	20	0.0468	Silt loam	0.8430
14	890	690	790	2	330	0.3706	Silt loam	0.8860
15	880	800	840	2	330	0.3706	Silt loam	0.8860
16	3000	2820	2910	10	20	0.0468	Stony loam	0.7866
17	1240	1080	1160	4	30	0.0170	Silt loam	0.8519
18	1150	1150	1150	4	4	0.1226	Silt loam	0.8590
19	1320	1680	1500	5	55	0.0076	Silt loam	0.8397
20	1400	1820	1610	5	55	0.0076	Silt loam	0.8397
21	1540	1610	1575	8	55	0.0076	Silt loam	0.8044

Table B5. Environmental variables for **Rehabilitation Plots**.

Polygon	Plot	Elevation	Slope	Aspect	Rescaled Aspect	Soil Type	Heat Load Index
1	**14**	760	0	0	0.1464	Silt loam	0.8956
1	**19**	730	0	0	0.1464	Silt loam	0.8956
1	**37**	880	0	0	0.1464	Silt loam	0.8956
2	**1**	650	1	60	0.0170	Sand	0.8849
2	**3**	670	2	20	0.0468	Sand	0.8751
2	**8**	640	2	110	0.2887	Sand	0.8832
3	**1**	700	7	98	0.1991	Silt loam	0.8388
3	**2**	760	5	295	0.6710	Silt loam	0.8955
3	**3**	730	5	325	0.4132	Silt loam	0.8738
3	**4**	760	3	30	0.0170	Silt loam	0.8631
3	**6**	900	5	0	0.1464	Silt loam	0.8514
3	**14**	660	0	0	0.1464	Silt loam	0.8956
3	**T9-6**	620	0	0	0.1464	Sandy loam	0.8956
3	**T13-2**	880	4	340	0.2887	Silt loam	0.8701
3	**T13-8**	800	35	340	0.2887	Silt loam	0.5966
4	**2**	760	1	30	0.0170	Silt loam	0.8849
4	**36**	700	3	340	0.2887	Silt loam	0.8768
4	**90**	850	5	240	0.9830	Silt loam	0.9217
SS	**7**	1300	5	40	0.0019	Silt loam	0.8393
SS	**13**	1180	7	10	0.0904	Silt loam	0.8260
SS	**T7-1**	1200	2	10	0.0904	Silt loam	0.8765
DS	**3**	650	0	0	0.1464	Sand	0.8956
DS	**6**	640	0	0	0.1464	Sand	0.8956
DS	**7**	600	1	240	0.9830	Sand	0.9012
DS	**9**	600	3	240	0.9830	Sand	0.9118
DS	**11**	530	0	0	0.1464	Sandy loam	0.8956
DS	**13**	520	0	0	0.1464	Silt loam	0.8956
DS	**15**	500	0	0	0.1464	Sandy loam	0.8956
DS	**18**	480	1	205	0.9698	Sandy loam	0.9010
DS	**19**	500	0	0	0.1464	Sandy loam	0.8956

Table B6. Environmental variables for **Sagebrush Refugia Plots.**

Plot	Elevation	Slope	Aspect	Soil Type
R-1	960	3	20	Silt loam
R-2	680	<2	20	Silt loam
R-3	600	0	20	Sand
R-4	670	<2	20	Sandy loam
R-5	695	3	220	Silt loam
R-6A	860	<2	20	Silt loam
R-6B	860	<2	20	Silt loam
R-7	645	3	173	Silt loam
R-8	950	<2	10	Silt loam
R-9	850	<2	30	Silt loam
R-10	630	<2	80	Silt loam
R-11	660	5	10	Sandy loam
R-12	700	5	340	Silt loam
R-13	740	2	50	Silt loam
R-14	820	2	340	Silt loam
R-15	670	2	20	Sand
R-16	710	2	75	Silt loam

Table B7. Environmental variables for **Sagebrush Survival Plots.**

Polygon	Plot	Slope	Aspect	Rescaled aspect	HLI	Elev'n	Percent Cover - 2004			
							Bromus tectorum	*Agropyron spicat*um	All perennial grasses	All grasses
A	4	8	115	0.329	0.8475	1000	1.9	17.1	27.1	29.0
A	5	5	330	0.371	0.8702	1080	1.2	17.2	26.0	27.2
A	9	2	30	0.017	0.8741	1200	0.1	13.8	23.5	23.6
A	15	3	135	0.500	0.8874	1040	5.7	18.0	28.0	33.7
A	17	5	290	0.711	0.8989	1020	0.2	23.6	31.6	31.8
B	5	2	100	0.213	0.8807	1120	1.5	17.8	29.3	30.8
B	6	5	40	0.002	0.8393	1180	17.2	18.7	29.8	29.8
B	9	2	40	0.002	0.8736	1120	13.8	22.1	36.9	37.3
C	1	5	55	0.008	0.8397	1200	18.0	22.5	28.0	47.0
C	4	3	200	0.953	0.9103	1200	23.6	21.2	25.8	41.2
C	11	5	140	0.544	0.8848	1120	17.8	19.8	29.4	38.9
D	1	5	360	0.146	0.8514	1280	0.0	26.4	39.5	39.5
D	2	10	50	0.002	0.7791	1350	5.1	25.7	29.8	43.2
D	3	8	380	0.047	0.8096	1360	3.1	19.7	36.9	32.5
F	1	5	360	0.146	0.8514	1110	0.6	22.2	28.0	33.2
F	3	5	30	0.017	0.8405	940	2.3	21.4	25.8	34.2
F	6	3	40	0.002	0.8623	1080	1.6	18.4	29.4	30.6
J	1	5	30	0.017	0.8405	1280	0.0	17.6	39.5	30.6
J	2	5	25	0.030	0.8416	1320	0.1	17.6	29.4	29.5
J	4	8	380	0.047	0.8096	1390	0.1	13.8	29.4	29.5
K	4	5	45	0.000	0.8391	1240	0.0	28.8	40.3	40.3
K	5	5	40	0.002	0.8393	1200	4.8	10.8	23.4	28.2
K	24	5	380	0.047	0.8430	1260	0.0	12.5	24.5	24.5
LM	2	2	70	0.047	0.8751	1120	3.3	26.4	38.0	41.3
LM	3	2	380	0.047	0.8751	1080	0.0	16.9	29.7	29.7
LM	5	2	90	0.146	0.8784	1080	0.5	17.8	30.2	30.7

Appendix C. Scientific names of vascular plant species encountered in this study. Complete references are listed below. N = native; I = introduced; A = annual; B = biennial; P = perennial. Boldface indicates nomenclatural change since Hitchcock and Cronquist (1973).

Shrubs and Hemishrubs

N / I	A / B / P	Hitchcock and Cronquist (1973)	Kartesz (1999)	Field Code
N	P	*Atriplex spinosa*	***Grayia spinosa***	GRSP
N	P	*Artemisia tridentata*	*Artemisia tridentate ssp. wyomingensis*	ARTR
N	P	*Artemisia tripartita*	*Artemisia tripartita*	ARTP
N	P	*Antennaria dimorpha*	*Antennaria dimorpha*	ANDI
N	P	*Chrysothamnus nauseosus*	***Ericameria nauseosus***	CHNA
N	P	*Chrysothamnus viscidiflorus*	*Chrysothamnus viscidiflorus*	CHVI
N	P	*Erigeron filifolius*	*Erigeron filifolius*	ERFI
N	P	*Erigeron piperianus*	*Erigeron piperianus*	ERPI
N	P	*Eriogonum heracloides*	*Eriogonum heracloides*	ERHE
N	P	*Eriogonum sphaerocephalum*	*Eriogonum sphaerocephalum*	ERSP
N	P	*Eriogonum strictum*	*Eriogonum strictum*	ERST
N	P	*Eriogonum thymoides*	*Eriogonum thymoides*	ERTH
N	P	*Eurotia lanata*	***Krascheninnikovia lanata***	EULA
N	P	*Galium multiflorum*	*Galium multiflorum*	GAMU
N	P	*Haplopappus stenophyllus*	***Stenotus stenophyllus***	HAST
N	P	*Phlox hoodii*	*Phlox hoodii*	PHHO
N	P	*Phlox longifolia*	*Phlox longifolia*	PHLO
N	P	*Phlox speciosa*	*Phlox speciosa*	PHSP
N	P	*Purshia tridentata*	*Purshia tridentata*	PUTR
N	P	*Ribes cereum*	*Ribes cereum*	RICE
N	P	*Symphoricarpos oreophilus*	*Symphoricarpos oreophilus*	SYOR
N	P	*Tetradymia canescens*	*Tetradymia canescens*	TECA

Continued

Graminoids

N / I	A / B / P	Hitchcock and Cronquist (1973)	Kartesz (1999)	Field Code
I	P	*Agropyron cristatum*	***Agropyron desertorum***	AGCR
N	P	*Agropyron dasystachyum*	***Elymus lanceolatus ssp. lanceolatus***	AGDA
N	P	*Agropyron spicatum*	***Pseudoroegneria spicata***	AGSP
I	A	*Bromus tectorum*	*Bromus tectorum*	BRTE
N	P	*Elymus cinereus*	***Leymus cinereus***	ELCI
N	P	*Festuca idahoensis*	*Festuca idahoensis*	FEID
N	A	*Festuca microstachys*	***Vulpia microstachys***	FEMI
N	A	*Festuca octoflora*	***Vulpia octoflora***	FEOC
N	P	*Oryzopsis hymenoides*	***Achnatherum hymenoides***	ORHY
N	P	*Poa juncifolia*	***Poa secunda***	POAM
I	P	*Poa bulbosa*	*Poa bulbosa*	POBA
N	P	*Poa cusickii*	*Poa cusickii*	POCU
N	P	*Poa sandbergii*	***Poa secunda***	POSA
N	P	*Poa scabrella*	***Poa secunda***	POSC
I	AB	*Secale cereale*	*Secale cereale*	SECE
N	P	*Sitanion hystrix*	***Elymus elymoides***	SIHY
N	P	*Stipa comata*	***Hesperostipa comata***	STCO
N	P	*Stipa thurberiana*	***Achnatherum thurberianum***	STTH
I	A	*Triticum aestivum*	*Triticum aestivum*	TRAE

Continued

Forbs

N / I	A / B / P	Hitchcock and Cronquist (1973)	Kartesz (1999)	Field Code
N	P	*Abronia mellifera*	*Abronia mellifera*	ABME
N	P	*Achillea millefolium*	*Achillea millefolium*	ACMI
N	P	*Agastache occidentalis*	*Agastache occidentalis*	AGOC
N	P	*Agoseris grandiflora*	*Agoseris grandiflora*	AGGR
N	A	*Agoseris heterophylla*	*Agoseris heterophylla*	AGHE
N	A	*Ambrosia acanthicarpa*	*Ambrosia acanthicarpa*	AMAC
N	A	*Amsinckia lycopsoides*	*Amsinckia lycopsoides*	AMLY
N	A	*Amsinckia tessellata*	*Amsinckia tessellata*	AMTE
N	P	*Astragalus caricinus*	*Astragalus caricinus*	ASCA
N	P	*Astragalus columbianus*	*Astragalus columbianus*	ASCO
N	P	*Astragalus conjunctus*	***Astragalus conjunctus var. rickardii***	ASCORI
N	P	*Astragalus purshii*	*Astragalus purshii*	ASPU
N	P	*Astragalus sclerocarpus*	*Astragalus sclerocarpus*	ASSC
N	P	*Astragalus spaldingii*	*Astragalus spaldingii*	ASSP
N	P	*Astragalus sclerocarpus*	*Astragalus sclerocarpus*	ASSC
N	P	*Astragalus succumbens*	*Astragalus succumbens*	ASSU
N	P	*Balsamorhiza careyana*	*Balsamorhiza careyana*	BACA
N	P	*Balsamorhiza rosea*	*Balsamorhiza rosea*	BARO
N	P	*Brodiea douglasii*	***Triteliea grandiflora***	BRDO
N	P	*Calachortus macrocarpus*	*Calachortus macrocarpus*	CAMA
N	P	*Castilleja thompsonii*	*Castilleja thompsonii*	CATH
I	P	*Cardaria draba*	*Cardaria draba*	CADR
I	AB	*Centaurea diffusa*	*Centaurea diffusa*	CEDI
I	P	*Centaurea solstitialis*	*Centaurea solstitialis*	CESI
I	P	*Centaurea repens*	***Acroptilon repens***	ACRE
N	P	*Chaenactis douglasii*	*Chaenactis douglasii*	CHDO
N	A	*Chenopodium leptophyllum*	*Chenopodium leptophyllum*	CHLE
I	P	*no record	***Chondrilla juncea***	CHJU
I	A	*Chorispora tenella*	*Chorispora tenella*	CHTE
i	p	*Cirsium arvense*	*Cirsium arvense*	CIAR

Continued

N / I	A / B / P	Hitchcock and Cronquist (1973)	Kartesz (1999)	Field Code
N	A	*Collomia grandiflora*	*Collomia grandiflora*	COGR
N	A	*Collinsia parviflora*	***Collinsia grandiflora***	COPA
N	P	*Comandra umbellata*	*Comandra umbellata*	COUM
N	P	*Crepis atribarba*	*Crepis atribarba*	CRAT
N	P	*Crepis barbigera*	***Crepis atribarba ssp. originalis***	CRBA
N	A	*Cryptantha pterocarya*	*Cryptantha pterocarya*	CRPT
N	P	*Cymopteris terebinthinus*	***Pteryxia terebinthina***	CYTE
N	P	*Delphinium nuttallianum*	*Delphinium nuttallianum*	DENU
N	A	*Descurainia pinnata*	*Descurainia pinnata*	DEPI
I	A	*Descurainia sophia*	*Descurainia sophia*	DESO
N	P	*Dodecatheon cusickii*	***Dodecatheon pulchellum ssp. cusickii***	DOCU
I	A	*Draba verna*	*Draba verna*	DRVE
N	A	*Epilobium paniculatum*	***Epilobium brachycarpum***	EPPA
N	P	*Erigeron corymbosus*	*Erigeron corymbosus*	ERCO
N	P	*Erigeron poliospermus*	*Erigeron poliospermus*	ERPO
N	P	*Erigeron pumilis*	*Erigeron pumilis*	ERPU
N	A	*Eriogonum baileyi*	*Eriogonum baileyi*	ERBA
I	A	*Erodium cicutarium*	*Erodium cicutarium*	ERCI
N	B	*Erysimum asperum*	***Erysimum capitatum***	ERAS
N	P	*Fritillaria pudica*	*Fritillaria pudica*	FRPU
N	A	*Galium aparine*	*Galium aparine*	GAAP
N	A	*Gilia sinuata*	*Gilia sinuata*	GISI
N	P	*Hackelia arida*	***Hackelia diffusa var. arida***	HAAR
N	P	*Helianthus cusickii*	*Helianthus cusickii*	HECU
N	P	*Heuchera cylindrica*	*Heuchera cylindrica*	HECY
I	A	*Holosteum umbellatum*	*Holosteum umbellatum*	HOUM
I	A	*Lactuca serriola*	*Lactuca serriola*	LASE
N	A	*Lagophylla ramosissima*	*Lagophylla ramosissima*	LARA
N	A	*Lappula redowskii*	***Lappula occidentalis***	LARE
N	A	*Layia glandulosa*	*Layia glandulosa*	LAGL
I	P	*Lepidium latifolium*	*Lepidium latifolium*	LELA

Continued

N / I	A / B / P	Hitchcock and Cronquist (1973)	Kartesz (1999)	Field Code
N	A	*Lithophragma parviflora*	*Lithophragma parviflora*	LIPA
N	P	*Lithospermum ruderale*	*Lithospermum ruderale*	LIRU
N	P	*Lomatium dissectum*	*Lomatium dissectum*	LODI
N	P	*Lomatium grayi*	*Lomatium grayi*	LOGR
N	P	*Lomatium gormanii*	*Lomatium gormanii*	LOGO
N	P	*Lomatium macrocarpum*	*Lomatium macrocarpum*	LOMA
N	P	*Lomatium triternatum*	*Lomatium triternatum*	LOTR
N	P	*Lupinus laxiflorus*	***Lupinus arbustus***	LULA
N	P	*Lupinus leucophyllus*	*Lupinus leucophyllus*	LULE
N	A	*Lupinus pusillus*	*Lupinus pusillus*	LUPU
N	P	*Lupinus sulphureus*	***Lupinus bingenensis***	LUSU
N	P	*Machaeranthera canescens*	*Machaeranthera canescens*	MACA
N	P	*Mertensia longiflora*	*Mertensia longiflora*	MELO
N	A	*Mentzelia albicaulis*	*Mentzelia albicaulis*	MEAL
N	P	*Microseris troximoides*	***Nothocalais troximoides***	MITR
N	A	*Microsteris gracilis*	***Phlox gracilis***	MIGR
N	A	*Montia perfoliata*	***Claytonia perfoliata***	MOPE
N	P	*Oenothera pallida*	*Oenothera pallida*	OEPA
N	P	*Penstemon acuminatus*	*Penstemon acuminatus*	PEAC
N	P	*Penstemon glandulosus*	*Penstemon glandulosus*	PEGL
N	P	*Phacelia hastata*	*Phacelia hastata*	PHHA
N	A	*Phacelia linearis*	*Phacelia linearis*	PHLI
N	A	*Plantago patagonica*	*Plantago patagonica*	PLPA
N	A	*Plectritis macrocera*	*Plectritis macrocera*	PLMA
I	A	*Ranunculus testiculatus*	***Ceratocephala testiculata***	RATE
I	A	*Salsola kali*	***Salsola tragus***	SAKA
N	P	*Sedum leibergii*	*Sedum leibergii*	SELE
N	P	*Senecio integerrimus*	*Senecio integerrimus*	SEIN
N	P	*Silene douglasii*	*Silene douglasii*	SIDO
I	A	*Sisymbrium altissimum*	*Sisymbrium altissimum*	SIAL
N	P	*Sphaeralcea munroana*	*Sphaeralcea munroana*	SPMU

Continued

N / I	A / B / P	Hitchcock and Cronquist (1973)	Kartesz (1999)	Field Code
I	P	*no record	***Sphaerophysa salsula***	SPSA
N	P	*Stephanomeria tenuifolia*	***Stephanomeria minor***	STTE
I	AB	*Tragopogon dubius*	*Tragopogon dubius*	TRDU
N	P	*Zigadenus venenosus*	*Zigadenus venenosus*	ZIVE

SOURCES

Hitchcock, C.L., and A. Cronquist. 1973. Flora of the Pacific Northwest. University of Washington Press, Seattle, WA.

Kartesz, J.T.. 1999. A Synonymized Checklist and Atlas with Biological Attributes for the Vascular Flora of the United States, Canada, and Greenland. First edition. In J.T. Kartesz and C.A. Meacham, Synthesis of the North American Flora. CD-ROM version 1.0. North Carolina Botanical Garden, Chapel Hill, NC.

Sackschewsky, M.R., and J.L. Downs. 2001. Vascular Plants of the Hanford Site. PNNL-13688. Pacific Northwest National Laboratory, Richland, WA.
Available online at http://www. pnl.gov/ecology/Library/PNNL13688.pdf