# TIN CUP FIRE RECOVERY REPORT

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# ACKNOWLEDGEMENTS

This project was the result of the collaborative efforts of many people and organizations. The project was initiated by discussions between Robert D. Jones (DOE-ID), Mike Pellant (BLM) and others following the Tin Cup Fire that burned in 2000 on BLM and DOE managed lands. In the spring of 2001, a proposal team was assembled to prepare a proposal for the Joint Fire Sciences Program. That team included the following members.

- Robert D. Jones, Department of Energy – Idaho Operations Office
- Mike Pellant, Bureau of Land Management, Idaho State Office
- Greg Dawson, Bureau of Land Management, Idaho Falls Office
- Jay E. Anderson, Idaho State University.
- Julie Hilty, Boise State University
- Gregory J. White, Idaho National Laboratory
- Roger D. Blew, S.M. Stoller Corp – INL Environmental Surveillance, Education and Research Program

In the fall of 2001, the proposal team was expanded and submitted a proposal to The Nature Conservancy's Rodney Johnson and Katherine Ordway Stewardship Endowment. Additions to the team for that proposal included:

- Alan Sands, The Nature Conservancy
- Trish Klahr, The Nature Conservancy

Additional researchers involved in the final design of the project included,

- Amy D. Forman, S.M. Stoller Corp – INL Environmental Surveillance, Education and Research Program
- Robert Unnasch, The Nature Conservancy

A review of the experimental design and statistical methods was provided by Charles R. Bonham, Uniscale and CSU (emeritus).

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This report is dedicated to the memory of two team members lost during and soon after completion of this research. Jay E. Anderson provided mentoring, leadership and guidance to RDB and ADF, and provided the basic hypotheses on which this research was based. Robert D. Jones was an avid supporter of the Idaho National Environmental Research Park and developed numerous collaborations of scientists to make use of the Research Park, including this project. He recognized the intimate linkages between the natural world and our nation’s energy security, and worked tirelessly to support the science necessary to understand both.

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***Roger D. Blew and Amy D. Forman***

## Introduction

The BLM and other managers of Idaho rangelands must decide whether burned areas need stabilization and rehabilitation treatments to prevent soil erosion and inhibit the invasion of exotic species such as cheatgrass (*Bromus tectorum*). Most of these rangelands have historically been dominated by big sagebrush (*Artemisia tridentata),* which does not re-sprout after fire. Sagebrush provides critical food and habitat for sage grouse, a species proposed for listing under the Endangered Species Act. With the accelerating loss of native sagebrush communities and habitat for sage grouse and other sagebrush-obligate species, sagebrush reseeding following fire has become an important consideration, as has the issue of livestock grazing following fire on recovering native vegetation and seeded areas. Between 1994 and 2000, approximately 70% of the sage grouse habitat in eastern Idaho's Big Desert has been burned by wildfire. Fire suppression and rehabilitation costs are rising, and the threats to human life and property are increasing in eastern Idaho.

The study took place on the Idaho National Laboratory (INL), located in the Upper Snake River Plain of southeast Idaho. The INL has experienced numerous, well-documented fires. Additionally, many of the sagebrush-dominated plant communities on the INL have high native species richness, and low incidence of communities dominated by exotic species. Plant communities on the INL have also experienced a range of grazing histories, including allotments that currently allow multiple permitees and livestock classes to those areas that have not been grazed by livestock for approximately sixty years. Opportunities to evaluate the natural progress of succession on good condition sagebrush steppe rangelands after a wildfire are very rare in southern Idaho due to the increasing dominance of cheatgrass and other invasive species. Thus, the INL is an ideal location for studying natural post-fire recovery, as well as the effects of rehabilitation treatments like reseeding and livestock grazing on plant community recovery in high quality sagebrush steppe rangelands.

This study has two objectives that address management concerns relative to natural plant community recovery following fire in good ecological condition rangeland. These objectives include assessing the changes in plant communities in the years following fire and those factors associated with the return of *A. tridentata* to burned areas.

A study of short-term (up to four years post-fire) plant community change was designed to provide information that could be applied to direct rehabilitation actions immediately following a fire. A long-term study covering a chronosequence of fires up to about 90 years since fire was designed to provide insight into the role of fire in the overall dynamics of sagebrush communities, how fire functions as a transition mechanism between states, and the potential for fire to cause a transition across a threshold to an undesirable, permanent state.

Return of *A. tridentata* to a burned area was assessed in three types of studies. The first was to examine the efficiency (economical as well as biological) of artificially seeding sagebrush on a recently burned area. The second was designed to examine the potential for improving aerial seeding by planting an area that had burned several years earlier that had already developed a perennial grass and forb community. The third study examined natural patterns of A. tridentata invasion onto burned areas.

## Objectives

***Objective 1:*** **Describe post-wildfire trajectories in community composition and structure in high quality sagebrush steppe rangeland.**

Most of the remaining sagebrush steppe ecosystem is not in good ecological condition. West (2000) reported that 75 percent of remaining sagebrush steppe was in one of three successional states: 1) Depauperate Late Seral Sagebrush Steppe, 2) Brush With Introduced Annuals in understory, or 3) Cheatgrass/Medusahead. Much of what is presently known about sagebrush steppe recovery from wildfire is based on studies conducted on those successional states. Fires in these sites generally drive them to annual grass domination or reinforce that dominance. Management planning for these sites generally means emergency fire rehabilitation by seeding introduced or native perennial species.

West (2000) also reported that only one percent of remaining sagebrush steppe was in Late Seral Sagebrush Steppe. Sites in this successional state are usually relict sites that have restricted use by livestock because of terrain, lack of water or are designated special use sites. Establishment of the INL in 1950 set aside the largest existing reserve of this ecosystem type and provides a unique opportunity for this kind of study.

Anderson and Inouye (2001) reported that total vascular plant cover on plots in an area removed from livestock grazing on the INL in 1995 was 38 percent and was nearly evenly divided between shrubs and herbaceous understory plants. They concluded that the composition and cover of plant communities on the INL are similar to that of nearby ungrazed kipukas (Tisdale et al. 1965, Passey et al. 1982), which are considered to be relatively pristine.

The INL has experienced a number of large wildfires since 1994. Data collected on areas burned in 1994 and 1995 showed rapid re-establishment of native herbaceous species within two to three years following fire (Buckwalter 2002). Similar post-fire native species cover was described on sagebrush rangeland near Pocatello, Idaho (Ratzlaff and Anderson 1995).

This is in contrast to those areas where the perennial understory has been replaced by exotic annuals. In areas dominated by annuals, fire is likely to promote only more annuals (West and Hassan 1985). Return of perennials on such sites generally requires intervention by artificial seeding.

Most fire recovery studies have looked only at the short-term aspects of recovery and may only extend to two or three years after fire. There have been very few studies addressing the long-term patterns of vegetation change following fire in sagebrush steppe (Humphrey 1984, Neele et al 2000, and West and Yorks 2002). In this study we considered vegetation change on a chronosequence of fires covering a period of more than 90 years. A better understanding of long-term vegetation dynamics following fire could result in strategies that direct vegetation change in a direction favorable to land management goals. The results of the long-term recovery patterns will be considered in light of the proposed successional models for managing sagebrush steppe. These include the classical rangeland climax model based on Clements' climatic climax theory (Dyksterhuis 1949) and the more recently developed state and transition models (West 2000 and Stringham et al 2003).

This objective is addressed in Section 2 "Vegetation change following fire on high quality sagebrush steppe rangeland in southeastern Idaho" and Section 3 "Long-term patterns of vegetation change following fire in sagebrush steppe." Section 6 "Vegetation cover sampling methods on the Tin Cup Plots" addresses the methods used for measuring and analyzing vegetative cover data used in Section 2.

***Objective 2:* Examine the return of *A. tridentata* onto burned areas through both natural processes and following artificial seeding during the first ten years following fire.**

Although the 1994 and 1995 fires on the INL were dominated by native perennials through at least 1998, big sagebrush (*A. tridentata*) remained absent (Buckwalter 2002). Big sagebrush does not re-sprout following fire and must regenerate from seed. Meyer (1994) listed a number of factors that limit natural regeneration of big sagebrush after a fire. Wyoming big sagebrush is the most common subspecies at the INL.

Big sagebrush seeds typically germinate in the first spring following their dispersal and are reported to not form a persistent seed bank in the soil (Young and Evans 1989). Seed dispersal is generally limited to about 30 m and most seed falls within 1 m of the parent plant (Meyer 1994). Many of the recent fires burned completely, leaving few or no remnant big sagebrush plants to act as a source for seed dispersal. Because the recent fires have been quite large, it has been suggested that decades may be required for natural seed dispersal to repopulate the burned areas with sagebrush.

Soil microtopography controls the amount and quality of safe-sites available for seedling establishment and is strongly influenced by the presence of biological soil crusts. Even though germination is regulated to coincide with conditions that favor seedling growth (Meyer and Monson 1992), most seedlings are not recruited into the population because of adverse biotic and abiotic environmental conditions (Meyer 1994). The primary factor limiting establishment is weather, with drought being the principal cause of mortality. This is especially important with Wyoming big sagebrush because seedlings must endure dry summers annually, but they need sufficient time in the first growing season to become drought hardy (Meyer 1994).

Artificially seeding big sagebrush after a fire overcomes the problems of a limited seed bank and dispersal distances. However, selecting plant material that is not site adapted may lead to eventual failure. Since germination is regulated to coincide with conditions that favor seedling growth (Meyer and Monsen 1992), using seed that germinates too soon or too late can result in failure of the entire seeding. Monsen (2000) reported that plantings of Utah-origin big sagebrush in Idaho established well; however, the stands would likely fail because no natural recruitment was recorded for at least 20 years.

To address these concerns regarding *A. tridentata* return on burned areas, we conducted three studies. The first was to investigate the efficiency (economical as well as biological) of artificially seeding sagebrush. This was accomplished on the Tin Cup fire which burned in 2000 and was aerially seeded over snow cover in February 2001. We also studied the effectiveness of seeding on an already established stand of native perennial grasses and forbs seven years after fire. Finally, we studied the natural recruitment patterns of *A. tridentata* on areas that had burned in the preceding 10 years to determine if seed availability on the burn was a limiting factor.

This objective is addressed in Section 4 "Effectiveness of aerial seeding of Wyoming big sagebrush following fire in southeast Idaho" and Section 5 "Natural recruitment patterns in Wyoming big sagebrush following fire in southeast Idaho."

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# SECTION 2: SAGEBRUSH STEPPE VEGETATION CHANGE DURING THE FIRST FOUR YEARS FOLLOWING WILDLAND FIRE IN SOUTHEAST IDAHO.

***Roger D. Blew, Amy D. Forman, Jackie R. Hafla, Mike Pellant***

## Introduction

Historically, fire was both a natural ecosystem process and an infrequent event in the sagebrush steppe. Fire rotation intervals in sagebrush steppe dominated by Wyoming big sagebrush, are on the order of 100 to 240 years (Baker 2006). The inability of sagebrush to resprout following fire defines the fire ecology of these ecosystems as stand replacing with regard to sagebrush (Paysen et al. 2000, Keeley 2006). The long-term loss of sagebrush on a burned area, however, does not necessarily lead to a depauperate plant community. Other native perennial species recover by re-sprouting with little apparent negative effect due to fire, or the effects are only short lived. Humphrey (1984) noted that perennial forbs and grasses were the dominant growth forms during the first few years following fire in big sagebrush and Utah juniper sites in southeast Idaho. Humphrey (1984) also reported only a gradual increase in shrub dominance after those first few years. Other reports of fire effects in sagebrush steppe in southeast Idaho indicate that a few years post-fire, plant community composition is very similar to the pre-fire community with only the exception of sagebrush (Ratzlaff and Anderson 1995, Anderson and Inouye 2001, Buckwalter 2002).

The increasing risk of invasion and dominance by annual grasses, however, has the potential to change the recovery trajectory of sagebrush steppe rangelands throughout their range. Consequently, the need for more detailed information about natural recovery processes is clear. This is especially true for rangelands that have burned and have the potential for either unassisted recovery of native species or are at risk for transition to annual grass dominated sites. Understanding natural recovery processes may be key to setting a management strategy more likely to result in the former rather than the latter. The current risk of significant loss of the sagebrush steppe ecosystem to invasive species further emphasizes the need for successful, cost-effective restoration strategies.

One of the most immediate rangeland management issues affected by wildland fire and cheatgrass invasion is maintaining adequate habitat for sensitive sagebrush obligate species. Sagebrush provides critical food and habitat for sage grouse, a species proposed for listing under the Endangered Species Act. Between 1994 and 2000, approximately 70% of the sage grouse habitat in eastern Idaho's Big Desert was burned by wildfire. Therefore, the BLM and other managers of Idaho rangelands are under increasing pressure to actively manage the region for sagebrush habitat. With the accelerating loss of native sagebrush steppe communities and habitat for sage grouse and other sagebrush-obligate species, land managers must decide whether burned areas need stabilization and rehabilitation treatments to prevent soil erosion and inhibit the invasion of exotic species such as cheatgrass (*Bromus tectorum*). Thus, replanting following fire has become an important habitat management consideration, as has the issue of livestock grazing following fire on recovering native vegetation and seeded areas.

Managers are often faced with conflicting information on how best to manage fire recovery to promote habitat health. Recently several writers have offered suggestions on how to manage sagebrush rangeland after fire to promote recovery. Sanders (2000) suggested that burned areas should be treated using rest-rotation grazing principles as for low condition range by grazing after seed ripe the first growing season following fire (if adequate stands of perennial grasses survived the fire) to increase seed germination by providing a mechanism for covering seed with mineral soil. The hypothesis behind this suggestion is to improve seed germination to provide more seedlings to support grass recovery. Others have indicated that most vegetative recovery within the first few years post-fire results from resprouting perennial species (Ratzlaff and Anderson 1995 and Buckwalter 2002). Furthermore, soil disturbance from planting or livestock may harm the resprouting perennial species already present and hinder the recovery process. Because of these conflicting recommendations, it remains unclear if short-term sagebrush-grass recovery following fire requires or can be improved by livestock grazing.

Nearly all recent fires in sagebrush steppe rangeland are in areas that have been degraded and have undergone some degree of invasion to annual grasses. Much of our land management response to fire has been guided by observations following fires in these degraded rangelands. Allen-Diaz and Bartolome (1998) noted that most of our knowledge of the behavior of rangelands is based on studies of deteriorated rangeland and is more about processes associated with rangeland decline. They called for studies of the responses of rangelands to natural disturbances and changes in management.

West (2000) reported that only one percent of remaining sagebrush steppe was in Late Seral Sagebrush Steppe. Perhaps because so little good condition sagebrush steppe remains, little emphasis has been placed on understanding the role of fire and its effects on native vegetation in sagebrush steppe. Thus, the Tin Cup Fire Recovery Study was designed to better address post-fire land management recovery strategies by better understanding natural recovery processes. The specific purpose of this study was to assess the short-term changes in vegetation following wildland fire in high quality sagebrush steppe rangeland in the absence of livestock grazing. We were interested in understanding whether the post-burn community had adequate natural recovery without seeding and whether fire followed by grazing exclusion results in an unacceptable trajectory.

## Methods

### *Study Site*

The study was located on the Idaho National Laboratory (INL) in an area burned by the Tin Cup Fire in late July of 2000 (43º08’N 113º33’W). The fire burned 14358 ha (35480 acres) most of which was high severity burn as evidenced by the high consumption of live plants. Mean annual precipitation is 220mm and is predominantly snow. However, May and June are the wettest months and during that time precipitation is rain.

Soils on this area are primarily loess or sand over basalt. Soil mapping units include Nargon-Deuce-Lava Flows complex, Atom Silt Loam, and Malm-Bondfarm-Mathesen complex (Olson et al. 1995). The project area is within a BLM grazing allotment.

Prior to the fire, the vegetation on the area was primarily Wyoming big sagebrush (*Artemisia tridentata* spp. *wyomingensis*), bluebunch wheatgrass (*Pseudoroegnaria spicata* spp. *spicata*), and green rabbitbrush (*Chrysothamnus viscidiflorus*). Anderson and Inouye (2001) concluded that the composition and cover of plant communities on the INL are similar to that of nearby ungrazed kipukas (Tisdale et al. 1965, Passey et al. 1982), which are considered to be pristine. Classical range succession models would consider this site to have been in excellent condition. The state and transition model of Laycock (1991) would put this in Stage I (open stand of sagebrush with productive herbaceous perennial understory). This site could be described as being in a "Relictual Condition" (West 2000) as a remnant that has had little direct impact from recent human influences.

### *Sampling Methods*

We established ten plots separated by about 1 km. Around the plots, we constructed livestock exclosures 100m by 100m in size. The exclosures were three-strand fences with the upper two of barbed wire and the bottom one smooth wire to allow pronghorn passage.

Density, frequency and species richness were measured using Modified Whittaker plots (Stohlgren et al. 1995). These plots are 20m X 50m in size and contain nested subplots. The subplots include ten 0.5m X 2.0m, two 2m X 5m, and one 5m X 20m subplots. These plots were surveyed by doing cumulative species counts in each subplot and finally over the entire plot. This nested sampling design allows for estimation of total species richness using species area curves. Density and frequency were measured in the 0.5m X 2.0 m plots.

The point sighting frame method of Floyd and Anderson (1987) was used for measuring cover. We measured aerial cover of shrubs and forbs, and basal cover of grasses. Cover by litter, lichen, moss, rock and bare ground were also recorded. We used the same 20m X 50m plot described above for species richness sampling to guide sighting frame placement for cover measurements. Ten transects were established perpendicular to the long edge of each plot at 5.0 m intervals starting at 2.5 m from the plot corner. Five contiguous sighting frames were read on each of the ten transects. The starting point on each transect was randomly selected. The point sighting frame measurements were repeated in the same locations each year.

### *Data Analysis*

Cover, density, and species richness data were analyzed using One-Way Repeated Measures Analyses of Variance to compare changes in mean values over the three growing seasons during which the study was conducted (Zar 1999). Data did not meet assumptions of normality in some circumstances, and could not be transformed to do so. Because ANOVAs tend to be robust, and because we were more interested in identifying potential relationships among growing seasons than in the absolute statistical certainty of those relationships, we proceeded with analyses of untransformed data, even when the data did not meet assumptions of normality. Assumptions of equal variance were met, however, for all of the ANOVAs performed. We assigned statistical significance at the P = 0.05 level. Multiple comparisons were analyzed using the Holm-Sidak method (Maxwell and Delaney 1990).

## Results

Total absolute vascular plant cover nearly doubled during the study period, ranging from an average of 11.5 % in 2002 to an average of 20.1% in 2004 (Figure 2-1). Total plant cover was dominated by native perennial species, with nearly half of average total cover resulting from resprouting shrubs. After native resprouting shrubs (*A. tridentata* was not present), the functional group with the greatest average cover was native perennial forbs, followed by native perennial grasses. Relative to perennial species, cover by native and introduced annual and biennial species contributed little to total plant cover, especially in 2002 and 2003.

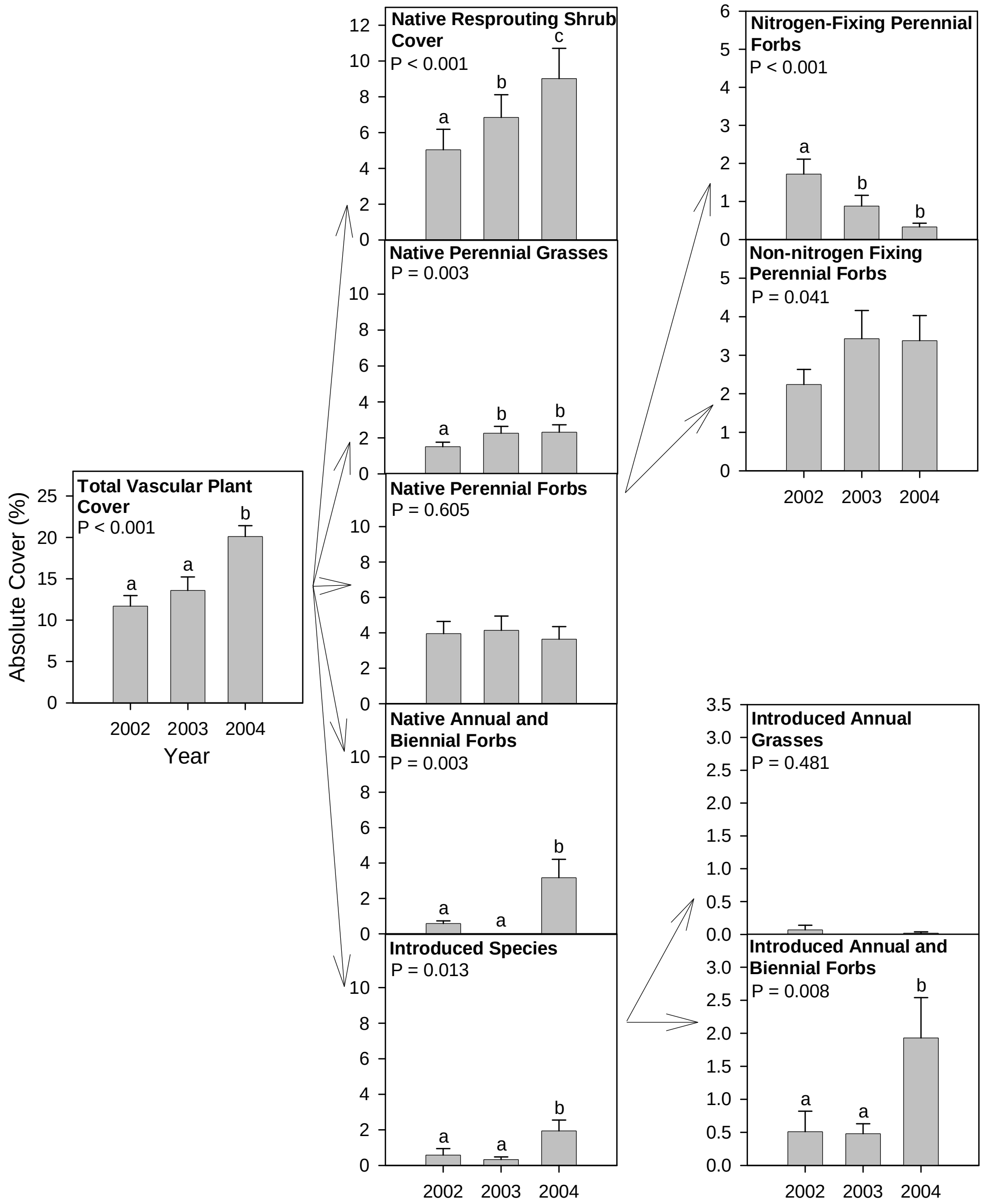

Figure 2-1. Mean absolute plant cover for various functional groups two to four growing seasons post-fire on the Tin Cup burn.

Native perennial grass cover increased significantly between 2002 and 2003 and total perennial forb cover remained constant throughout the study period (Figure 2-1). Within the native perennial forbs functional group, however, cover of legumes decreased significantly through the study period, while cover of non-leguminous perennial forbs remained unchanged. Average cover of native resprouting shrubs increased significantly each year of the study period. However, a significant increase in total vascular plant cover occurred only between 2003 and 2004. This increase in total cover was due to the significant increase in shrub cover between those years, as well as significant increases in the cover of two other functional groups, native annual and biennial species and introduced annual and biennial species. Although cover of native and introduced annual and biennial species relative to cover of native perennial species was low throughout the study period, annual and biennial species did have significantly greater cover in 2004 than during the previous years of the study, which contributed to the significant increase in total vegetation between 2003 and 2004.

All of the vegetation cover from the native annual and biennial species functional group was from forbs, and the vast majority of cover from forbs in that functional group was due to annuals, rather than biennials (Figure 2-1). Thus, significant changes in cover of this functional group can be attributed primarily to annual forbs. Likewise, the significant increase in cover of introduced species from 2003 to 2004 was due primarily to the annual forb species in that functional group. Introduced annual grass cover was quite low and did not change significantly throughout the study period.

Ten native annual forbs and five introduced annual forbs were counted in the density frames during density/frequency sampling in at least two of the three years density data were collected (Table 2-1). All ten native annual forb species were absent from the density frames sampled in 2003; only two introduced annual forb species were absent during the same time period. Annual forb density was highly variable both spatially and temporally throughout the study period. At the plot level, the large spatial variability of each species is evidenced by the high standard deviation values associated each species' mean density value. In many cases, the standard deviation is two to three times greater than the mean density for an individual annual species. The high temporal variability in annual species density is demonstrated by the large differences of individual species' means from one year to the next. In many cases, the mean density of a native or introduced annual species varies by an order of magnitude over two sampling periods. However, these large differences in annual mean densities were not statistically significant for many species. The lack of statistical significance is likely due to the large spatial variation in species density discussed above.

Four native annual species did have statistically significant increases in density from 2003 to 2004, and three of those same species had statistically significant increases from 2002 to 2004 (Table 2-1). Although significant differences in density from year to year occurred in only a handful of annual species, mean density did show a trend upward from 2002 and 2003 to 2004 in nine of the native annual species and four of the introduced annual species. The trend of greater densities in native and introduced annual species in 2004 coincides with significantly greater cover of those functional groups in 2004. Thus, increases in annual species cover in 2004 are primarily due to increases in the density and distribution of individuals rather than in the size of individuals (Figure 2-1 and Table 2-1). Although not all native annual species showed statistically significant differences in density between years, it is important to note that the density of all native annual species was zero in 2003.

**Table 2-1. Mean density (individuals per m$^2$) and standard deviation for ten native annual forbs and five introduced annual forbs during three seasons of data collection on the Tin Cup burn.**

| | 2002 Mean | 2002 SD | 2003 Mean | 2003 SD | 2004 Mean | 2004 SD | P Value |
|---|---|---|---|---|---|---|---|
| **Native Annuals** | | | | | | | |
| *Chenopodium leptophyllum* | 0.20 | 0.29 | 0.00 | 0.00 | 7.65 | 9.99 | **0.011** |
| | **a** | | **a** | | **b** | | |
| *Descurainia pinnata* | 0.13 | 0.27 | 0.00 | 0.00 | 2.16 | 1.37 | 0.118 |
| *Eriogonum cernuum* | 0.01 | 0.03 | 0.00 | 0.00 | 0.34 | 0.54 | **0.042** |
| *Eriastrum wilcoxii* | 0.54 | 0.66 | 0.00 | 0.00 | 6.22 | 7.76 | **0.009** |
| | **a** | | **a** | | **b** | | |
| *Gayophytum diffusum* | 0.11 | 0.31 | 0.00 | 0.00 | 0.78 | 1.16 | **0.032** |
| | **ab** | | **a** | | **b** | | |
| *Gayophytum ramosissimum* | 0.03 | 0.09 | 0.00 | 0.00 | 0.06 | 0.19 | 0.573 |
| *Gilia leptomeria* | 0.18 | 0.29 | 0.00 | 0.00 | 4.50 | 9.38 | 0.136 |
| *Lappula occidentalis* | 1.96 | 5.55 | 0.00 | 0.00 | 1.61 | 2.18 | 0.322 |
| *Mentzelia albicaulis* | 0.16 | 0.27 | 0.00 | 0.00 | 2.32 | 2.98 | **0.013** |
| | **a** | | **a** | | **b** | | |
| *Townsendia florifer* | 0.01 | 0.03 | 0.00 | 0.00 | 0.03 | 0.07 | 0.327 |
| **Introduced Annuals** | | | | | | | |
| *Alyssum desertorum* | 25.01 | 77.90 | 3.83 | 9.84 | 3.14 | 6.38 | 0.48 |
| *Descurainia sophia* | 0.02 | 0.06 | 0.00 | 0.00 | 0.03 | 0.03 | 0.608 |
| *Halogeton glomeratus* | 0.01 | 0.03 | 0.22 | 0.70 | 2.70 | 8.54 | 0.389 |
| *Salsola kali* | 0.15 | 0.24 | 7.17 | 17.77 | 18.93 | 39.34 | 0.129 |
| *Sisymbrium altissimum* | 0.01 | 0.03 | 0.00 | 0.00 | 0.07 | 0.22 | 0.387 |

With the exception of *Poa secunda*, densities of perennial graminoids did not change significantly during the study (Table 2-2). *Poa secunda* density increased from 0.05 in 2002 to 0.22 in 2004.

Average plot-level species richness was significantly higher in 2002 and 2004 than in 2003 (Figure 2-2). Within vegetation functional groups, species richness remained remarkably stable from year to year for native shrubs, native perennial grasses, and introduced species. Species richness dropped slightly, but significantly from 2002 to 2003 for native perennial forbs. This drop in average species richness did not appear to be attributable to any particular species or plots. The greatest annual variation in species richness was due to native annual and biennial forbs, which differed significantly each year of the study.

**Table 2-2. Mean density for native perennial grasses on the Tin Cup burn two to four growing seasons post-fire. Bunch grass densities were calculated as number of bunches per m², and rhizomatous grass densities were calculated as number of tillers per m².**

| | 2002 | 2003 | 2004 | P Value |
|---|---|---|---|---|
| **Bunch Graminoids** | | | | |
| *Achnatherum hymenoides* | 0.16 | 0.13 | 0.08 | 0.201 |
| *Elymus elymoides* | 0.17 | 0.56 | 0.22 | 0.235 |
| *Hesperostipa comata* | 0.35 | 0.48 | 0.25 | 0.387 |
| *Poa secunda* | 0.05 | 0.17 | 0.22 | **0.022** |
| | **a** | **ab** | **b** | |
| *Pseudoroegneria spicata* | 2.23 | 1.66 | 2.35 | 0.204 |
| **Rhizomatous Graminoids** | | | | |
| *Carex douglasii* | 2.01 | 3.57 | 2.42 | 0.297 |
| *Elymus lanceolatus* | 20.99 | 40.29 | 25.40 | 0.275 |
| *Pascopyrum smithii* | 7.04 | 3.34 | 1.53 | 0.532 |

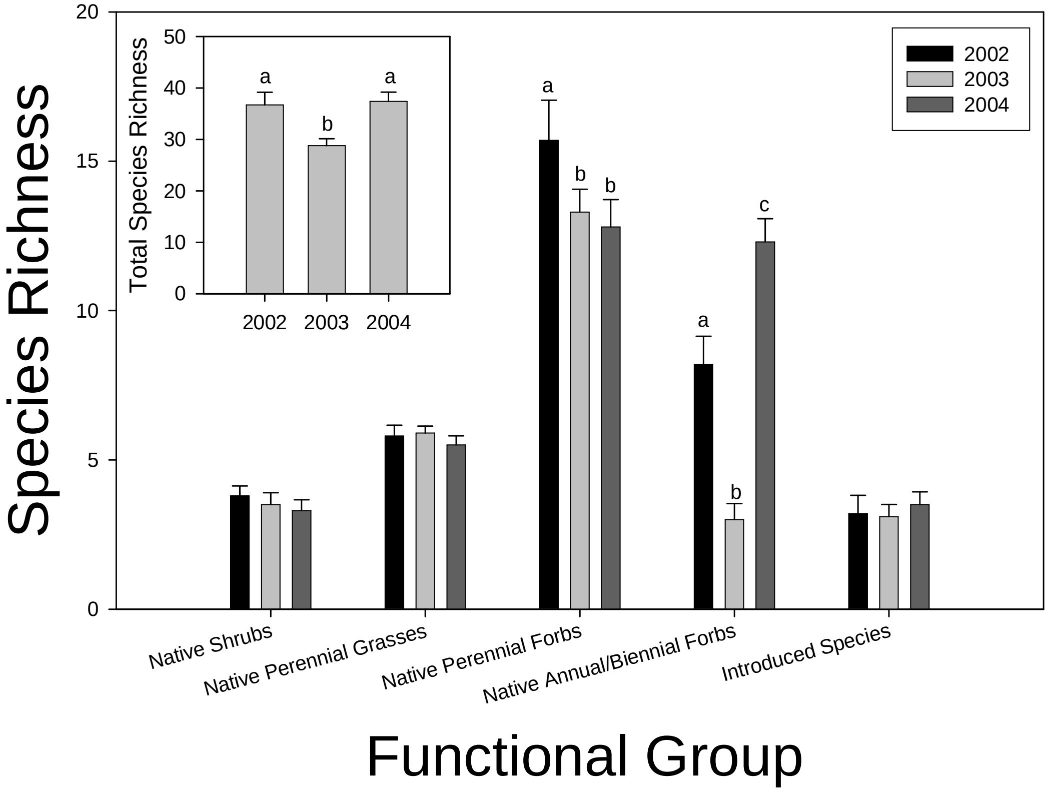

**Figure 2-2. Mean species richness for various functional groups two to four growing seasons post-fire on the Tin Cup burn.**

Annual precipitation was well below average during the year in which the Tin Cup fire burned (2000) and it remain below average several years post-fire, including the three years during which the study was conducted (Figure 2-3). In fact, total annual precipitation during two years of data collection, 2002 and 2003, was less than one half average total precipitation. Considering the severity of the post-fire drought conditions, the significant increases in native shrub cover each year of the study were remarkable. Likewise, the small but significant increase in the basal cover of perennial grasses between 2002 and 2003 (Figure 2-1) was notable given that 2003 was the driest year during the 64 years on record.

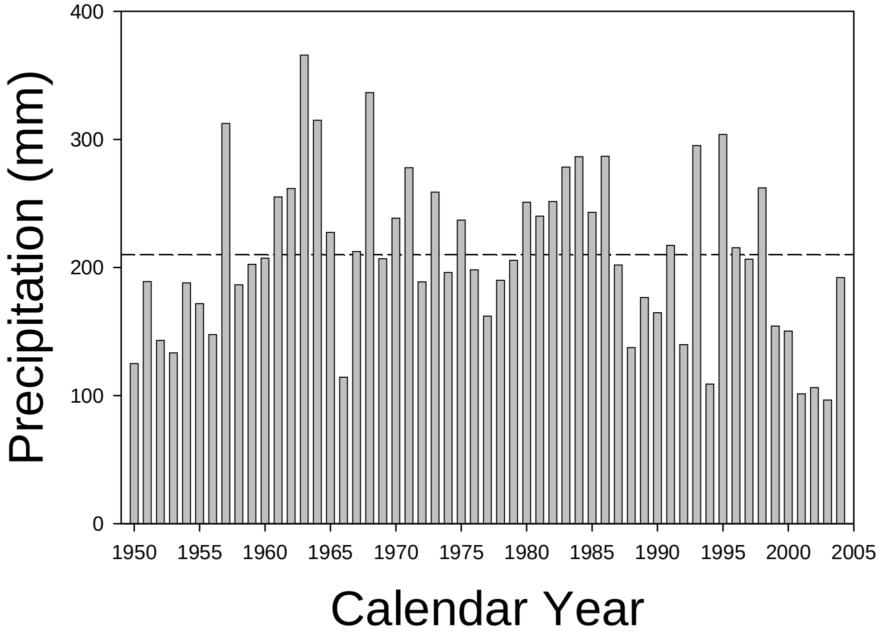

**Figure 2-3. Annual precipitation from the Central Facilities Area, Idaho National Laboratory (NOAA data) from 1950 through 2004. The dashed line represents average annual precipitation during the same 64 year period.**

Annual precipitation patterns were quite similar in 2002 and 2003, with most months only receiving about one half of long-term average monthly precipitation (Figure 2-4). Annual precipitation patterns through May of 2004 were also similar to the patterns of the previous two years. Of the three study years, 2003 was actually the wettest year and 2004 was the driest year through May. Beginning in June, though, dramatic changes in precipitation patterns between years occurred. In 2002 and 2004, total June precipitation was slightly below average, but less that one millimeter of rain fell in June of 2003 (Figure 2-5). During the month of July, no precipitation occurred in 2003 and the total precipitation that fell in July of 2002 was a fraction of the average. Total average monthly precipitation for July 2004, however, was more than double the long-term average for that month. Thus, although annual precipitation in the years during which the study was conducted was well below the long-term annual average, the timing of the occurrence precipitation during each year was unique.

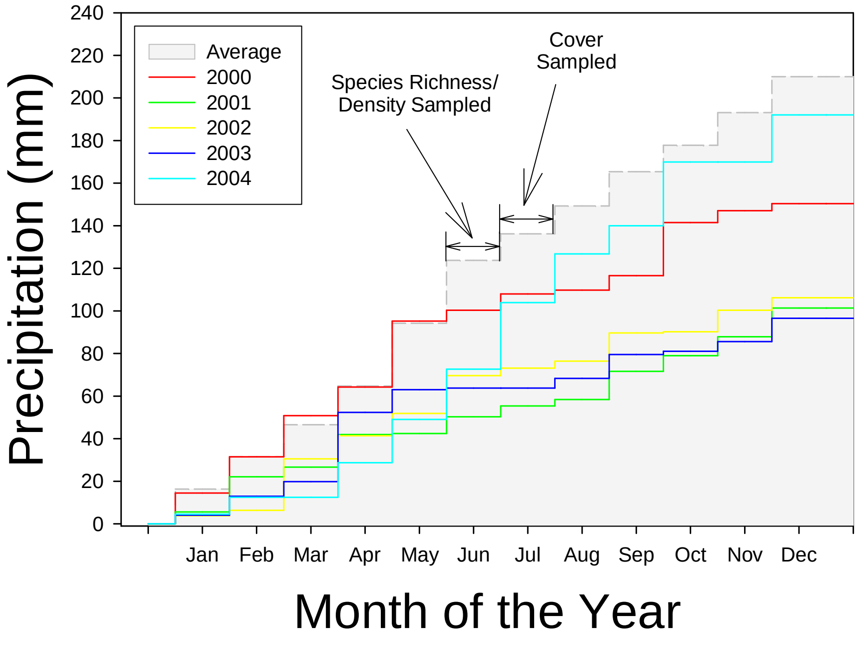

**Figure 2-4. Cumulative annual precipitation totaled by month, from the Central Facilities Area, Idaho National Laboratory (NOAA data) during the year of the fire through the final year of the study period. The dashed line represents cumulative annual precipitation using monthly averages from 1950 through 2004.**

## Discussion

### *Grasses*

Initial post-fire total plant cover was similar to that reported by Buckwalter (2002) on the INL in an area that burned in 1995. Perennial grass basal cover did not increase after 2003. West and Hassan (1985) reported that perennial grasses recovered to pre-burn conditions following two years. The basal cover results indicate the response of the grasses to fire must have been vegetative since seedlings likely would have continued to increase in basal cover for more than two to three years following germination. Because basal cover values following fire were similar to that reported on unburned sites elsewhere on the INL (Anderson and Inouye 2001), it is unlikely that seedlings could have reached the same basal cover without a concomitant significant increase in density. The lack of an increase in density of the perennial grass species other than *P. secunda* (a minor component of total perennial grass basal cover) indicates that the increases in perennial grass basal cover associated with response following fire are not driven by recruitment, but rather from resprouting.

Young and Evans (1978) also reported that the density of native perennial bunchgrasses, with the exception of *Poa secunda*, remained stable during the first four years after fire in sagebrush steppe. In that study *B. tectorum* increased exponentially during those same four years. Young and Evans (1978) concluded that “downy brome has the inherent ability to close seral communities to seedlings of perennial grasses.” Given that non-native annual grasses were a minor component of the plant community in the present study, our results suggest that the lack of seedlings of perennial grasses in the early post-fire period may not be the result of interspecific competition, but rather it is perhaps a characteristic of the natural history of these grasses that recruitment is limited to very specific environmental conditions. Hassan and West (1986) reported few seeds of native bunchgrasses and *Artemisia* spp. in both burned and unburned soils. Akinsoji (1988) also reported low propagule reserves of shrubs, forbs and perennial grasses in burned and unburned sagebrush steppe in Idaho.

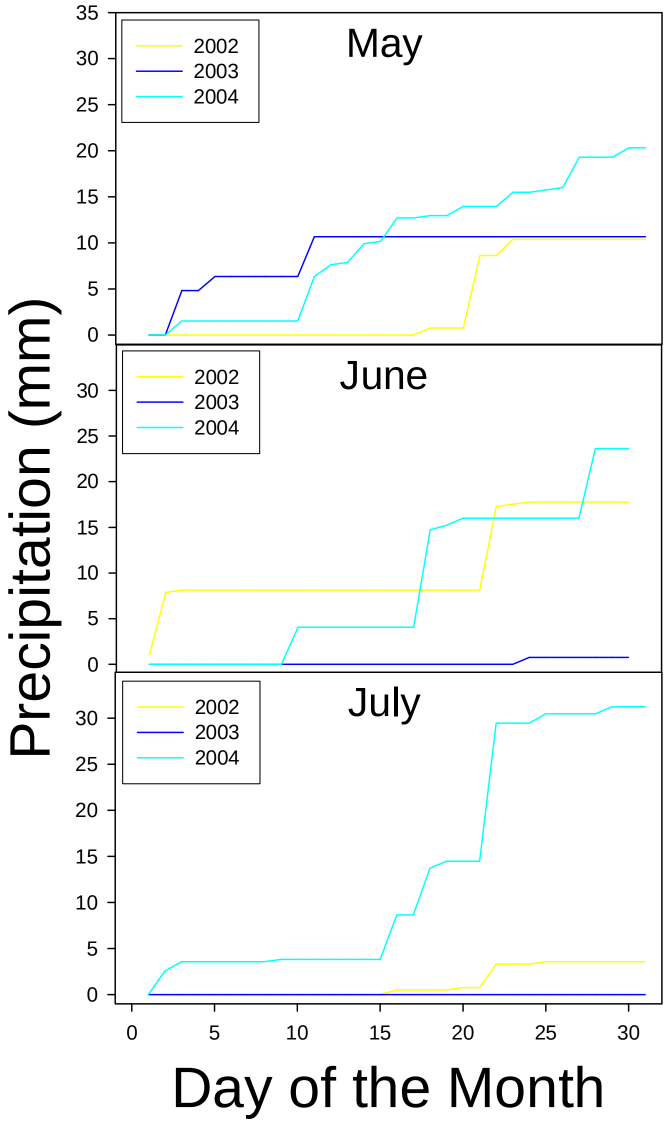

**Figure 2-5. Cumulative monthly precipitation, using daily totals, for May, June, and July 2002 through 2004. Precipitation data are from Central Facilities Area, Idaho National Laboratory (NOAA data).**

There have been few studies of natural recruitment of native perennial plants in sagebrush steppe communities (West et al. 1979) and this represents an important aspect of sagebrush steppe ecology that needs be better understood. However, West et al (1979) did conclude that the stability of plant communities is the result of plasticity in the plants already established rather than through changes in age-class distribution, which we interpret to include recruitment. Since the response of a healthy community to fire does not include recruitment, rehabilitation strategies based on improving seed germination are not necessary. Instead, management should focus on strategies that promote post-fire vigor in native perennial grasses.

### *Forbs*

Perennial forbs did not increase in cover during the study. This suggests that before the third year following fire, they had completed recovery.

The post-fire flush of legumes was not expected, but may provide clues to the role of nitrogen availability in the response of the plant community to fire or, conversely, the role the immediate post-fire plant community plays in nitrogen cycling in sagebrush steppe. We are not aware of any other similar reports for a flush of legume species following fire. Johnson and Rumbaugh (1981) reported on nodulation and nitrogenase activity in legumes in the sagebrush steppe. Although they found few species with nodules, the utility of their results are somewhat limited because they sampled during the driest part of the growing season. Unfortunately, this appears to be the only research on nodulation in legumes from sagebrush steppe in North America.

The role of native legumes in the nitrogen cycle of rangelands globally has not been well researched either. The most comprehensive survey of nodulation in rangeland legumes appears to have been done in Pakistan (Athar 2005). After surveying 161 species of legumes native to Pakistan, all were found to be nodulated except for the nine species representing the subfamily Caesalpinioideae. Given that of the 409 vascular plant taxa recorded on the INL 31 are native legumes (Anderson et al. 1996), it is likely they play an important role in the nitrogen cycle of the sagebrush steppe. The results of Athar (2005) and the ubiquity of the Fabaceae in the sagebrush steppe suggest that a renewed effort to understand the ecology of native legumes of the sagebrush steppe is warranted. The early flush of legumes suggests the immediate post-fire period is important for nitrogen input to sagebrush steppe.

### *Annual Species*

Annuals and biennials contributed little toward total cover and did not play a role in defining a community-level post-fire trajectory. Humphrey (1984) reported that annuals were not particularly prominent in the first few years after fire in southeastern Idaho. He reported *Bromus tectorum* was abundant on a site three years after fire, but rare on sites two and six years after fire. However, Young and Evans (1978) reported that native annual forbs had equal to or greater frequency than *B. tectorum* during the first year following fire, but were nearly absent by years three and four. They concluded that competition from *B. tecorum* was responsible for the decline in native annual species. Native annual species did not reflect any directional change in the plant community in our study and similar results were reported by Humphrey (1984). Native annuals were present only in 2002 and 2004 and were completely absent from the density plots in 2003 (Table 2-1). Their presence appeared to be associated with the amount and timing of precipitation. This indicates that they were not functioning as pioneer species as suggested by the classical rangeland model (sensu Young and Evans 1973). Rather, they seemed to function as ephemerals responding to specific environmental conditions perhaps driven by precipitation timing. Just because native annuals aren’t likely good criteria for evaluating rangeland health, however, doesn’t mean that they don’t potentially play an important role in sagebrush steppe community structure.

It is possible that the post-fire pattern of native annual species in the study reported by Young and Evans (1978) may have been coincidental with favorable precipitation conditions resulting in an ephemeral flush rather than due to direct competition with *B. tectorum*. However, this is not to say that competition form *B. tectorum* would not eventually eliminate native annuals from the plant community. Because of this result, it may be important to consider that the presence of native annuals may not necessarily be a sign of poor range condition, but rather are components of a healthy plant community, much as we recognize the role of annual forbs in the warm deserts (Smith et al. 1997).

### *Species Richness*

Because both average species richness and average total cover increased significantly from 2003 to 2004, cover and species richness appear to be positively related (Figures 2-1 and 2-2). However, native resprouting shrub cover increased significantly each year of the study without any related increase in species richness of native resprouting shrubs. Conversely, high yearly variations in absolute cover of annual species were reflected in high yearly variations in the species richness of native annuals. Therefore, changes in total vegetative cover seem to be driven by a combination of directional changes in the cover of native perennials and yearly fluctuations in the cover of native and introduced annual species. However, changes in plot-level species richness appear to be driven solely by fluctuations in species richness of native annual forbs, rather than directional changes in the species richness of perennial species. Interestingly, high yearly fluctuations in cover were similar for native and introduced annual forbs, but high yearly fluctuations in species richness were only characteristic of native annual forbs. Thus, total species richness is driven by native annuals and isn't likely a good criteria for measuring vegetation recovery or community health. Species richness of native perennial species is probably a much more important indicator of community health in terms of stability and function.

Even though there have been a number of studies pointing to a link between species richness and numerous aspects of ecosystem function, few have been conducted in sagebrush steppe rangeland. In one of those few studies, Anderson and Inouye (2001) concluded that greater species richness was related to stability in plant cover and to resistance to domination by *B. tectorum*.

### *Precipitation Effects*

Because native shrub cover increased significantly and proportionately during each year of the study, and plot-level species richness of native shrubs remained stable throughout the study period, the timing of precipitation appeared to have very little effect on the recovery trajectory of that functional group. A significant increase in the basal cover of perennial grasses occurred only between 2002 and 2003, while cover remained the same between 2003 and 2004. Recall that less than one millimeter of precipitation occurred in June of 2003, while precipitation for June and July of 2004 were at and above normal, respectively. Species richness remained constant for perennial grasses during the study period. Therefore, precipitation timing also had no perceptible effect on the recovery of native grasses during the study period. Likewise, recovery of native perennial forbs, in terms of cover and species richness, was not affected by the timing of precipitation. Conversely, precipitation timing did have a substantial impact on the relative importance of annual species throughout the study. The relatively dry June of 2003 resulted in a loss of native annuals from density sampling and a significant decrease in species richness from 2002. The relatively wet period in June and July of 2004 translated to significant increases in cover and substantial, if not significant, increases in density of both native and introduced annual species. Precipitation timing only appeared to impact species richness of native species.

### *Invasive Species*

Many authors have suggested that where exotic annual grasses were present prior to fire; competition for moisture may cause problems for perennial seedling establishment and presumably successful recovery following fire. Long-term monitoring plots elsewhere on the INL show widespread presence of *B. tectorum* (Anderson and Inouye 2001) and we can reasonably assume that it was present on the study area prior to the fire. From this we would have expected that *B. tectorum* would be a significant competitor on these plots following a fire and result in a transition to an annual grass dominated state.

West and Yorks (2002) stated that “changes in state are what happen when thresholds of permanent change in the abiotic environment occur, usually following accelerated erosion.” On upper Snake River Plain, soils are primarily eolian in origin and subject to severe wind erosion following fire. Erosion bridges collocated with two of the plots used in this study showed a loss of 95 and 126 tons of soil in the first year following fire (Darwin Jeppesen, BLM, unpublished data). This suggests that our sites should likely undergo a transition (i.e. cross a threshold) to annual grass domination due to accelerated erosion. Since that transition did not occur, it indicates that accelerated erosion following fire is by itself not enough to cause a transition in excellent condition sagebrush rangeland in the upper Snake River Plain across a threshold to a state dominated by annual grasses. Nor should it alone necessarily be considered an indicator of reduced health in this region.

In defining state and transition models, Stringham et al (2003) noted that a state is composed of the soil base and the vegetation structure. They further noted that the integrity of the soil resource is reflected by site hydrology and nutrient cycling. The high erosion rates associated with this site apparently did not appreciably impact the hydrology or nutrient cycling of the site, and therefore the integrity of the soil resource was not impacted. In other words, the accelerated soil erosion evident at this site did not result in a permanent change in the abiotic environment necessary for crossing a threshold. In order to allow for management to recognize site-specific conditions, using soil resource integrity as an indicator of a threshold may be more reasonable than simply accelerated soil erosion. However, it should not be concluded that high erosion rates will not result in a transition to an annual grass dominated state in sagebrush rangelands in other areas.

Peters and Bunting (1994) suggested a difference in sagebrush steppe between the upper and lower Snake River Plain. They suggest a line between Arco and American Falls is the general dividing line. They report differences in plant animal communities, and fire history between the upper and lower plain during early exploration and settlement of the Snake River Plain in the 19th century. They also suggest that in the upper plain, *B. tectorum* is present, but does not dominate sagebrush steppe communities. The results of our study support that assertion by Peters and Bunting (1994).

It is possible that sagebrush rangelands in the upper Snake River Plain respond differently to fire than sagebrush rangelands elsewhere in the Great Basin, the Interior Columbia Basin and the Colorado Plateau. These differences likely exist even among communities dominated by the same species and/or subspecies of sagebrush. It is likely that managing and restoring sagebrush rangelands will require regionally, if not locally, derived data on ecosystem and community responses to disturbances or management activities. This echoes the conclusion by Allen-Diaz and Bartoleme (1998) that large, long-term, site-specific data sets are required to quantitatively develop state and transition models called for in ecological site descriptions.

It is possible that the resistance of this site to dominance by *B. tectorum* following fire, as has been reported elsewhere on good condition sagebrush-grass sites (West and Hassan 1985), is in some way related to the geographic delineation noted by Peters and Bunting (1994), perhaps as a function of seasonally different precipitation patterns.

*Bromus tectorum* distribution on the INL has been described as being somewhat limited by edaphic factors (Anderson and Inouye 2001). It is present in most areas that have coarse textured soils, but rarely found in areas with fine-textured soils, likely because of soil crusting (Rasmuson 1996). Exceptions to this on the INL are areas that have been grazed excessively (Buckwalter 2002). Anderson and Inouye (2001) concluded that "…where native plant populations in sagebrush steppe are thriving, *B. tectorum* does not pose the threat that it does where populations of native perennials have been depleted." This agrees with the conclusion by Keeley (2006) that grazing coupled with fire can exacerbate problems with invasive plants. Keeley (2006) also hypothesized that control of invasive plants is most likely to be successful under conditions that limit the use of further disturbances, including livestock grazing. This is consistent with the recent findings that exotic herbivores promote exotic plant dominance and richness through reducing the abundance of native plants (Parker et al. 2006).

Recent laboratory studies of *B. tectorum* grown under elevated concentrations of $CO_2$ have shown increased production and fuel accumulation rates (Ziska et al 2005) and changes in foliage chemistry consistent with greater heat release during combustion (Blank et al 2006). Ziska et al (2005) noted that increasing atmospheric $CO_2$ concentrations may have already resulted in higher *B. tectorum* fuel loads which may have lead to effects on fire frequency and intensity. Although reluctant to extrapolate their results to field conditions, Blank et al (2006) noted that their laboratory results on $CO_2$ effects on time to sustained ignition of *B. tectorum* suggest that the risk of ignition is greater now than 150 years ago.

## Management Implications

Sanders (2000) suggested that low recruitment rates for native perennial grasses following fire needs to be addressed. Sanders (2000) further suggests that a need for livestock to graze the first year following fire to get soil coverage on seed to promote germination. In the upper Snake River Plain, seeds and seedlings do not appear to be an important factor in the response of native perennials following fire in a healthy sagebrush rangeland. Our results indicate that native perennial grasses respond to fire through vegetative regrowth and not by recruitment. Improving site conditions for seed germination with livestock is likely not an important management consideration on healthy sagebrush steppe rangelands in the upper Snake River Plain. Managing for vigor of the perennial grasses using management strategies including livestock exclusion should be the higher priority on burned areas.

Although invasive annual species, including *B. tectorum*, were not a substantial part of the post-fire community, both native and non-native annual species were present. This suggests that seeds of these species did survive the fire or were otherwise transported onto the burn and did find suitable conditions for germination. Young and Evans (1973) concluded that light raking to cover B. tectorum seeds likely improves its success on less favorable sites. Although we have demonstrated in this study, as did West and Yorks (2002), that *B. tectorum* does not out-compete established native perennial grasses; *B. tectorum* has been shown to competitively exclude seedlings of native perennial grasses (Young and Evans 1973). Likewise, Evans and Young (1978) reported that perennial grass seedlings in a grazed treatment were lost within four years and that “excellent stands” established following seeding and livestock exclusion. They concluded that rehabilitation without livestock removal is futile. Thus, managing for improved seed germination using livestock would likely favor exotic annual species over native perennial grasses.

This study confirms the results found by other studies in the upper Snake River Plain (Ratzlaff and Anderson 1995, Buckwalter 2002). These studies found that if healthy perennial plant populations are present before a fire, they will dominate the post-fire plant community and can resist invasion by non-native species. Ratzlaff and Anderson 1995, Buckwalter 2002) also reported that drill seeding as part of a post-fire restoration plan can adversely affect the recovery of an otherwise healthy plant community following fire by disturbing the existing native perennial plants.

These results suggest that, at least in the upper Snake River Plain, knowledge of the pre-fire ecological condition is critical for planning burned area rehabilitation. Areas with vigorous native perennial plant communities are less at risk to post-fire invasions and are less likely to require active restoration activities to establish a healthy plant community following fire. A healthy pre-fire plant community can increase the ability of a site to recover even under very adverse conditions such as severe drought, much like that experienced during our study period. However, the lack of sagebrush recruitment in the short-term is a source of concern in terms of the long-term recovery of sagebrush steppe following wildfire.

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# SECTION 3: LONG-TERM PATTERNS OF VEGETATION CHANGE FOLLOWING FIRE IN SAGEBRUSH STEPPE

***Roger D. Blew, Amy D. Forman, Jackie R. Hafla, Mike Pellant***

## Introduction

Historically, large fires were both a natural ecosystem process and an infrequent event in the sagebrush steppe. Fire rotation interval in sagebrush steppe dominated by Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) is on the order of 100 to 240 years (Baker 2006). The inability of *A. tridentata* to resprout following fire defines the fire ecology of these ecosystems as stand replacing with regard to sagebrush (Paysen et al. 2000, Keeley 2006).

Much has been written about the short-term post-fire vegetation patterns in sagebrush steppe, but less attention has been paid to the long-term changes in vegetation following fire. West and Yorks (2002) reported that after 20 years since fire, there appeared to be no trend toward a community representative of the pre-burn condition or to a different potential natural vegetation community. The only apparent trends regarding individual species was a negative relationship between perennial species cover and that of *Bromus tectorum*. They also observed little recovery of *A. t.* spp *wyomingensis* even though unburned patches existed within 50 m of the burned plots. West and Yorks (2002) concluded that bunchgrasses were probably more abundant in the past than today because of a lengthened fire return interval.

Humphrey (1984) reported on vegetation change patterns up to 36 years following fire in *Artemisia*-grass sites in southeast Idaho. He reported that alpha diversity changed little up to 36 years following fire and concluded that patterns of vegetation change observed were the result of the particular traits possessed by those species present.

Neele et al (2000) reported on long-term patterns of vegetation change since fire in *A. t.* ssp. *vaseyana*, but only in terms of sagebrush and the forbs thought to be used by sage grouse. They considered any burned areas older than 37 years to be equivalent to unburned areas and considered differences in vegetation between sites to be the result of differences in topography, fire intensity, grazing pressure and post-fire climate conditions. Neele et al (2000) reported no differences in forb cover associated with time since fire and considered this to be the result of the combination of drought and livestock grazing. They found that cover of *A. t.* ssp. *vaseyana* at 14 years was only half that of unburned areas.

Colket (2003) reported on the recovery of *A. t.* ssp. *wyomingensis* on a chronosequence of fires up to 92 years old. In that study, sagebrush density increased to levels similar to nearby unburned plots at a time approaching 92 years since fire.

A better understanding of long-term vegetation dynamics following fire could result in new management strategies that direct vegetation change onto a trajectory favorable to land management goals. There have been a number of studies on the potential to cross a threshold to annual grass dominated communities immediately following fire. However, very little consideration has been given to the potential for thresholds to become evident at longer times since fire, especially in high quality sagebrush steppe at low risk of domination by annual grasses. The presence of these kinds of thresholds has been implied in the structure of some proposed state and transition models for sagebrush steppe. One of these implied thresholds requiring management input is the transition from perennial grass dominated to sagebrush dominated communities. State and transition models for sagebrush steppe (Laycock 1991 and West 2000) indicate that return to sagebrush-grass dominated communities following fire requires light or conservative grazing by livestock. The suggestion that livestock grazing is necessary for this transition implies a threshold that requires management input to cross.

The purpose of this study is to provide a better understanding the role of fire in the overall dynamics of sagebrush steppe communities, how fire and recovery from fire function as mechanisms of community change, and the potential for fire to result in transitions across a threshold to undesirable, permanent states. The primary objective was to examine long-term changes in vegetation following fire in sagebrush steppe. A secondary objective was to consider these changes in terms of the models used to guide management of sagebrush steppe rangelands.

## Methods

### Study Site

The study was located on the Idaho National Laboratory (INL) (43º08’N 113º33’W) in areas that burned between 1910 and 2000. Mean annual precipitation is 220mm and is predominantly snow. However, May and June are the wettest months and during that time precipitation is rain. Soils on this area were described as loess, sand over basalt and terrace (alluvial) soils (Olson et al. 1995). Some of the burns are within BLM cattle and/or sheep grazing allotments.

Vegetation on unburned portions of the INL are primarily Wyoming big sagebrush (*Artemisia tridentata* spp. *wyomingensis*), bluebunch wheatgrass (*Pseudoroegnaria spicata* spp. *spicata*), and green rabbitbrush (*Chrysothamnus viscidiflorus*). Anderson and Inouye (2001) concluded that the composition and cover of plant communities on the INL are similar to that of nearby ungrazed kipukas (Tisdale et al. 1965, Passey et al. 1982), which are considered to be pristine.

### Survey Methods

Thirty-two plots were selected in areas that burned in 1910, pre-1949 (likely 1930s, see Colket 2003), 1980, 1983, 1994, 1995, 1996, 1999, and 2000. Density, frequency and species richness were measured using Modified Whittaker plots (Stohlgren et al. 1995). These plots are 20m X 50m in size and contain nested subplots. The subplots include ten 0.5m X 2.0m, two 2m X 5m, and one 5m X 20m subplots. These plots were surveyed by doing cumulative species counts in each subplot and finally over the entire plot. This nested sampling design allows for estimation of total species richness using species area curves. Density and frequency were measured in the 0.5m X 2.0 m plots.

### Data Analysis

To assess vegetation recovery trajectories on burns ranging from 3 to 94 years in age, data were analyzed to address recovery issues in two broad categories. The first investigates potential trends in the density and frequency of occurrence of vegetation functional groups and individual species of interest through time. The second set of data analyses examines potential associations between species richness, and related functional issues like stability and invasibility, as they pertain to vegetation recovery over several burn ages.

The data were collected across a variable landscape for which a number of factors could not be controlled. For example, the historical record of recent burns is much more complete than the record for burns greater than fifty year old, resulting in a distribution of plots sampled weighted towards younger burns. Likewise, the burns sampled varied greatly in size, occurred across several soil types and vegetation classes, and were sampled over two growing seasons. Thus, a statistically balanced and quantitatively rigorous analysis was not possible. Nevertheless, in an effort to describe general patterns in the long-term recovery trajectory, and the factors influencing those patterns, we summarized and statistically analyzed (when appropriate) data according to functional groups, and diversity measures such as species richness, using broad statistical approaches. Statistical tests included; one-way ANOVAs, two-way ANOVAs, simple correlations, and multiple comparison correlations (Zar 1984). Data were transformed, when necessary, to meet assumptions of equal variance. It is important to note, that because the sampling effort was not designed to be quantitatively rigorous, these analyses are focused towards detecting the potential for general trends, and the trends reported herein should be researched more thoroughly before being widely generalized.

## Results and Discussion

### Density and Frequency Trends

Plant density was plotted against time since fire for several functional groups to assess the potential for directional changes through time (Figure 3-1). For example, does perennial grass density tend to decrease as a community recovers, or does native forb density increase during the recovery process? Because the data did not have adequate distribution across the abscissa and did not meet assumptions of normality or equal variance due to the variability in the sampling design discussed above, regressions were not performed on the data. Instead, the data were qualitatively analyzed for general trends.

For all functional groups analyzed, plant density was highly variable, ranging from nearly zero to tens or hundreds of individuals per square meter in areas that had burned ten years previous to sampling or less. In areas burned more that twenty years previous to sampling, plant densities in most of the functional groups appeared to be less variable and low relative to some of the plots sampled in more recent burns. High variability in plant densities in more recent burns may result from higher variability in recruitment and seedling mortality in those post-fire plant communities. Hence, as these plant communities recover and mature, plant densities in most functional groups appear to become more stable. Alternatively, seed availability may have changed through time, for at least some functional groups, resulting in greater variability in plant densities in more recent burns. For example, cheatgrass is more widely distributed across the INL currently than if was fifty or more years ago (Anderson and Inouye 2001). Thus higher variability in annual grass density of more recently burned plots could be related to increases in distribution and seed availability of those species.

The only functional group for which variability in plant density did not appear to decrease as a function of time since fire was native perennial forbs. Interestingly, native perennial forb density was also less variable across plots sampled in more recently burned communities than were densities of plants in other functional groups in the same plots, as evidenced by the relatively smaller spread of the ordinate. Additionally, native perennial shrubs was the only functional group for which plant density tended to show an increase through time since fire, and this trend only applied to plots sampled in burns greater than twenty years old.

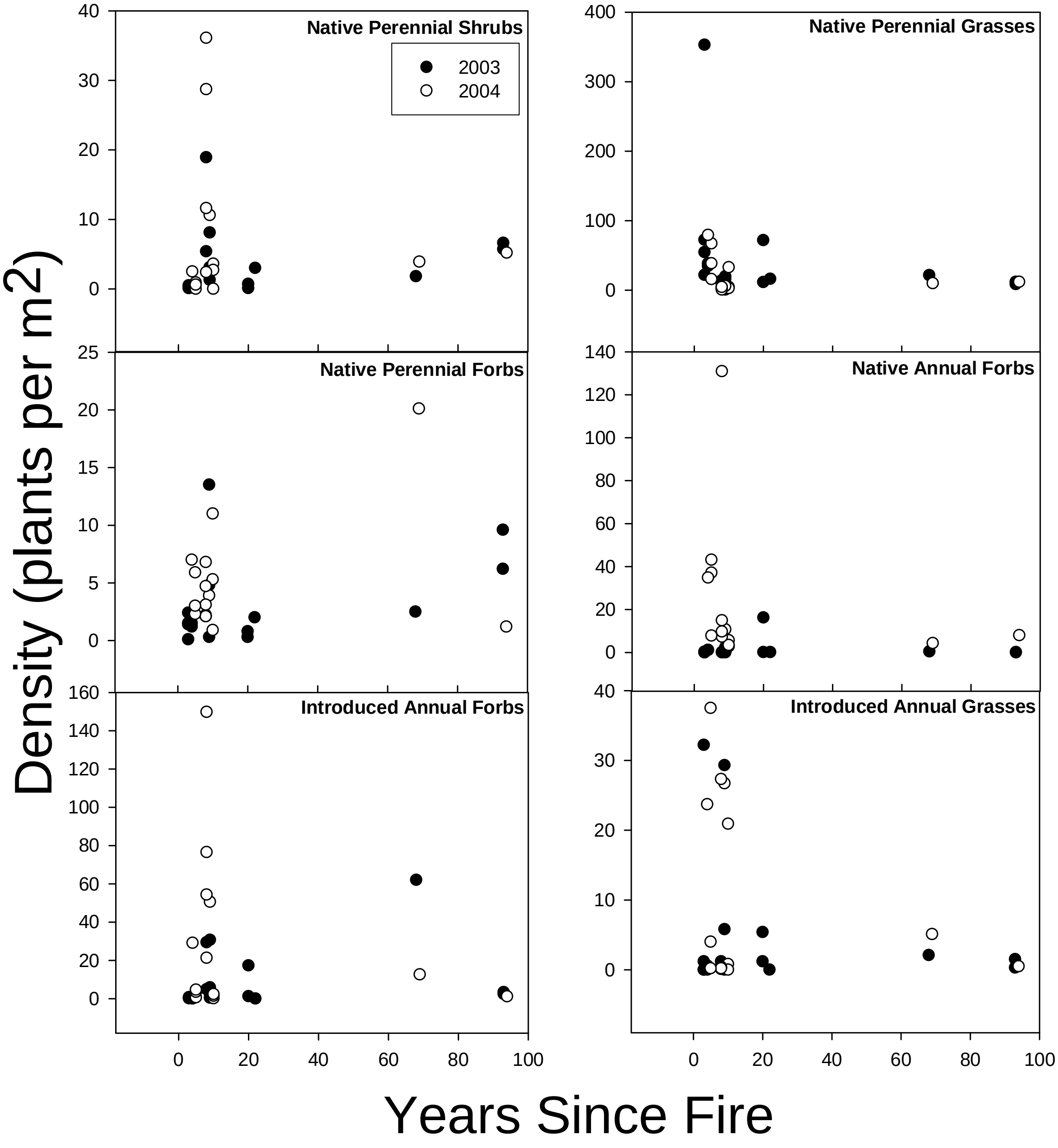


**Figure 3-6. Changes in densities of plant functional groups through time since fire. Open symbols denote plots surveyed in 2003 and closed symbols denote plots surveyed in 2004.**

Since density data were collected across two growing seasons with very different precipitation patterns (see Chapter 2), data were plotted separately for each year in which data were collected. The only functional groups for which summer precipitation may have affected density were native annual forbs and introduced annual forbs. Plots sampled in 2004 tended to have greater annual forb densities than plots sampled in 2003, as seen by some separation between sample years along the ordinate. However, the effect of sample year on density only appeared to affect annuals in post-fire communities that burned within the last ten years. Plots sampled in burns greater than twenty years old, did not exhibit the same pattern of separation of annual densities along the ordinate between sample years. Thus, annual forbs, both native and introduced, may be able to better take advantage of available summer moisture in plant communities burned more recently than in burns several decades old.

Density and frequency of several individual species were plotted against time since fire to determine whether any particular species of interest exhibits patterns of change that may indicate a particular successional role of that species in the long-term trajectory of the plant community (Figure 3-2). For example, a species that decreases in density through time may indicate that that species has an important role in early succession, but becomes increasingly less important as the community matures. Or, an increase in frequency may suggest that the distribution of that species becomes more homogenous as the plant community recovers.

Sagebrush density and frequency exhibited an increasing trend with burn age (Figure 3-2). This result was not unexpected as sagebrush must reestablish from seed, is rarely present in plant communities for several years post-fire, and can take several decades to recover to pre-fire density and frequency. Conversely, because green rabbitbrush resprouts in the growing season immediately subsequent to a fire and has a life history very different from that of big sagebrush, the trajectory of rabbitbrush, in terms of density and frequency, might be expected to be very different than that of sagebrush. In fact, in burns less than ten years old, rabbitbrush density and frequency were much more variable than density and frequency of sagebrush. These results indicate that rabbitbrush may play an important role in some, but not all, post-fire communities shortly after the fire. In burns greater than twenty years old, the density of rabbitbrush was less variable and low relative to younger burns. This result may indicate a decrease in the importance of rabbitbrush as a community recovers; however, the frequency of rabbitbrush in those burns was somewhat less variable and higher relative to younger burns. Because the frequency of rabbitbrush in post-fire communities increased with increasing burn age, the decrease in rabbitbrush density with burn age was more likely related to an increase in the size of maturing shrubs rather than a decrease in relative importance in the community.

Like green rabbitbrush, the life history of bottlebrush squirreltail, a short-lived, resprouting perennial bunchgrass, may make it an important species in early post-fire recovery that decreases in importance as the community matures. However, squirreltail density was quite variable across burn age, and frequency was more variable in younger burns than in older burns. Squirreltail frequency was relatively high in older burns when compared with burns less than ten years in age, again exhibiting an increase in distribution across the landscape rather than a decrease in importance, as might have been expected.

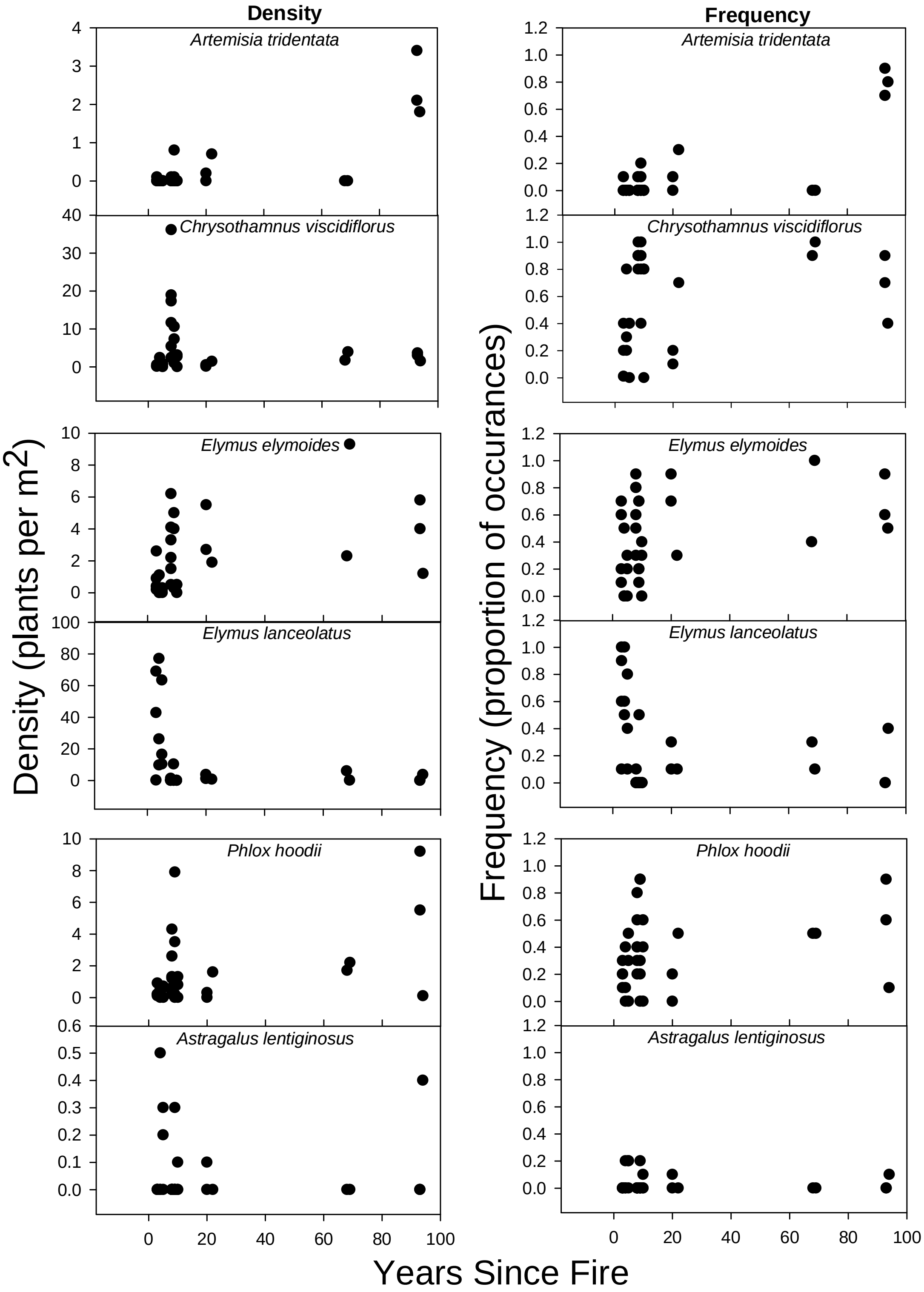


Figure 3-7. Changes in density and frequency of species though time since fire.

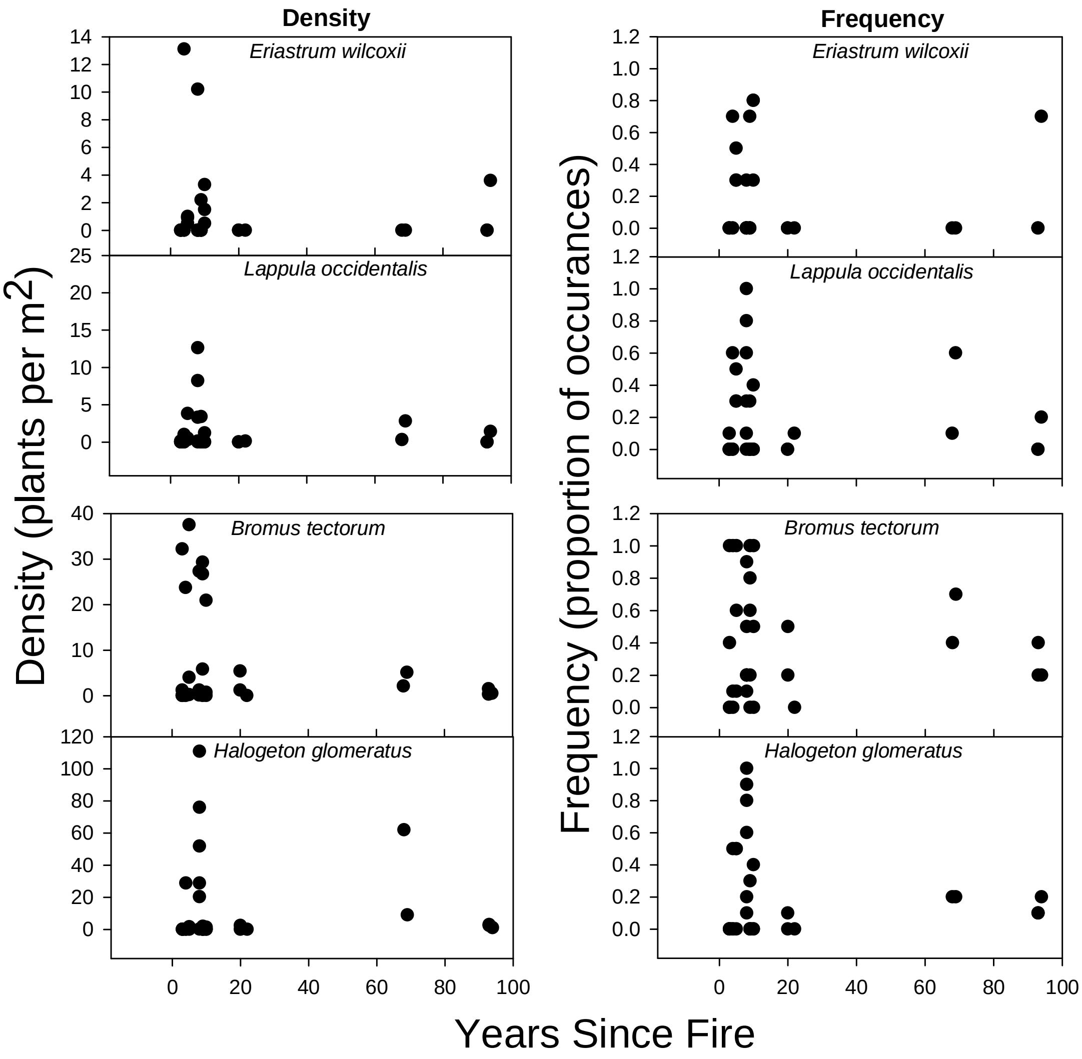


**Figure 3-2. Continued.**

With the exception of big sagebrush, high variability in density and frequency across burn age was the only pattern common among the individual species studied. Thus, none of the individual species studied, with the exception of sagebrush, appeared to have a distinct directional change in the post-fire plant community. Rather most species varied in importance across the landscape, and recovery time since fire did not emerge as an important factor controlling that variation. Aside from high variability, a few species individually exhibited noteworthy patterns in density and frequency. For example, freckled milkvetch, the most common nitrogen-fixing perennial forb sampled during the study, had a much more limited distribution within the plots sampled, as evidenced by relatively low frequency numbers, than did Hood's phlox, the most common non-nitrogen fixing forb. Additionally, cheatgrass was present on nearly every study plot in moderate frequencies, but cheatgrass densities were quite low across the study site.

One way ANOVAs were performed on transformed density data to determine whether significant differences existed between years of fire within each functional group (Figure 3-3). Native perennial shrubs and introduced annual forbs were the only two functional groups for which significant differences existed. Mean native shrub density was significantly greater in the 1995 and 1996 burns than it was in the 1983, 1999, and 2000 burns. Introduced annual forb density was significantly greater in the 1996 burn than in the 1910, 1994, 1999, and 2000 burns. As with previous analyses, no specific trend or pattern relating functional group densities to recovery time since fire emerged.

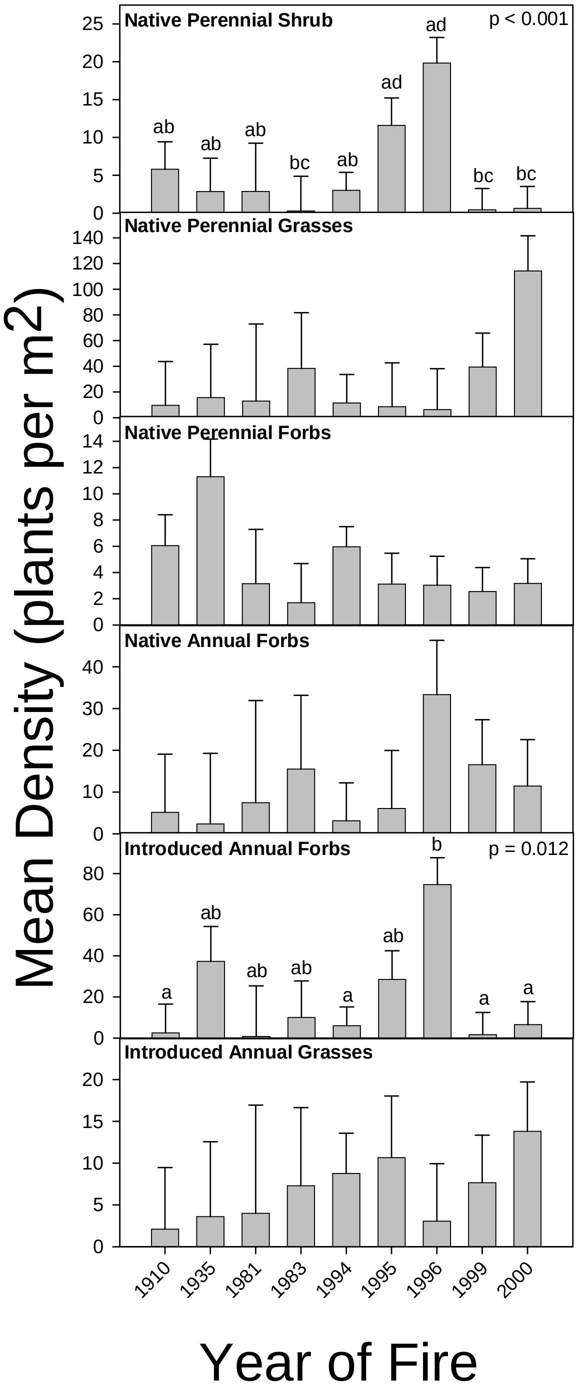


**Figure 3-8. Comparison of mean densities of functional groups between fires of different ages. Letters denote means not different.**

As a whole, the initial density and frequency analyses did not suggest recovery patterns that were directional, linear, or predictable for any functional group or individual species except sagebrush. Because the density and frequency analyses did not indicate that recovery time since fire was an important predictor of post-fire vegetation composition across the burned areas sampled in this study, we examined other factors that may have greater influences on post-fire vegetation composition of individual burns. The additional factors considered reflected some of the variability across the landscape on which the study was conducted, and included; variation within burns of the same age at different locations, variation among functional groups across broad soil types, and the potential interactions among functional groups. The additional data analyses presented here are strictly exploratory and are by no means comprehensive.

Since the density data collected during this study exhibited high variability and the data were collected across a variable landscape, we compared densities of three functional groups in three burns of the same age at spatially distinct locations on the landscape using one-way ANOVAs. The goal of this analysis was to determine whether spatial location on the landscape affects post-fire recovery trajectory. Because native perennial species contribute the greatest amount of plant cover and vegetative structure to post-fire communities on the INL (see Chapter 2), the functional groups analyzed included; native perennial shrubs, native perennial grasses, and native perennial forbs. Results from the ANOVAs do indeed support the hypothesis that location on the landscape may affect the post-fire recovery (Table 3-1). For example, shrub density was significantly greater in the Tin Cup burn than it was in the PBF burn. Additionally, standard errors of the mean were only half of the magnitude of the mean or less. When densities of functional groups were averaged according to year of fire, standard errors of the mean were often equal to the magnitude of the mean for many functional groups (Figure 3-3). Thus, at least some of the variation associated with post-fire vegetation patterns may be attributed to local site conditions.

**Table 3-1. ANOVA results comparing densities of shrubs, grasses and forbs between three fires occurring in 2000. Lower case letters indicate means not different. ANL = Argonne National Laboratory Fire, PBF = Power Burst Facility Fire, TC = Tin Cup Fire.**

| | | ANL | PBF | TC | P value |
|---|---|---|---|---|---|
| Shrubs | Mean | 1.40 | 0.23 | 2.07 | **0.039** |
| | SEM | 1.10 | 0.13 | 0.31 | |
| | | **ab** | **a** | **b** | |
| Grasses | Mean | 75.85 | 143.13 | 50.24 | 0.359 |
| | SEM | 3.45 | 105.31 | 14.69 | |
| Forbs | Mean | 4.20 | 1.33 | 5.36 | 0.132 |
| | SEM | 2.80 | 0.67 | 0.91 | |

Soil type is one spatially distributed variable that may influence post-fire plant communities. Therefore, we assessed the potential influence of soil types on functional group densities in post-fire communities by assigning each plot to one broad soil category, either sand over basalt, loess, or terrace soils, regardless of burn age and compared mean densities among soil types using one-way ANOVAs (Table 3-2). Native perennial shrub densities were significantly greater in loess soils than in sands, and introduced annual forb densities were greater in loess soils than in either sands or terrace soils. Interestingly, the significant differences within these two functional groups in this analysis coincide with significant differences within the same two functional groups in the ANOVAs comparing functional group to year of fire (Figure 3-3). The only burn for which all of the plots were in sandy soils was the 1996 burn; the 1996 burn was also the only burn to have significantly different mean native shrub and introduced annual densities from burns of other years. Hence, soil type likely plays an important role in defining the post-fire plant community.

**Table 3-2. ANOVA results comparing densities of plant functional groups between three soil categories. Lower case letters indicate means not different.**

| | | Sand | Loess | Terrace | P value |
|---|---|---|---|---|---|
| Native Perennial Shrubs | Mean | 2.41 | 10.99 | 3.55 | **0.023** |
| | SEM | 0.73 | 4.01 | 1.68 | |
| | | **a** | **b** | **ab** | |
| Native Perennial Grasses | Mean | 46.49 | 15.91 | 16.45 | 0.409 |
| | SEM | 18.88 | 7.42 | 6.56 | |
| Native Perennial Forbs | Mean | 2.97 | 5.42 | 6.70 | 0.168 |
| | SEM | 0.59 | 1.75 | 3.26 | |
| Native Annual Forbs | Mean | 7.16 | 20.80 | 1.45 | 0.279 |
| | SEM | 3.04 | 12.66 | 1.38 | |
| Introduced Annual Forbs | Mean | 8.19 | 41.31 | 1.62 | **0.012** |
| | SEM | 3.38 | 14.72 | 1.40 | |
| | | **a** | **b** | **a** | |
| Introduced Annual Grasses | Mean | 8.98 | 6.61 | 0.35 | 0.417 |
| | SEM | 3.16 | 3.23 | 0.21 | |

Finally, to address the potential importance of the spatial distribution and relative importance of one functional group on the distribution and relative importance of another functional group during recovery, we performed a multiple comparison correlation. Only three functional groups were significantly ($p < 0.05$), positively correlated. Native annual forb density was weakly correlated with introduced annual forb density ($r^2 = 0.52$) and introduced annual grass density ($r^2 = 0.20$). Notably, the density of individuals in the native perennial shrubs, grasses, and forbs functional groups had no significant relationships with one another on the study plots. These results suggest that the recovery of species in one functional group does not come at the expense of the recovery of species in another native perennial functional group. For example, native perennial grass density does not appear to decrease in response to increasing shrub density. Thus, there was no evidence that spatial distribution and variability across the landscape in densities of various functional groups were controlled by processes like competitive interaction.

In summary, data analyses directed towards detecting potential trends in the density of individuals, categorized into several vegetation functional groups, during the post-fire recovery process failed to identify directional changes across several burn ages. Likewise densities and frequencies of individual species did not suggest any serial replacement or substantial increases or decreases in the distribution or importance of any species, except sagebrush, across burns of various ages. Conversely, densities of species or of functional groups were highly variable in plots sampled among the various burn ages, indicating that factors other than the length of time that had passed since a plant community burned, strongly influence the post-fire structure of that community. Indeed, densities of native shrubs did differ significantly among two burns of the same age, implying that spatial variability of environmental conditions may affect post-fire community structure. This result was confirmed by significant differences in densities among some functional groups according to broad soil categories. Furthermore, for native perennial species, density of species in one functional group does not appear to affect the density of species in another functional group during the long-term recovery process.

### *Relationships Between Species Richness and Recovery*

Since results from density and frequency analyses of vegetation functional groups and individual species indicated that variable environmental conditions across the landscape likely had a greater impact on the plant community composition than time since fire, we hypothesized that recovery patterns may be more easily identified in broader metrics that are less sensitive to landscape scale variation than density and frequency. Therefore, we analyzed species richness among functional groups for patterns of post-fire recovery. Functional group species richness data were analyzed for potential upward or downward trends with time since fire, relationships between density and species richness, and relationships between species richness of various functional groups.

One-way ANOVAs were used to analyze total species richness and species richness in several functional groups for changes among burn ages (Figure 3-4). As with similar ANOVA analyses performed on density data (Figure 3-3), no patterns suggesting directional changes in species richness through time were evident. Mean species richness for native annual and biennial forbs was significantly greater in the 1910, 1999, and 2000 burns than in the 1996 burn, and introduced species mean species richness was significantly greater in the 1995 burn than in the 1981 burn. However, within native perennial functional groups, mean species richness did not significantly differ among burns only a few years post-fire to burns nearly a century into the recovery process. Furthermore, standard errors of the mean were quite low relative to the mean within most burn years and functional groups. Taken together, these results indicate that species richness, especially in native perennial functional groups, is remarkably stable within and among post-fire plant communities regardless of time since fire.

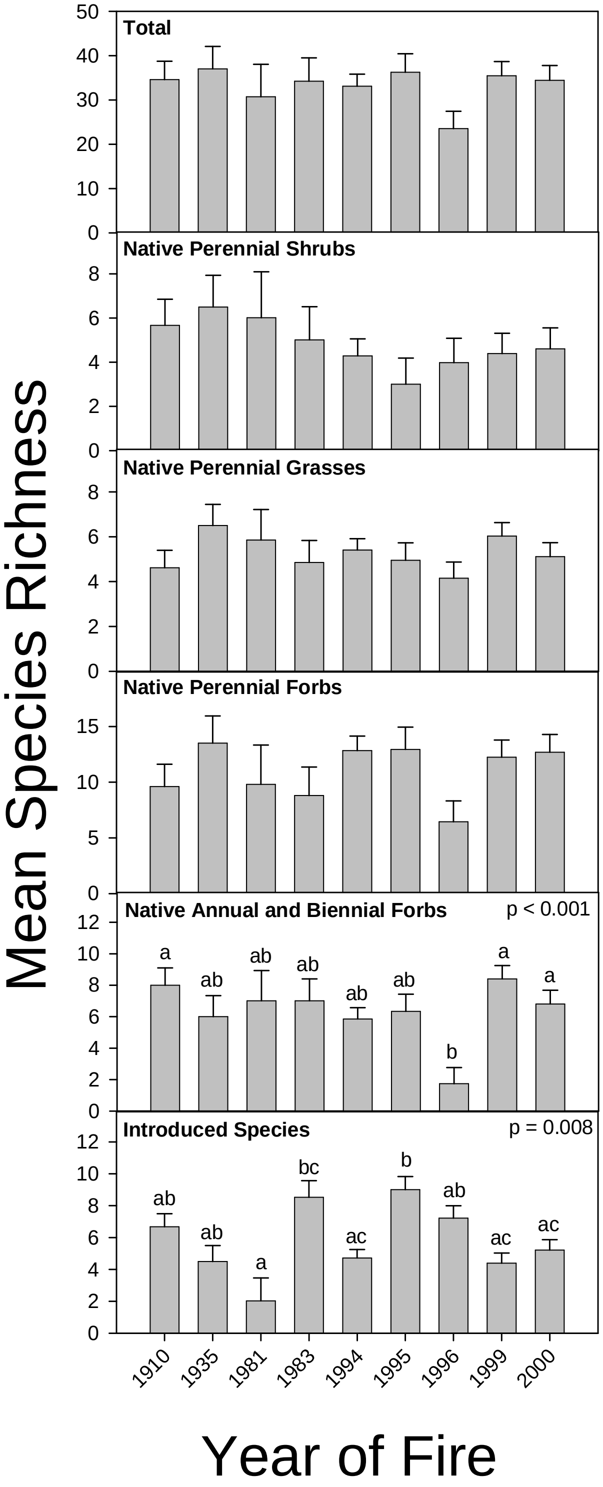


Figure 3-9. Comparison of mean species richness between fires of differnt ages.

In order to assess the potential relationship between species richness and abundance, in terms of density, we compared the species richness of each functional group to the density of individuals within that functional group using correlations (Figure 3-5). Again, only two functional groups, native annual and biennial forbs and introduced species (which consist primarily of annual forbs), had positive significant, but weak, correlations between species richness and density. Thus, an increase in the number of species in a post-fire community does not necessarily relate to an increase in the abundance of those individuals in the native perennial functional groups. It should be noted, however, that plant density does not closely approximate plant cover due to size differences among individuals. Therefore, conclusions about the effects of species richness on stability in cover, biomass, or community structure cannot be drawn from these data. Rather, results from these analyses indicate that patterns of species diversity, in terms of species richness aren't strongly related to community composition, in terms of densities of individuals of native perennial functional groups in post-fire plant communities.

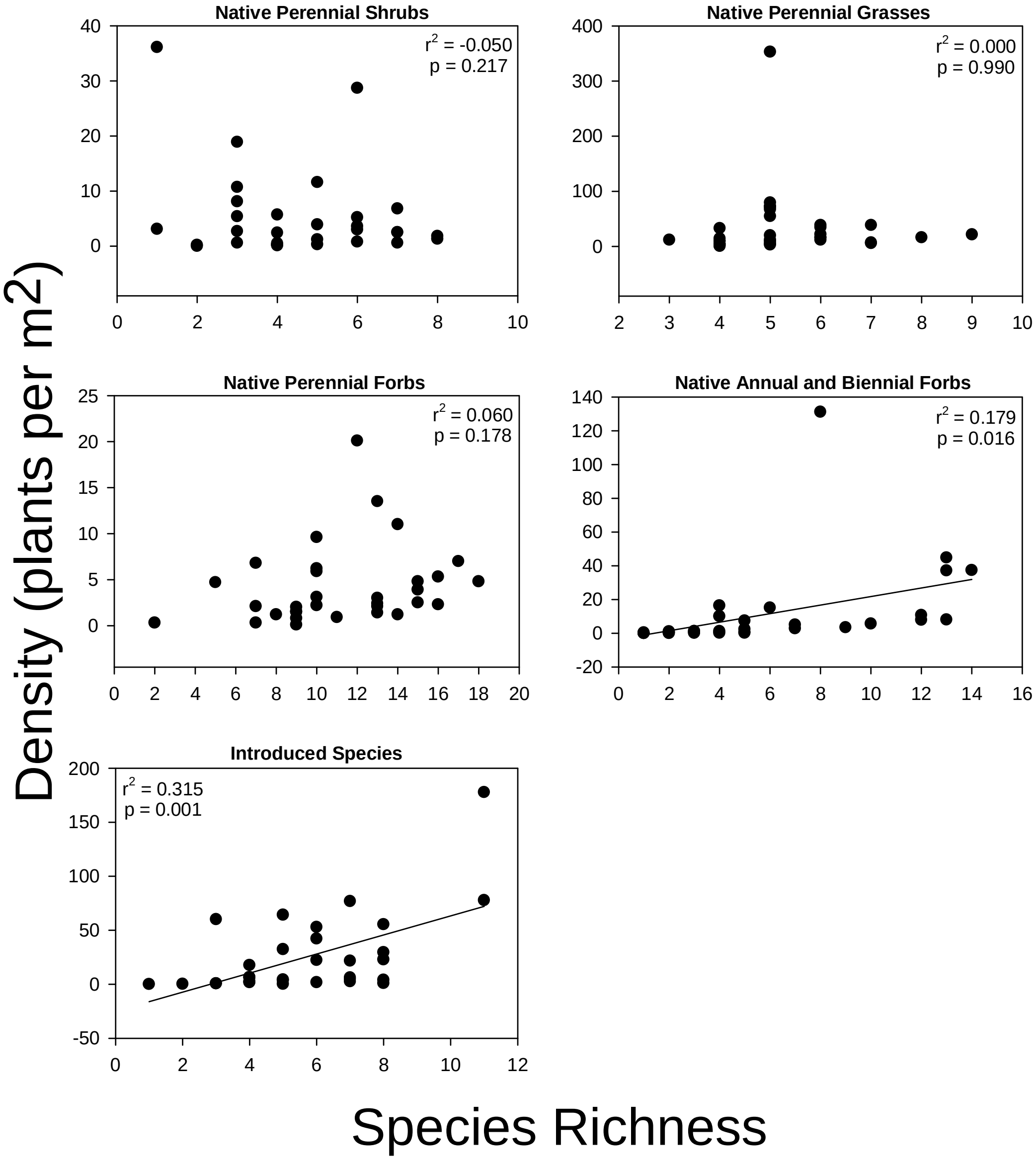


**Figure 3-10. Correlations between species richness and density for each plant functional group.**

Although species richness of native perennial shrubs, forbs, and grasses was not significantly correlated to densities of species within those functional groups, a multiple correlation analysis did suggest a relationship, albeit weak, among species richness within those groups when compared to one another. Species richness of native perennial forbs was significantly ($p = 0.008$), positively ($r^2 = 0.21$) correlated with species richness of native perennial shrubs, and significantly ($p = 0.032$), positively ($r^2 = 0.14$) correlated with native perennial grasses. Thus, conditions on the landscape which promote higher diversity in native perennial forbs, also promote higher diversity in other native perennial functional groups. The same appears to be true for native annual and biennial forbs and introduced species as species richness of those two functional groups were also weakly, significantly ($p = 0.039$) correlated ($r^2 = 0.134$). Interestingly, a weak negative correlation ($r^2 = -0.111$) between species richness of native perennial grasses and introduced species was marginally significant ($p = 0.062$).

In summary, species richness, especially among the native perennials, is remarkably stable regardless of time since fire and between different burns occurring during the same year. Within functional groups, species richness was not related to density. The only exception was with native annual and biennial forbs and introduced species (mostly annual forbs). For these two groups, increasing richness was associated with an increase in density. Conditions conducive to high species richness in native perennial forbs also promoted native shrubs and native perennial grasses.

### *Implications for Models of Vegetation Community Change in Sagebrush Steppe*

These results help to clarify the trajectory of plant community change following fire in sagebrush steppe dominated by *A. t.* ssp. *wyomingensis*. Much as Anderson and Holte (1981) and Anderson and Inouye (2001) concluded for long-term, landscape scale changes in vegetation since release from livestock grazing and drought, no directional changes in species richness, density or frequency were observed for perennial species following fire in sagebrush steppe dominated by *A. t.* ssp. *wyomingensis* with the only exception being the increase in density and frequency of *A. t.* ssp. *wyomingensis* itself with time since fire.

Although sagebrush steppe community response to fire has been referred to in the literature as secondary succession, in high quality sagebrush steppe the process appears to be analogous to a disclimax or subclimax (Clements 1936). This is true in that the result is a relatively stable community caused by a disturbance (fire) and slowly re-invaded by an overstory dominant woody species (*A. tridentata*). Secondary succession is the process of community development on a site where a pre-existing community has been removed (Odum 1971). There are few examples of secondary succession in sagebrush steppe reported in the literature. Hironaka and Tisdale (1963) reported on the return of native perennial species, primarily *Elymus elymoides*, but also *A. tridentata*, on plots that had been cultivated and abandoned.

The classical linear succession model for sagebrush steppe as described by Allen-Diaz and Bartoleme (1998) shows *Artemisia* sp. as present in Stage II and dominating or co-dominating in Stages III and IV (Figure 3-6a). This model shows that in Stage V, the “climax” condition, *Artemisia* sp. is no longer dominant, but is co-dominant or sub-dominant to perennial bunchgrasses. This suggests an inconsistency associated with the ways the range condition model and the classical sagebrush steppe linear succession model function in relation to the Clementsian climatic climax model. Fire moves the community from one dominated by

*Artemisia* sp. to one dominated by perennial grasses. However, the classical linear successional model for sagebrush steppe (Figure 3-6a) does not include an appropriate pathway for fire with the possible exception of a transition from Stage IV to Stage V. That is, according to the classical linear succession model for sagebrush steppe, fire would not act as a disturbance resetting succession further away from climax. Instead, according to the classical linear succession model, fire disturbance, by reducing sagebrush, moves the community *closer* to climax. This would be reflected in the Range Condition Model for sagebrush steppe (sensu Dyksterhuis 1949) as a change from fair or good condition to good or excellent condition (Figure 3-6a).

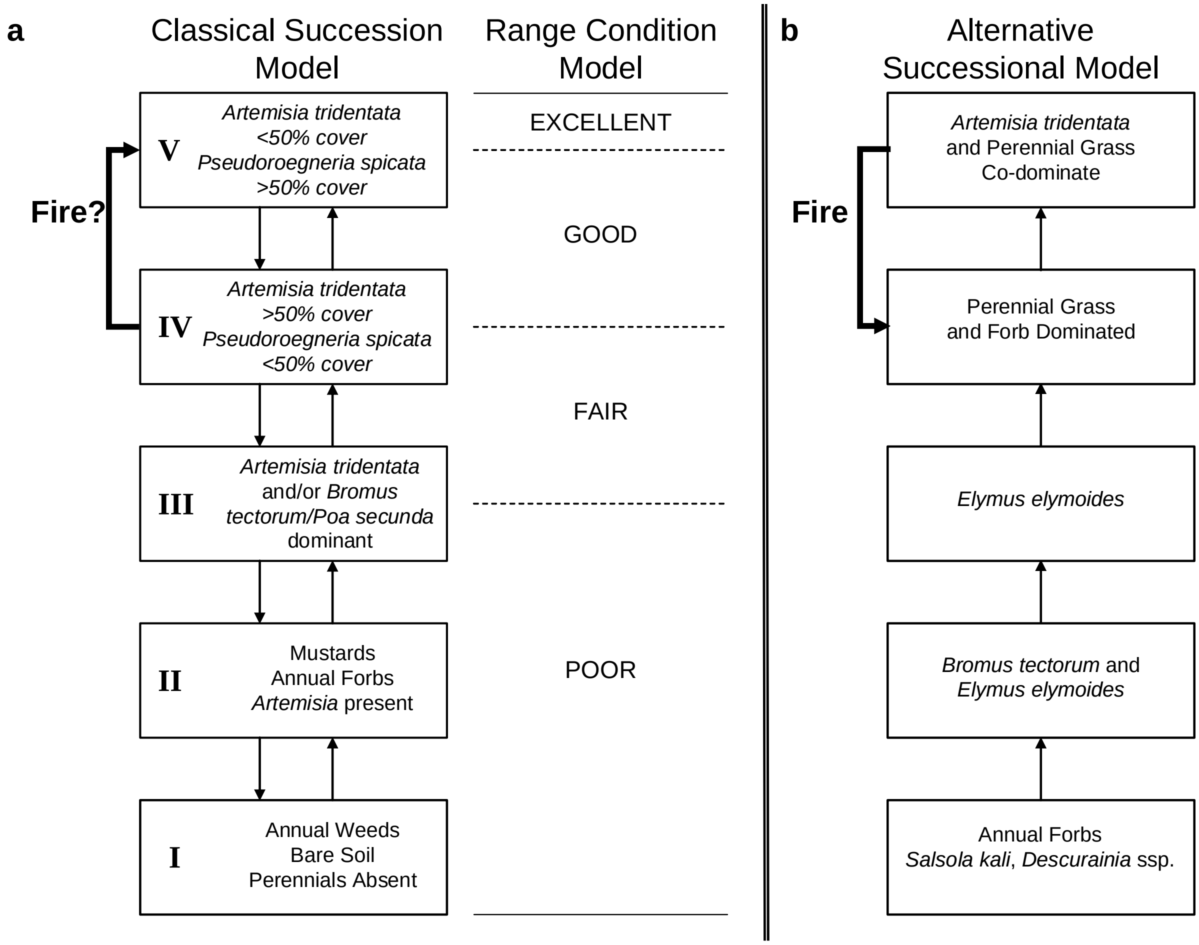


**Figure 3-11. (a) Classical linear successional model for sagebrush steppe and rangeland condition models (re-drawn from Allen-Diaz and Bartoleme 1998) noting the apparent role of fire in development of the climax community. (b) An alternative Clementsian successional model for sagebrush steppe derived from Hironaka and Tisdale (1963), Colket (2003) and results of this study, that shows fire resulting in a disclimax community.**

Laycock (1991) noted that the reason for low range condition is almost always because of "too much sagebrush." *A. tridentata* has historically been considered an "Increaser" (sensu Dyksterhuis 1949) when determining range condition. In this sense, *A. tridentata* increases in importance in the community as range condition decreases. However, and as we have described above, this is at odds with the only directional change found following fire disturbance in this study. That is, in the years and decades following fire *A. tridentata* increased in density which would be interpreted by Clementsian successional models as a trajectory *toward* climax. Although cover was not measured in this study, it can logically be assumed that the increase in density found would be amplified by the growth in size of individuals to an eventual point of dominance or co-dominance.

Once thought to be subclimax to the perennial bunchgrass dominated stage (Clements and Clements 1939), the *Artemisia*/grass stage has been reported to be generally stable (Anderson and Holte 1981, Anderson and Inouye 2001). The return of *A. tridentata* into a perennial grass dominated community, as found in this study, strongly suggests the perennial grass stage, although potentially long-lived, eventually gives way to a sagebrush-perennial grass co-dominated community. These results combined with those reported by Hironaka and Tisdale (1963) on secondary succession and Colket (2003) on invasion of *A. tridentata* following fire, allow for the description of an alternative Clementsian successional model for sagebrush steppe (Figure 3-6b). This model demonstrates how fire might act in sagebrush steppe as it would in other ecosystem types, resulting in a disclimax dominated by perennial grasses.

Many of the reported State and Transition Models carry similar assumptions regarding secondary succession and limit their ability to adequately describe the vegetation community changes associated with fire in high quality sagebrush steppe. More recent state and transition models for sagebrush steppe (West 2000, Stringham et al 2003) do not show a transition from “Late Seral” directly to “Native Perennial Grass Dominated” (Figure 3-7). The results reported in the present study include fires in what would be considered “Late Seral” condition leading to a transition to “Native Perennial Grass Dominated.” This shows a contradiction in the state and transition model as well. According to models described by Laycock, (1991), West (2000) and Stringham et al (2003), fires that lead to the “Native Perennial Grass Dominated” only occur in “Depauperate Late Seral” communities. However, West (2000) notes that fire in the “Depauperate Late Seral” community is unlikely because of the loss of fine connecting fuels. Their models do not show the possibility for “Late Seral” communities to burn.

Similarly, some of these models do not show a transition from “Depauperate Late Seral” to “Late Seral” except through “vegetation modification via grazing systems, prescribed burning, brushbeating and chaining” (West 2000). Reports from long-term studies have shown that this transition does occur without any management input other than rest from livestock grazing (Hironaka and Tisdale 1963, Anderson and Holte 1981, Anderson and Inouye 2001). However, this recovery appears to require a lag phase of two to three decades as species migrate back onto a site. A similar recovery period should be expected following the vegetation modifications listed by West (2000) as the same ecological process of migration must occur. Laycock (1991) described a state and transition model that does recognize this transition (“Depauperate Late Seral” to “Late Seral”) as possible, but suggests it is a transition across a threshold that can be difficult to cross. Stringham et al (2003) does not show this as a state change across a threshold, but rather as a pathway between two community phases (Figure 3-7).

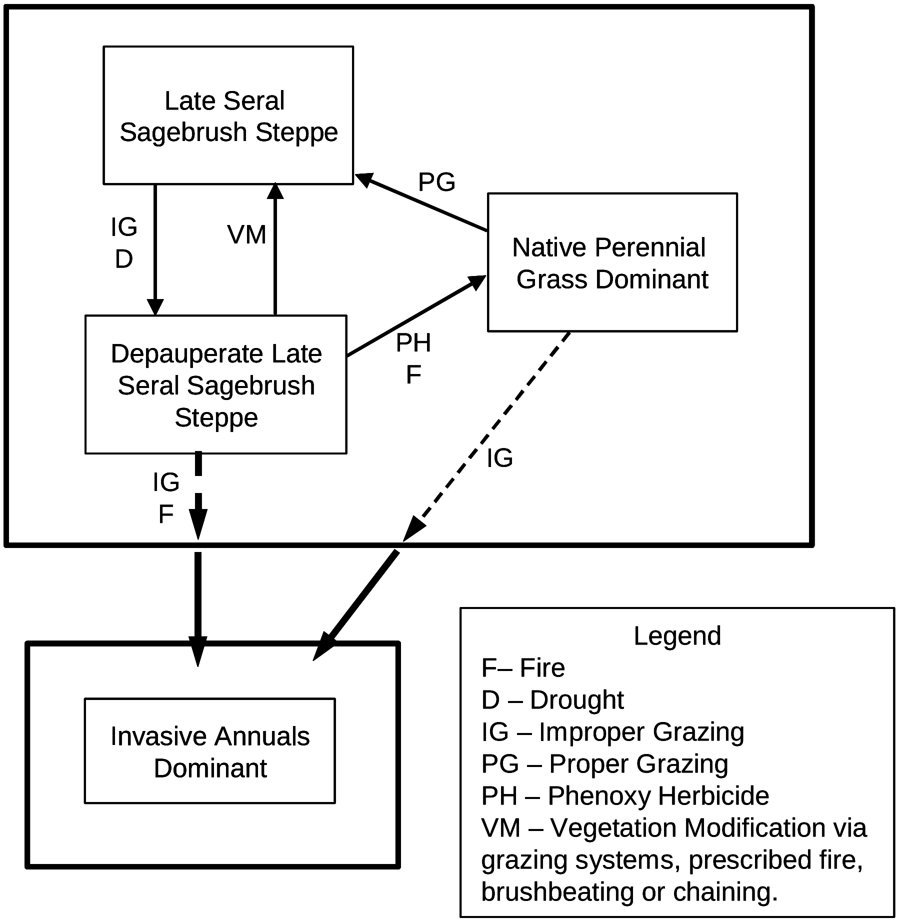
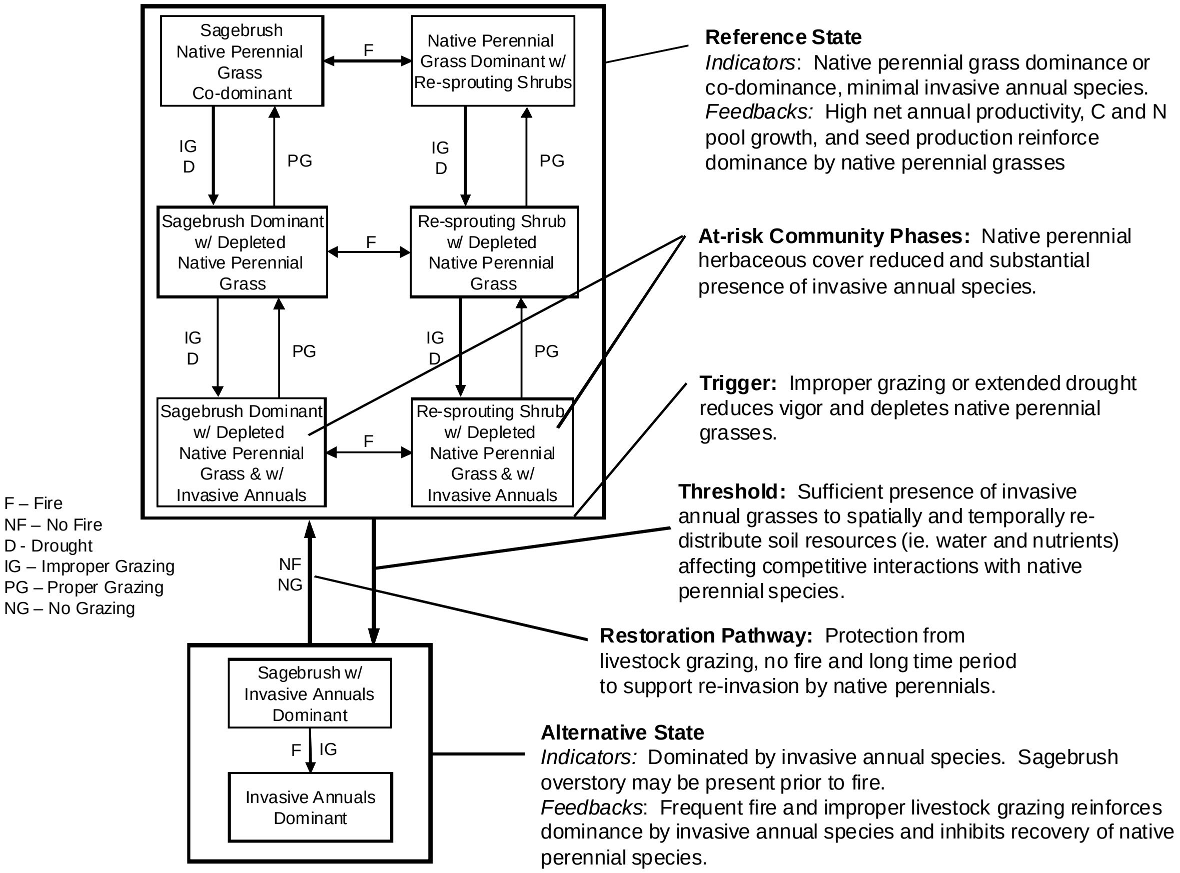


**Figure 3-12. State and transition model for sagebrush steppe redrawn from Stringham et al (2003), but with attributes found in West (2000) and Laycock (1991). These models did not allow for the potential shift from the Late Seral Sagebrush Steppe phase to the Native Perennial Grass Dominant phase.**

We offer a revised approach to state and transition models for sagebrush steppe based on the results of this study and those reported by Anderson and Inouye (2001) and other authors (Figure 3-8). As we report in this study, areas can remain dominated by native perennial grasses with re-sprouting shrubs common for a relatively long period of time. We consider it logical to include this phase as part of the reference state. Given this, we suggest that the indicator for the reference state is high native perennial grass cover with minimal invasive annual species and, therefore, recognize a total of six phases in this reference state.



**Reference State**
*Indicators*: Native perennial grass dominance or co-dominance, minimal invasive annual species.
*Feedbacks:* High net annual productivity, C and N pool growth, and seed production reinforce dominance by native perennial grasses

**At-risk Community Phases:** Native perennial herbaceous cover reduced and substantial presence of invasive annual species.

**Trigger:** Improper grazing or extended drought reduces vigor and depletes native perennial grasses.

**Threshold:** Sufficient presence of invasive annual grasses to spatially and temporally re-distribute soil resources (ie. water and nutrients) affecting competitive interactions with native perennial species.

**Restoration Pathway:** Protection from livestock grazing, no fire and long time period to support re-invasion by native perennials.

**Alternative State**
*Indicators:* Dominated by invasive annual species. Sagebrush overstory may be present prior to fire.
*Feedbacks*: Frequent fire and improper livestock grazing reinforces dominance by invasive annual species and inhibits recovery of native perennial species.

**Figure 3-13. Proposed state and transition model for sagebrush steppe using annotation following Briske et al (2008).**

The phase characterized by co-dominance of sagebrush and perennial grass has been considered in the Clementsian successional model as the “climax” condition. Fire in this phase would result in a second phase dominated primarily by perennial native grasses with substantial presence of re-sprouting shrubs (Figure 3-8). These two phases could be impacted by improper grazing management and/or extended drought such that the native herbaceous species are reduced, resulting in two more phases; one with sagebrush and one without. Anderson and Inouye (2001) reported recovery of native perennial herbaceous species following improper grazing and/or drought to a condition likely similar to the first phase described: sagebrush-perennial grass dominated. A similar recovery should be expected to occur on a site that had previously burned, leading to the second phase described above: perennial grass dominated with re-spouting shrubs present.

Invasion of these two phases by annual species, primarily *Bromus tectorum*, would result in the two at-risk phases (Figure 3-8). These phases are characterized by the depleted native herbaceous component and a substantial presence of invasive annual species. One of these phases has a remnant sagebrush overstory and the other not. These two phases are at risk of conversion to an alternative state dominated by non-native annual species with the trigger being continued improper grazing and/or drought. The threshold for this state change is sufficient presence of annual species and depletion of native perennial grasses to the extent that there is a re-distribution of limiting resources, primarily soil moisture but likely also mineral nutrients. This re-distribution alters the competitive interactions between these functional groups such that the non-native annuals are favored.

The transition to the alternative state does not require fire and can result in a sagebrush overstory still present even though the threshold to the alternative state has been crossed (Figure 3-8). Because the indicator for the reference state is high native perennial grass cover and not simply the presence of sagebrush, the model allows for a phase with an apparent dominant sagebrush overstory, but that is already functionally dominated by non-native annual species. Fire in this phase would result in a phase that is obviously dominated by non-native annual species. Feedbacks in this state include frequent fire that would inhibit the recovery or invasion by native perennial species.

The recovery pathway to the reference state would be through absence of fire, protection from livestock grazing and passage of sufficient time for re-invasion by native perennial species (Hironoka and Tisdale 1963).

Allen-Diaz and Bartolome (1998) noted that although the state and transition model approach has the potential to offer greater resolution in assessing rangeland condition, many of the proposed models were based on the classical successional and rangeland condition models. This may similarly limit the utility of the earlier state and transition models for assessing condition. They recommend overcoming this limitation by identifying stages based on data collected about the communities present in the project area rather than using assumed stages carried over from the classical linear succession or range condition models. In addition they developed their transitions by using the changes in management that brought about changes in state. They also noted that long-term, site-specific data will be required to fully develop state and transition models for predictive purposes. Unfortunately, very few long-term data sets exist that could serve this purpose.

## Conclusions

We found no evidence of directional changes in functional groups or species following fire except for the density of *A. tridentata* ssp. *wyomingensis* which showed a general trend toward increasing density with time since fire. Local environmental conditions, likely soil type, seemed to be more closely related to plant community characteristics than did time since fire. Species richness appeared to be quite stable through time and not related to time since fire. Conditions supporting high species richness of native annual forbs also supported high species richness of native shrubs and native perennial grasses. Both native and introduced annual forbs seemed to be better able to take advantage of moisture available in wet years in more recent burns than in older burns.

There was a trend toward decreasing variability in density for several functional groups. These results suggest that densities within functional groups become more stable with increasing time since fire. This appears to be most prominent during the first 20 years following fire. The perennial forbs were the only group that did not show this tendency toward decreasing variability in density through time.

The results did not show evidence of thresholds to community types that might require management action to correct. In the fires studied, there was no evidence of *B. tectorum* or other annual species dominating a community following fire. Likewise the trend toward increasing *A. tridentata* with time since fire indicates no apparent threshold preventing the eventual return of big sagebrush following fire in high quality sagebrush steppe.

## Management Implications

- Local conditions, likely soils, play an important role in plant community structure following fire and appear to be more important than time since fire.
- High species richness of native annual forbs tends to coincide with high species richness of native shrubs and perennial grasses. This suggests that high species richness of native annuals may not always be a good indicator of range condition.
- The tendency of native and introduced annual forbs to respond more to wet years in younger burns than in older burns similarly suggests that, given adequate perennial species are present, the presence of annual forbs may not always be an appropriate indicator of range condition.
- When preparing state and transition models that will be used to guide management, it is important that all potential states, transitions and thresholds be included in the model and that the model not be constrained by previous paradigms of vegetation community dynamics or management.
- Although fire is a rare but natural process in sagebrush steppe, it is essential to recognize burned areas can be part of the reference state.

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# SECTION 4: EFFECTIVENESS OF AERIAL SEEDING OF WYOMING BIG SAGEBRUSH (*ARTEMISIA TRIDENTATA* SSP. *WYOMINGENSIS*) FOLLOWING FIRE IN SOUTHEAST IDAHO

***Roger D. Blew, Mike Pellant and Amy D. Forman.***

## Introduction

Currently, seeding is the preferred method of assisted big sagebrush (*Artemisia tridentata*) recovery across its range (Shaw et al. 2005). Several reseeding methods have been used in sagebrush steppe rangelands over the past several decades in an attempt to hasten sagebrush reestablishment subsequent to wildland fires. These seeding methods include aerial seeding, hydroseeding, broadcasting, cultipacking, and drilling (Shaw et al. 2005). The effectiveness of various seeding methods is not well understood, and in some cases seeding may cause negative impacts to the post-fire plant community. For example, a study conducted on the effects of drill seeding in southeast Idaho indicate that drilling likely does more harm than good as it can impact the root systems of resprouting native perennial species and further disturb soils, thereby reducing native, perennial plant cover in sagebrush communities (Ratzlaff and Anderson 1995) that had an intact native, perennial understory prior to the fire. Moreover, successful establishment of seeds planted using that particular technique has been quite low (Boltz 1994).

Aerial broadcast seeding of Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) has been used extensively in southern Idaho over the past ten to fifteen years (Lysne 2005). This seeding technique has some distinct advantages over other common reseeding methods, including; fewer labor and equipment requirements, no soil disturbance, and the ability to drop seed on snow. However, a recent study assessing the effectiveness of aerial seeding on over thirty fire-rehabilitation projects in burned sagebrush steppe plant communities of south-central and southwest Idaho indicates relatively low success in terms of sagebrush reestablishment (Lysne and Pellant 2004). Differences in weather patterns and plant community condition, such as those found in southeast Idaho, may improve the outcome of aerial seeding efforts. For example, later timing of spring precipitation in southeast Idaho may augment establishment by reducing mortality related to water-stress through the dry summer months. Additionally, sagebrush steppe plant communities in southeast Idaho often have good recovery of native, resprouting species in the first few growing seasons post-fire when compared with typical sagebrush steppe plant communities of southwest Idaho. Resprouting perennial grasses and shrubs are thought to improve microclimatic conditions and create resource rich islands, which facilitate establishment of important non-resprouting species, such as big sagebrush (Meyer 1994, Monsen 1999). Thus, a few researchers have suggested that sagebrush seed establishment may be enhanced by seeding a plant community that already has some preexisting vegetative cover (Lysne and Pellant 2004).

Because the utility of aerial seeding of sagebrush has not yet been reported in southeast Idaho, our primary study objective was to determine the efficacy of big sagebrush aerial seeding, during the first winter post-fire, on sagebrush reestablishment in a good-condition sagebrush steppe plant community. However, we also assessed sagebrush establishment in a plant community with well-established grasses and resprouting shrubs that was aerially seeded six growing seasons post-fire in order to better understand the effects of existing vegetation on sagebrush establishment.

## Methods

To assess the utility of aerial seeding in southeast Idaho, we surveyed two post-burn plant communities on the INL for sagebrush seedling establishment one to two growing seasons subsequent to seeding. The INL is located within the Upper Snake River Plain of Southeast Idaho. The climate is typical of a cold desert with high seasonal and diurnal temperature fluctuations. Mean annual precipitation is 208 mm. Unburned plant communities adjacent to the burned study sites are dominated by Wyoming big sagebrush and bluebunch wheatgrass (*Pseudoroegneria spicata*). Species compostion of the burned plant communities was similar with the exception of sagebrush. Native, perennial forbs are abundant in the understory of both the burned and adjacent, unburned plant communities.

Portions of the two burns included in the study, the 2000 Tin Cup burn and the 1994 Butte City burn, were aerially seeded by the Bureau of Land Management with Wyoming big sagebrush in February of 2001 using helicopters. Seeding occurred during a period of persistent snow cover at an average rate of 0.133 pounds of pure live seed per acre. Because a full growing season had not elapsed between the date of the burn and the planting effort, existing vegetative cover was sparse on the 2000 burn. However, plant cover was substantial on the 1994 burn as six growing seasons had passed prior to seeding. Data collected during a 2001 study of that burned plant community provides an estimate of approximately 22% average cover for native perennial grasses and resprouting shrubs (Blew et al. 2002). Portions of the burns surveyed were within BLM grazing allotments. Several permitees utilize the allotments for sheep and cattle range. Season of use varies by permit; however the allotments are open to at least one permitee at any given time from April 1st through December 15th. Grazing intensity is quite variable across the allotments, ranging from very light to moderate.

In 2002 and 2003, we surveyed belt transects 1000 m in length, and perpendicular to the predominant wind direction, for seedlings in both planted and unplanted areas. In 2002, all seedling surveys were conducted on the 2000 Tin Cup burn, and belt transects consisted of seven people spaced 5 m apart, resulting in transects approximately 35 m wide. We surveyed six transects in an area that was seeded and was historically grazed. Eight additional transects, or four pairs of transects, were surveyed in an area that had not been previously grazed. Half of each pair was seeded and the other half was not. Thus, we sampled a total of 14 transects, distributed among three treatment combinations in 2002.

In 2003, we sampled the 2000 Tin Cup Burn and the 1994 Butte City Burn. Belt transects consisted of six people spaced 3 m apart, resulting in transects approximately 18 m wide. The width of the belt transects was reduced between the 2002 and 2003 surveys because of reduced sightability coincident with increased vegetative cover between sampling dates on the 2000 burn and considerable vascular plant cover on the 1994 burn. We surveyed a total of 24 transects, each 1000 m in length, distributed among four treatment combinations. Twelve transects were surveyed in each burn; six within the seeded area, and six in an area that had not been seeded with sagebrush during the 2001 planting. All transects sampled in 2003 were located within the Deadman grazing allotment. This allotment is grazed by cattle and sheep in spring and fall.

Due to the low seedling occurrences, 2002 and 2003 data were not analyzed statistically.

## Results

During the 2002 seedling survey of the Tin Cup burn, we found a total of 12 individual seedlings distributed among three groups. All three groups were located along one transect in the planted area. The seedling survey conducted on the Tin Cup burn in 2003 yielded only a handful of seedlings, mostly in lava outcroppings. Because the number of seedling occurrences documented during the surveys was so low, statistical analyses comparing seedling recruitment in planted areas versus recruitment in unplanted areas were not possible (Table 4-1). Seedling occurrences were much higher on the Butte City burn in 2003 than on the Tin Cup burn; however, seedling counts tended to be higher in unplanted areas of the Butte City burn than in planted areas of that burn, precluding the necessity of further analysis.

**Table 4-1. Seedling counts on the 1994 Butte City burn and the 2000 Tin Cup burn from surveys conducted in 2003.**

| Year Burned | Seeded | Not Seeded |
|---|---|---|
| 1994 | 533 | 945 |
| 2000 | 5 | 1 |

## Discussion

Results from the 2002 and 2003 seedling surveys clearly indicate that aerial seeding of big sagebrush failed to greatly improve sagebrush seedling recruitment in the post-fire plant community of the 2000 Tin Cup burn. Likewise, fewer sagebrush seedlings were present in the planted portion than in the unplanted portion of the 1994 Butte City burn, suggesting that the substantial vegetation cover present in the plant community at the time of seeding failed to increase the success of the seeding effort.

One potential cause of low establishment of Wyoming big sagebrush in response to aerial seeding is the loss of seed due to wind. Although some authors have suggested that sagebrush seed dispersal distances are generally less than a few meters, most of those studies considered dispersal distances with respect to a seed source, or mother plant (Meyer 1994). Established vegetation, such as a mature sagebrush individual may function as a windbreak, causing seed to be deposited on the leeward side of the shrub (Chambers 2000). Since very little vegetation was present on the Tin Cup burn during the first winter post-fire, it is possible that much of the seed blew away. In fact, studies in other ecosystems indicate that a majority of windborne seed was found accumulating at the downwind edge of the restoration site (Robinson and Handel 2000). Although, the loss of seed due to wind may explain the lack of aerial seeding success on the Tin Cup burn, a substantial amount of plant cover was present on the Butte City burn at the time of seeding, making a complete loss of seed on that burn unlikely. Thus, other factors probably contributed to lack of recruitment in response to seeding.

An additional factor that likely contributed to the failure of the aerial seeding effort may be related to the plant materials chosen for the project. Some researchers have suggested that using locally adapted seed is critically important to the success of sagebrush plantings (Meyer and Monsen 1991, Shaw et al. 2005). In a study to assess germination differences within and among big sagebrush subspecies, Meyer and Monsen (1992) found substantial variation in germination success of seeds, collected from several sites, under controlled greenhouse conditions. This study indicates big sagebrush is adapted to its site of origin at the population level, creating ecotypes. If sagebrush seed is not adapted to the area being planted, it may germinate too early or too late, causing failure of the entire planting. In a study addressing the efficacy of aerial seeding in southern Idaho Lysne and Pellant (2004) reported that some of the seed material used to aerially seed 35 burns wasn't the correct subspecies, much less the correct ecotype.

Colket (2003) reported that big sagebrush recruitment in burned areas at the INL closely tied to periods of high precipitation. Big sagebrush establishment on burns on the INL occurs episodically and at about ten-year intervals (Colket 2003). This suggests that successful establishment by aerial seeding on the INL would likely have only a 10 percent probability of success in any given year.

## Management Implications

Aerial seeding was not an effective planting strategy for reestablishing Wyoming big sagebrush on two recent burns in southeast Idaho. Furthermore, allowing some native plant recovery to occur before reseeding big sagebrush on one of those burns in did not improve the efficacy of aerial seeding. Several factors may have contributed to the failed seeding effort on the two burns studied, including; loss of seed, use of inappropriate plant material, and weather conditions that were not conducive to germination and establishment.

Some factors that may affect the success of a restoration effort, such as choice of plant materials, can and should be carefully controlled by land management professionals. Conversely, weather-related factors, such as wind, temperature, or precipitation are entirely stochastic. Because stochastic events cannot be predicted or controlled, focusing restoration money and effort on techniques that rely less on favorable weather conditions may be a more effective restoration management strategy. For instance, planting sagebrush seedlings rather than seeds circumvents the critical germination and establishment phase in which sensitive seedlings may be lost to frost or seasonal drought. Although aerial seeding, and seeding in general, are attractive restoration techniques because of the relatively large amount of area that can be treated cost-effectively, the risk of losing the entire seeding to a stochastic event is high. On the contrary, only comparatively small areas can be treated with more intensive techniques, like seedlings, at the same cost as seeding. However, a stochastic event will be much less likely to affect established seedlings, which ultimately results in a more successful attempt at assisting big sagebrush recovery.

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# SECTION 5: NATURAL RECRUITMENT PATTERNS OF WYOMING BIG SAGEBRUSH (*ARTEMISIA TRIDENTATA* SSP. *WYOMINGENSIS*) FOLLOWING FIRE IN SOUTHEAST IDAHO

***Amy D. Forman, Roger D. Forman, Jeremy P. Shive, Jackie R. Hafla***

## Introduction

Natural recruitment of Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis*) can be a slow process on the time scale at which we manage rangelands. In fact, 50 to 120 years may be required for sagebrush cover on a burned site to reach levels comparable to nearby unburned areas (Baker 2006). Colket (2002) found that it took nearly 90 years for sagebrush in one burned stand to reach densities similar to areas sampled outside of the burn scar, and other stands sampled in that study may require an even longer time period to fully recover. Due to the immediate habitat needs of sagebrush obligate species, land managers would often like to reduce the amount of time necessary for natural sagebrush reestablishment (Connelly et al. 2000).

Limitations of sagebrush seed dispersal are thought to be one of the primary factors influencing the amount of time necessary for full recovery of sagebrush cover, and related habitat structure, of a burned site (Mueggler 1956, Johnson and Payne 1968, Young and Evans 1989). Seeding or planting big sagebrush is one restoration approach often used with the intention of overcoming natural seed dispersal limitations and accelerating the rate at which sagebrush becomes reestablished in post-fire plant communities. Some author have even suggested specific planting techniques like creating sagebrush islands to facilitate sagebrush reestablishment (Longland and Bateman 2002, Lysne 2005). However, the success of restoration techniques, such as seeding or planting islands, likely relates to how well they overcome all inherent constraints on natural sagebrush reestablishment, not just seed dispersal limitations.

One commonly accepted model used to describe patterns of natural sagebrush recruitment, the exponential decay model, is primarily based on the shape of the seed dispersal curve alone. Researchers have demonstrated that sagebrush seed does exhibit a negative exponential decrease in abundance with increasing distance downwind from a seed source (Young and Evans 1989). Several studies have also confirmed that the vast majority of seed lands within a few meters of the mother plant, and less than a fraction of a percent travels more than 30 m from the mother plant (as summarized in Meyer 1994), and in some ecosystems, recruitment patterns may closely reflect the exponential decay pattern of seed dispersal (Hastings et al. 2005). However, spatial patterns of “effective seed dispersal, that is, dispersal plus establishment,” don’t always correspond with patterns of actual seed dispersal (Cain et al. 2000).

Differences between spatial patterns of seed dispersal and spatial patterns of actual recruitment are likely related to other factors that constrain germination and establishment. Several specific aspects of the recruitment process such as annual variation in seed production (Harniss and McDonough 1976, Young et al. 1989), persistence of seed in the seed bank (Young and Evans 1989, Meyer 1994), seed dispersal patterns (Johnson and Payne 1968, Young and Evans 1989), microclimatic conditions necessary for germination (Johnson and Payne 1968, Meyer and Monsen 1992, Meyer 1994), and weather conditions required for establishment (Young and Evans 1989, Maier et al. 2001) have been investigated. The effects of the interactions between seed dispersal patterns and germination and establishment factors on the spatial patterns of actual seedling recruitment aren't well characterized. However, restoration approaches with the objective of accelerating natural sagebrush recruitment may become more powerful when considered within the context of how the collective recruitment processes translate into patterns of reestablishment at the scale of a burn scar.

For example, weather patterns often define ecosystem processes in semi-arid environments (Call and Roundy 1991), and often influence microclimatic conditions. Therefore, spatial patterns of species occurrence in sagebrush steppe ecosystems may be affected by temporal events, such as variation in the quantity and timing of precipitation. Precipitation tends be much more variable in arid and semiarid environments than in more mesic ecosystems, which has lead some authors to speculate that plant community structure in semiarid ecosystems is influenced by sporadic stochastic events to a much greater extent than it is by average conditions (Call and Roundy 1991). Accordingly, spatial patterns of sagebrush reestablishment in a post-fire plant community may be influenced by stochastic events as well as seed dispersal patterns.

The objective of this study was to characterize spatial patterns of sagebrush reestablishment in five plant communities that burned four to ten years previous and had not been artificially reseeded. We also wanted to know whether that pattern was strongly influenced by the hypothesized seed dispersal curve or indicative of multiple, interacting processes. We addressed this objective by:

1. Testing the exponential decay model of seed dispersal against actual sagebrush seedling distributions,
2. Analyzing data for spatial seedling occurrence patterns other than those predicted by the exponential decay model,
3. Comparing seedling reestablishment patterns with post-fire precipitation patterns.

## Methods

### Study Site

Our study site consisted of five burns, ranging from four to ten years in age, including an unplanted portion of a 1994 burn and four additional unplanted burns. The additional study sites included burns from 1995, 1996, 1999, and 2000. The resulting burn scars ranged from 2,765 ha to 16,090 ha in size. All five burns are located on the Idaho National Laboratory (INL) within the Upper Snake River Plain, Idaho. The INL is located on 2315 $km^2$ of land that was withdrawn from the public domain by the Department of Energy from 1950 through 1957. It is positioned at the northern extent of the Great Basin and is characterized by cold desert sagebrush steppe vegetation. Plant communities on the INL have been classified into between eight and twenty vegetation types (McBride et al. 1978, Anderson et al. 1996). Annual precipitation averages 208 mm, with May and June typically being the wettest months (see Chapter 2). Snow cover may persist for two weeks to several months in the winter. Mean annual temperature for the INL is 5.6 °C; however, high diurnal and seasonal temperature fluctuations are normal (Anderson and Inouye 2001). Windy conditions are typical and are closely restricted to two primary directions (Figure 5-1). Wind direction is predominately from the southwest, but changes to the northeast for a few early morning hours daily. Mean elevation of the INL is 1500 m. Surficial geology is strongly influenced by volcanic activity and soils include windblown sand or loess over basalt and a few small alluvial deposits. Because soil movement patterns are influenced by abundant basalt outcrops and frequent windy conditions, transitions between soils types and textures may be quite abrupt.

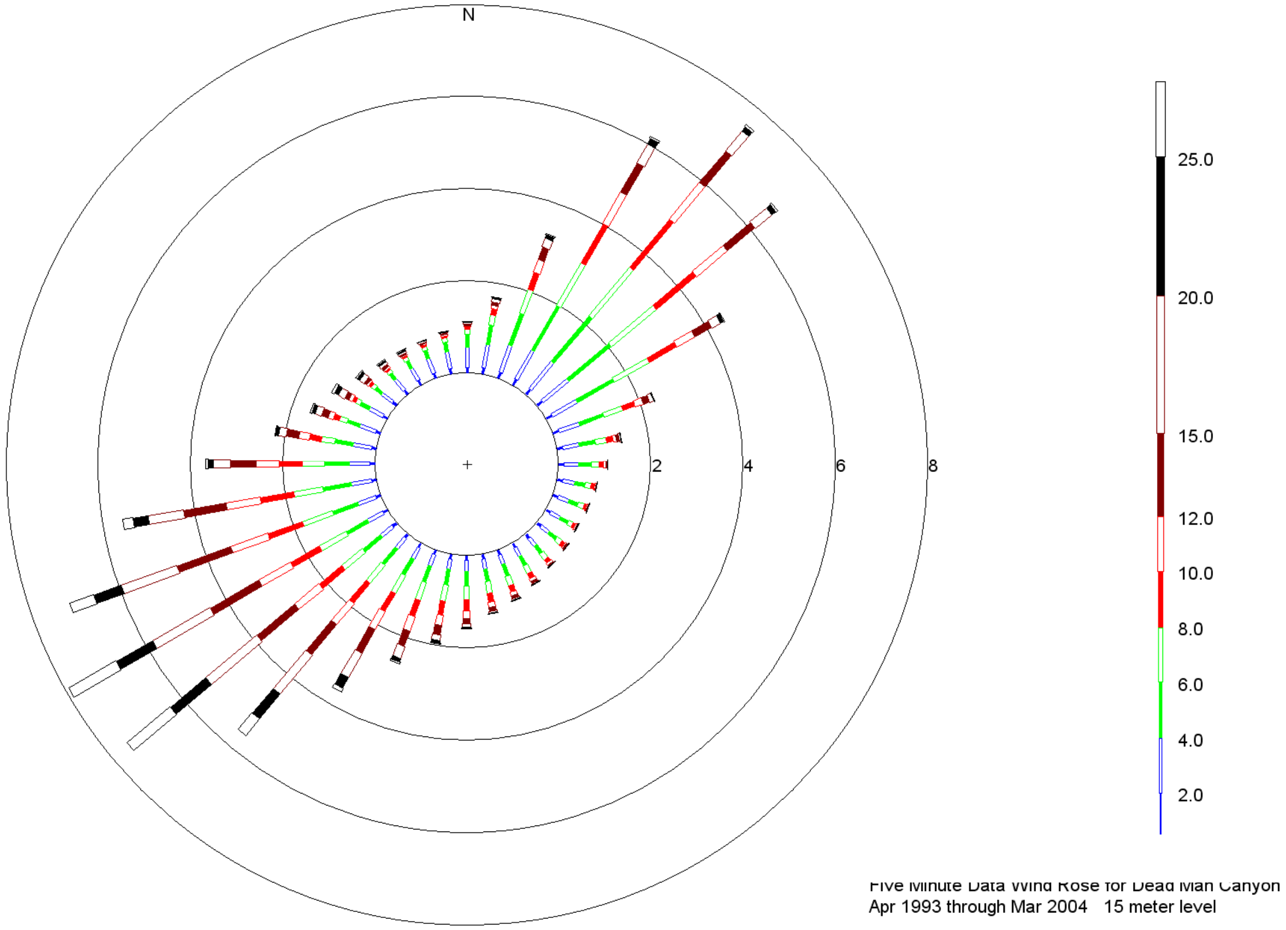


**Figure 5-1. Five minute data wind rose for the Dead Man micrometeorological station located on the INL (unpublished NOAA data). Data are from 1993 through 2004 collected 15 m above the ground surface.**

Mature plant communities adjacent to the burned study sites are dominated by Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*), with a diverse, native, perennial grass and forb understory. Basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*) occurs occasionally on the leeward side of ridges in these plant communities, but doesn't contribute substantially to vegetative cover or vertical structure. The most abundant understory grass species at the study sites include; bluebunch wheatgrass (*Pseudoroegneria spicata*), Indian ricegrass (*Achnatherum hymenoides*), bottlebrush squirreltail (*Elymus elymoides*), thick-spike wheatgrass (*Elymus lanceolatus*), and needle-and-thread grass (*Hesperostipa comata*). One or more of these species dominate the understory of a given study site depending on the elevation and soil texture of that particular site. Introduced species are common in the understory, but densities of those species are quite low (see Chapter 3). Post-fire communities are generally dominated by resprouting shrubs, primarily green rabbitbrush (*Chrysothamnus viscidiflorus*). Resprouting native perennial grasses and native forbs are also abundant in recently burned plant communities. As with plant communities not recently burned, introduced species are often present, but not abundant.

### Survey Methods

We established transects 1000 m in length, approximately 150 m apart, and parallel to the predominant wind direction. Circular plots, each with a 15 m radius were positioned such that the center of the first plot was located 20 m downwind from the unburned edge; subsequent plots were placed along the transect on 50 m centers from the center of the first plot. We surveyed six transects in the 1994, 1996, 1999, and 2000 burns, and we surveyed five transects in the 1995 burn. Transect and plot locations were mapped using GeoXM™ GPS receivers (Trimble Navigation Limited).

In May of 2004, we completed an extensive, but not exhaustive, survey in each plot for sagebrush seedlings that may have become established at any point subsequent to the fire. Plots were located such that unburned islands were not sampled. We occasionally removed individuals at the soil surface and counted rings to estimate the possible size range of individuals established post-fire and insure that we were not sampling in unburned islands.

We then mapped unburned sagebrush islands within the study sites on the 1994,1995,1996,1999 and 2000 burns in June of 2005 using GeoXM™ GPS receivers . This exercise was straightforward on all but the 1994 burn. In all burns younger than the 1994 burn, seedlings established post-fire (determined by occasional ring counts) were less than 15 cm in height, while unburned islands contained several individuals, generally more than 30 cm in height, grouped together on the landscape. On the 1994 burn, however, many individuals that became established post-fire began to approach the size of individuals established pre-fire, making unburned islands more difficult to delineate. Some of the plots we surveyed in 2004 overlapped areas later mapped as sagebrush islands. Because we used ring counts to insure we did not locate plots on unburned islands in the 2004 seedling survey, we likely overestimated the size and extent of unburned islands on the 1994 burn during the 2005 mapping effort.

### Statistical Analyses

We analyzed patterns of seedling abundance within study sites using seedling count data from the plots sampled in 2004. We did not want to mask seedling distribution patterns within each burn by combing data from several burns, so data from each study site (burn) were analyzed separately to compare patterns among burns. We used regressions (Zar 1999), one-way ANOVA (Guenther 1964), heterogeneity chi-squared analyses (Zar 1999), Poisson goodness-of-fit chi-squared analyses (Haight 1967), Moran's *I* index of spatial autocorrelation (Moran 1950), and the Getis-Ord General G clustering index (Getis and Ord 1992) to characterize post-fire seedling recruitment patterns. We performed the regressions and one-way ANOVAs using SigmaStat (Systat Software Inc.), and we used ArcGIS 9.2 (Environmental Systems Research Institute, Inc.) spatial statistics tools to calculate the Moran's *I* index of spatial autocorrelation and the Getis-Ord General G clustering index.

In order to determine whether a directional relationship between seedling abundance and distance from the upwind, unburned edge existed, we initially used regression models. Because we hypothesized an exponential decay model of decreasing seedling abundance from the unburned edge, but wanted to facilitate interpretation using linear regression, we log-transformed the seedling count data prior to regression analyses. We considered the slope of a regression equation to be significantly different from zero at the p = 0.05 level.

Next, we analyzed seedling data from each study site using one-way ANOVA to compare mean seedling counts among the 20 plot positions along transects from the upwind, unburned edge towards the center of the burn. These analyses allowed us to determine whether significant differences in mean seedling counts occurred among plots at various distances from the burn edge without regard to directional trends. We again assigned significance at p = 0.05.

We used heterogeneity and Poisson goodness-of-fit chi-squared analyses to characterize the spatial distribution of seedlings at the plot level in each burn (i.e. do plots tend to have similar seedling counts across the landscape). Heterogeneity chi-squared analyses can be used to determine whether seedlings are uniformly distributed among the plots in each transect, and if seedling distributions among plots in one transect are similar to seedling distributions among plots in other transects. Poisson goodness-of-fit chi-squared analyses indicate whether seedlings occur in random, clumped, or uniform distributions at the plot level. For example, a clumped distribution is characterized by more plots with higher than average numbers of seedlings and lower than average numbers of seedlings than would be expected by chance alone.

We then analyzed relationships of seedling distributions among plots within each study site to determine whether there was clustering of sagebrush seedling counts (i.e. are plots with certain seedling counts spatially related to other plots with similar counts). Moran's *I* index is used to quantify departure from a spatially random distribution, but does not necessarily identify which data values (i.e., high or low seedling counts) are driving the spatial autocorrelation. The Getis-Ord General G index will also identify clustering of similar data values, spatial autocorrelation, but in addition reports whether high or low clustered data values are driving that relationship. Both indices consider the geographic position and the seedling count variable from the database tables. The clustering indices provide a corresponding Z-score result to interpret the statistical significance of spatial pattern, and whether we can reject the null hypothesis, that data values are randomly distributed in space.

To quantify any spatial relationship between proximity to unburned, upwind sagebrush islands and seedling establishment within each plot, we analyzed the unburned sagebrush island data. We used ArcGIS to calculate the distance from the closest sagebrush island to each individual plot along all transects. We applied a circular buffer to each seedling plot with an increasing radius of 1, 5, 10, 20, 30, 40, 50, 60, 70, 80, 90, 100, 200, 300, 400, and 500 m. All buffered plots were intersected with the sagebrush island polygons and the distance to each polygon was added to the plot database tables. We filtered all of the plot database records for redundant and increasingly larger distances, and only kept the single closest distance record for each plot.

Since we hypothesized sagebrush seed dispersal would be driven by wind, we only wanted to include sagebrush island distances along a SW-NE bearing (i.e., the primary wind directions for our study sites). A linear transect polygon was created at a 15 m radius from each plot center point. This distance was selected to match the plot area sampled during the field data collection. This polygon was intersected with the results described above to filter all distances occurring at bearings off-axis to the primary wind direction. For example, the closest sagebrush island may be located due east of a particular plot, but if wind is responsible for seed dispersal, and the direction of wind does not flow from east to west, then seedling establishment should not have been caused by the island to the east. We used regression analyses, similar to those described above, to assess potential relationships between seedling counts and distance to the closest upwind, remnant island.

Finally, in an attempt to understand the potential effects of temporal variability in precipitation on spatial variability in sagebrush recruitment, we compared cumulative annual precipitation and average monthly precipitation to patterns in seedling occurrence. Precipitation data from a meteorological tower at a location central to the study sites was used for this analysis.

## Results

Frequency distributions of mean seedling counts among plots sampled in five burns during the 2004 seedling survey failed to indicate a consistent, recognizable pattern of seedling distribution from the upwind unburned edge (Figure 5-2). Mean seedling counts were generally higher in plots sampled on the 1994 burn, approximately 10.5 seedlings per plot (149 seedlings per hectare), than in plots sampled in burns from subsequent years, less than three seedlings per plot (42.5 seedlings per hectare). However, the predicted exponential decay model was clearly supported by actual seedling distributions only in the 2000 burn. In the other four burns, mean seedling counts were at least as high in plots sampled several hundred meters from the unburned edge as they were in plots sampled within one hundred meters of the unburned edge. Notably, plots sampled more than 500 m from the upwind unburned edge contained at least a few seedlings in all of the burns sampled.

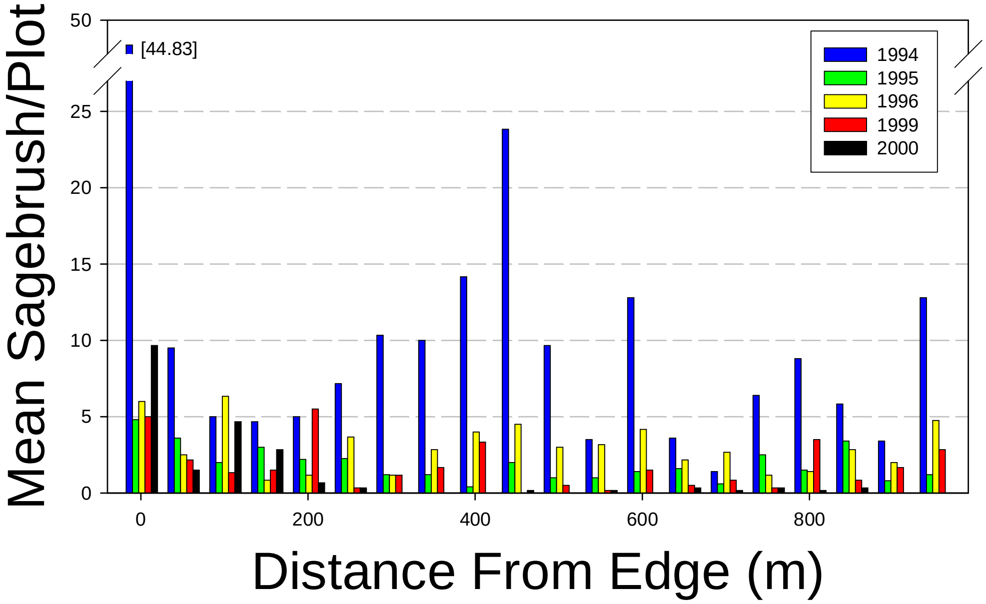


**Figure 5-2. Mean sagebrush seedling abundance in 15 m radius plots located five to one thousand meters downwind from the unburned edge of five burns up to ten years in age.**

Regression analyses confirmed a statistically significant negative relationship between distance from the unburned edge and seedling count in the 2000 burn (Figure 5-3e). A similar relationship between distance and seedling abundance was found to be significant on the 1995 burn (Figure 5-3b). Using regression analyses, we did not find a statistically significant relationship between seedling count and distance from the unburned edge in the 1994, 1996, and 1999 burns. On the two burns for which the relationship was significant, the $r^2$ values were very low, suggesting that the seedling count is only weakly dependant on distance from the unburned edge (Figure 5-3).

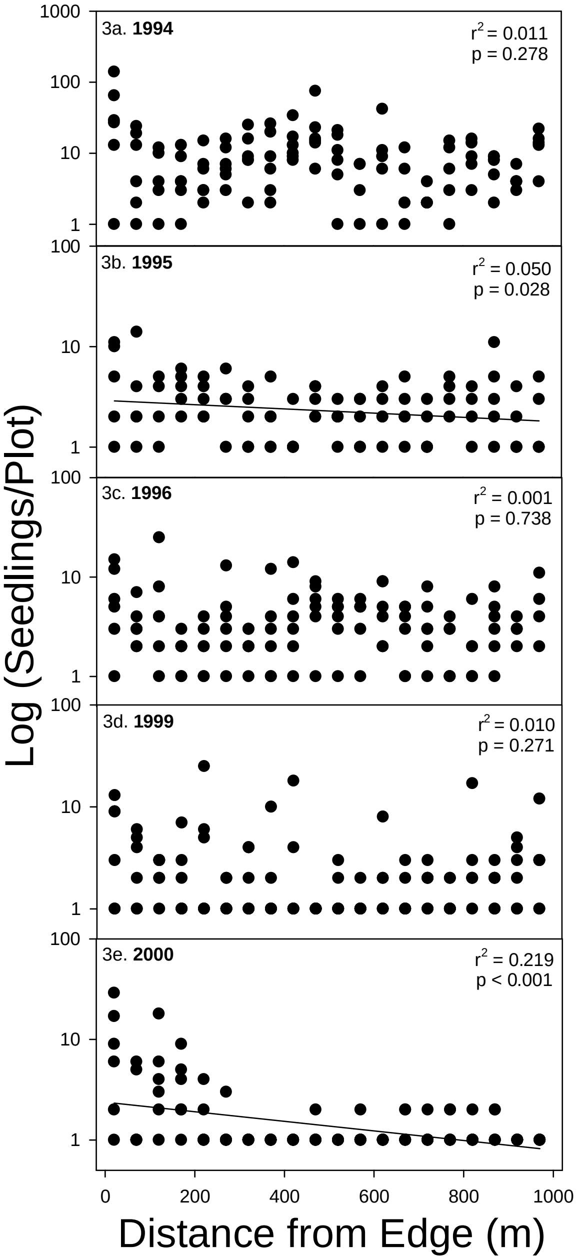


**Figure 5-3. Results from linear regressions (on a log scale) comparing seedling abundance to plot distance from the upwind, unburned edge of study sites located in five burns, ranging from four to ten years in age.**

Results from ANOVAs comparing seedling counts among plot position from the unburned edge within each burn were statistically significant for two burns (Figure 5-4). In the 1994 burn, the plot position closest to the unburned edge had significantly higher mean sagebrush seedling counts than plots in most other positions. However, the plots centered closest to the unburned edge, at approximately 20 m, did not have significantly higher mean seedling counts than plots centered 420 m, 470 m, 620 m and 970 m, from the unburned edge (Figure 5-4a). Thus, in the 1994 burn, plots sampled closest to the unburned edge didn't have significantly different mean seedling counts from plots sampled farthest from the unburned edge. In the 2000 burn, mean seedling counts in the plots closest to the unburned edge were significantly greater than mean seedling counts in plots at all other positions except plots centered 120 m from the unburned edge (Figure 5-4e).

The heterogeneity chi-square analyses yielded two results important to understanding patterns of sagebrush seedling recruitment on the burns sampled. First, seedling counts among plots in each of 28 of the 29 transects, sampled across all burns, significantly ($p < 0.05$) differed from a uniform distribution. Secondly, patterns of seedling abundance along each transect differed significantly ($p < 0.05$), or were heterogeneous, among transects within each burn. Therefore, the plot position from the burn edge with the highest and lowest seedling counts varied greatly from transect to transect and from burn to burn (Figure 5-5). Results from the goodness of fit tests on the Poisson distribution confirmed that seedling distributions tend to be clumped rather than distributed uniformly or randomly at the scale of the plots sampled. In other words, more plots had either higher or lower seedling counts than would be predicted if sagebrush seedlings exhibited a random distribution across the landscape.

Results from the geostatistical spatial analyses indicated that the 1994, 1996, and 1999 study sites showed a random plot distribution based on seedling count. The results from the 1995 and 2000 study sites indicated spatial clustering that was statistically significant (Table 1). Plots in those two burns tended to be autocorrelated based on seedling count (Moran's *I* index), and that pattern of autocorrelation is driven by high count plots being located close to other plots with high seedling counts (Getis-Ord General G clustering index). Thus, the spatial distribution of sagebrush seedlings at the plot scale is significantly clustered in all burns; however, that clustered distribution remains significant at the scale of the study site in only two of the five burns sampled.

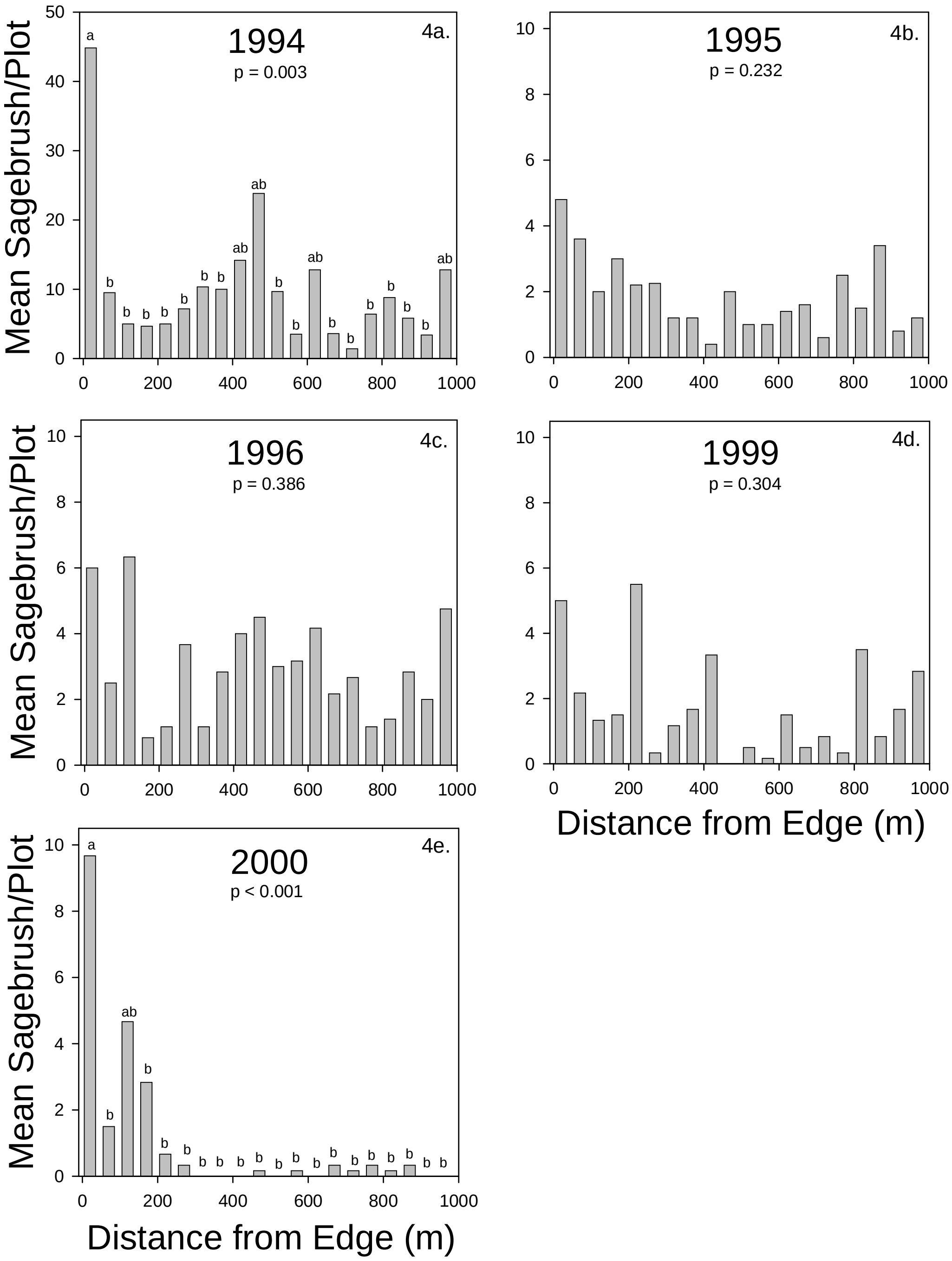
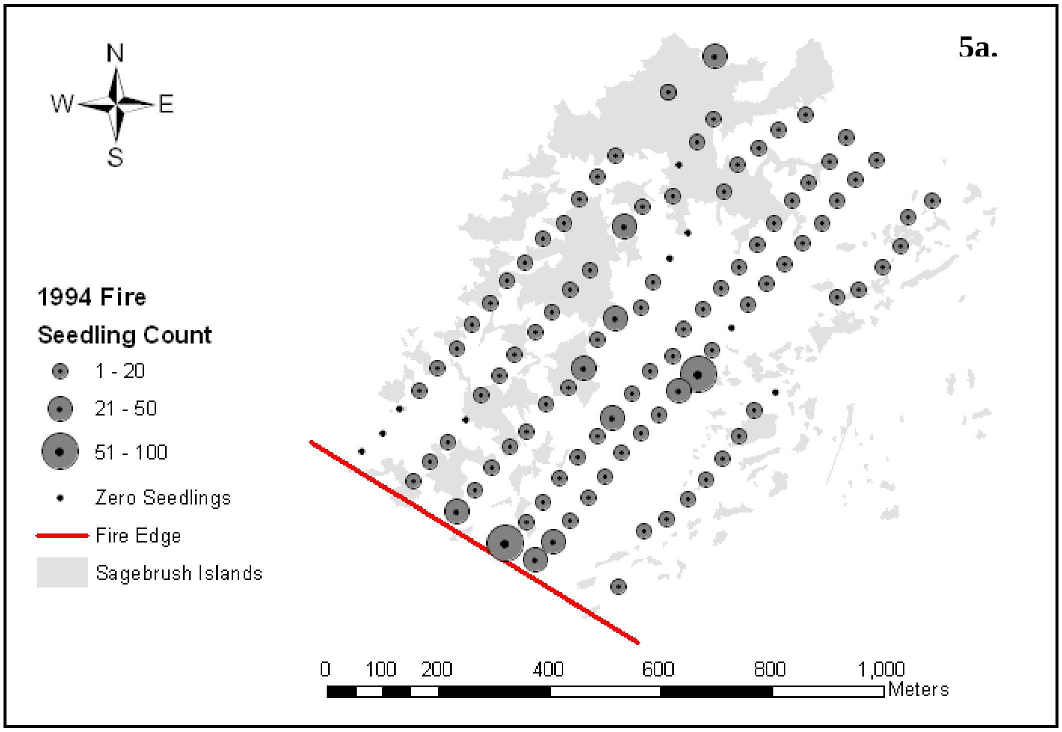


**Figure 5-4. Results from one-way ANOVAs comparing mean seedling abundance among plot positions from the upwind unburned edge in five study sites burned four to ten years previous to sampling.**



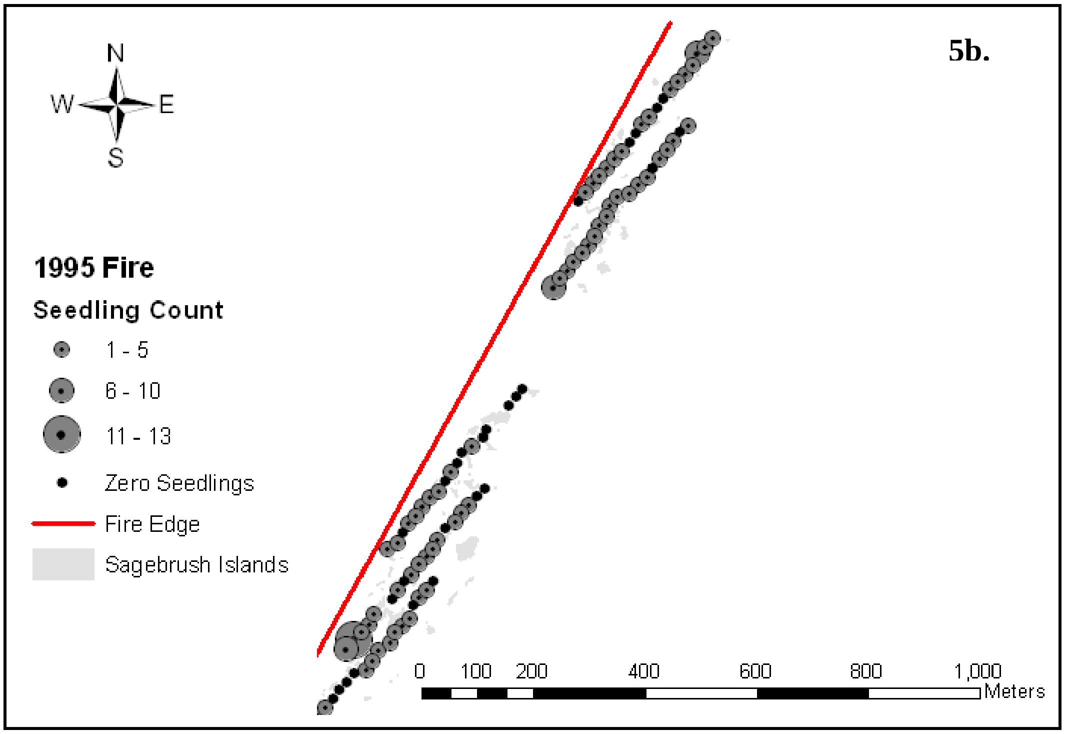
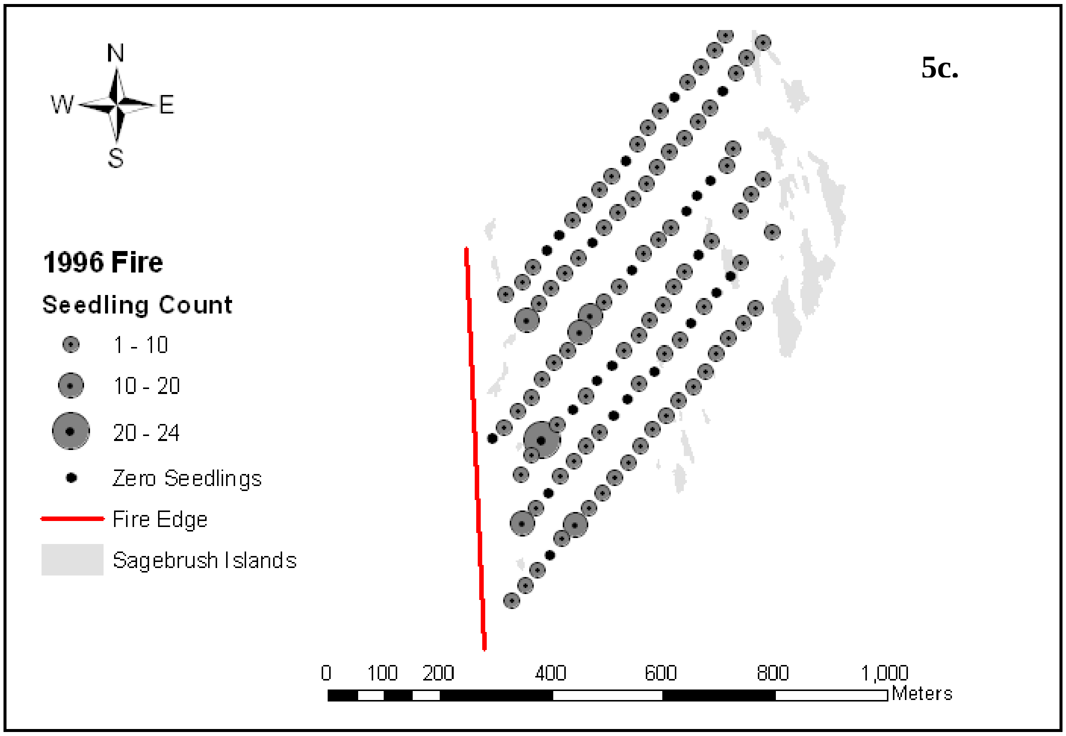


**Figure 5-5. Maps depicting the study site, associated sample plots, and unburned island locations within each burn.**



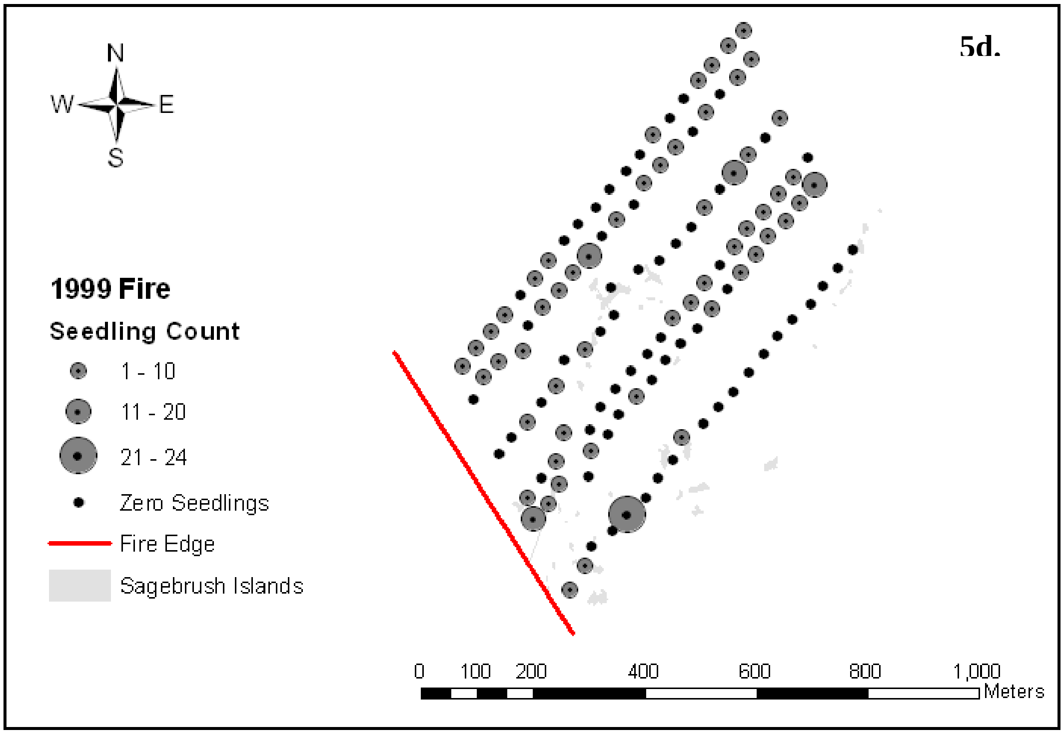


Figure 5-5. (continued)

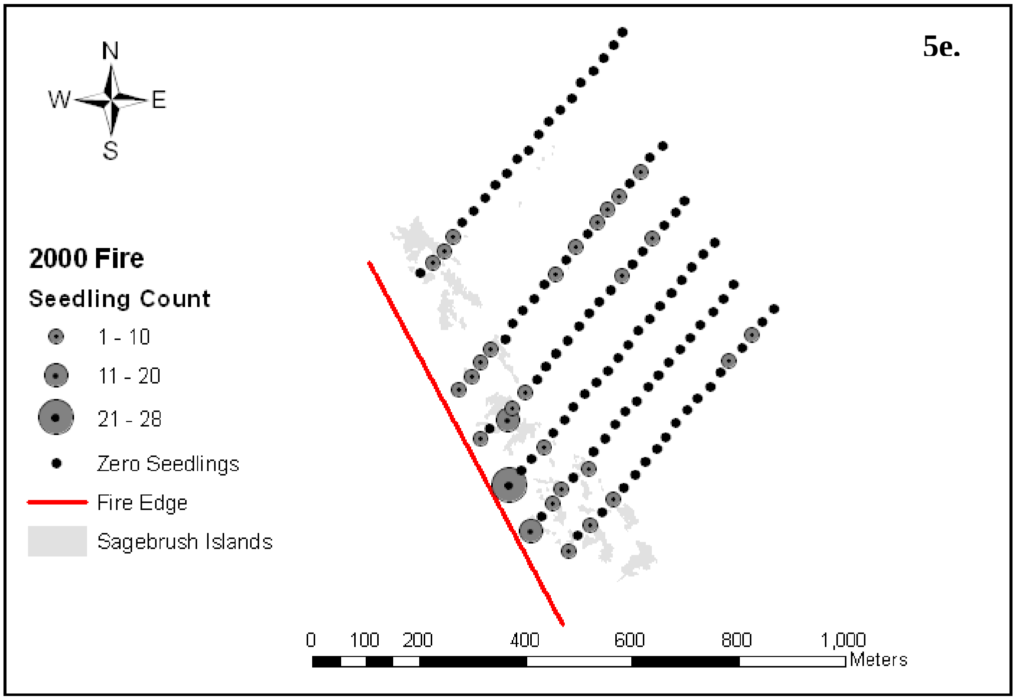


Figure 5-5. (continued)

Table 5-1. Clustering index summary results for each study site in the 1994, 1995, 1996, 1999, and 2000 burns.

| Year | Moran's I | | Getis-Ord General G | |
|---|---|---|---|---|
| | Z-Score | p<0.05 | Z-Score | p<0.05 |
| 1994 | 1.09 | no | 0.33 | no |
| 1995 | 2.38 | yes | 2.03 | yes |
| 1996 | -0.04 | no | -0.76 | no |
| 1999 | 0.94 | no | 0.27 | no |
| 2000 | 6.53 | yes | 5.84 | yes |

Regression analyses, comparing distance from upwind, unburned islands to seedling counts had results comparable to the regression analyses comparing distance from the unburned edge to seedling counts (Figure 5-6). The 2000 burn again had a statistically significant negative relationship, with a similar $r^2$ value (Figure 5-6e). Interestingly, most of the unburned islands in the 2000 burn were located near the burn edge (Figure 5-5e). The 1995 burn also had a significant result with a very low $r^2$ value; however, the relationship was positive, rather than negative in this analysis (Figure 5-6b). No significant relationships were apparent between distance from upwind unburned islands and seedling for the 1994, 1996, and 1999 burns. Because a significant negative relationship occurred in only one of the five burns, and that relationship had a relatively low r$^2$ value, seedling count appears to be even less dependant on distance to upwind unburned islands than it is to the upwind unburned edge.

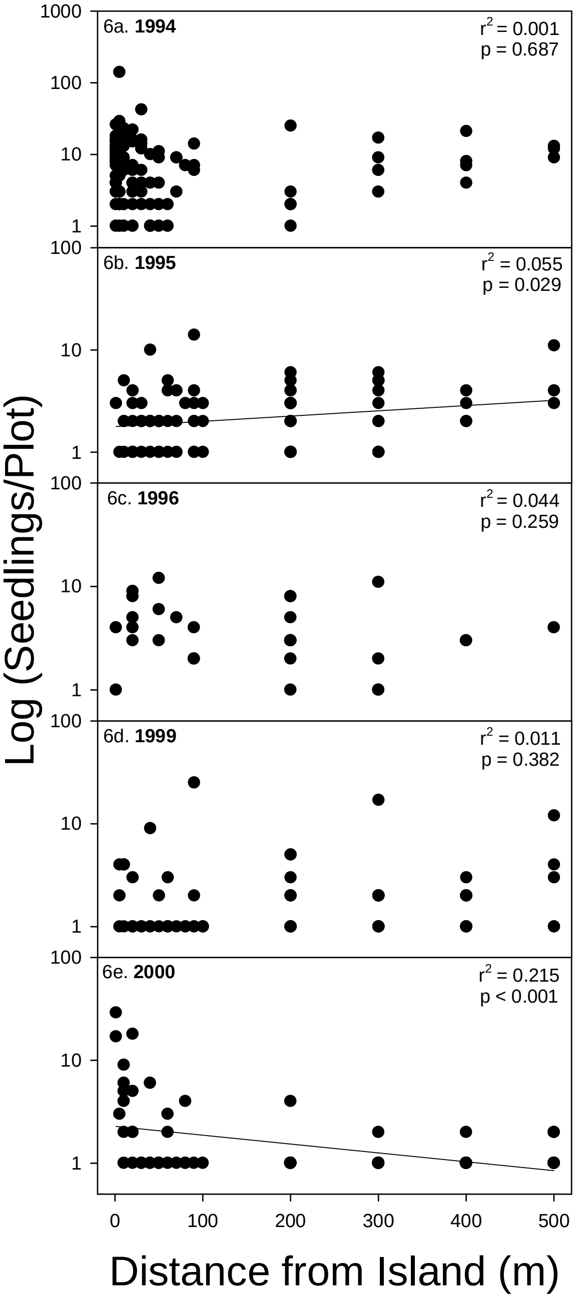


**Figure 5-6. Results from linear regressions (on a log scale) comparing seedling abundance to plot distance from upwind, unburned island in five burns, ranging from four to ten years in age.**

During the ten years prior to the 2004 seedling surveys, total annual precipitation was greatest in 1995 and lowest in 2003. Annual precipitation was greater than the long-term average (208 mm) in only two years, 1995 and 1998. Precipitation during the time period between the 2000 fire and the seedling surveys was substantially lower than precipitation during the other years considered in the study and lower than average annual precipitation (Figure 5-7). Precipitation timing varied dramatically among the ten years between the year subsequent to the first burn and the year the study was conducted. March, May and June precipitation were far greater than average in 1995. In 1996, March, April and May precipitation were above average, and in 1997, precipitation tended to be a little higher than average through the mid to late summer months. During the 1998 growing season, monthly precipitation was greater than average in May, June and July, and in 1999, monthly precipitation was greater than average in April and May. From 2000 to 2004, monthly precipitation for May and June were well below average (Figure 5-8).

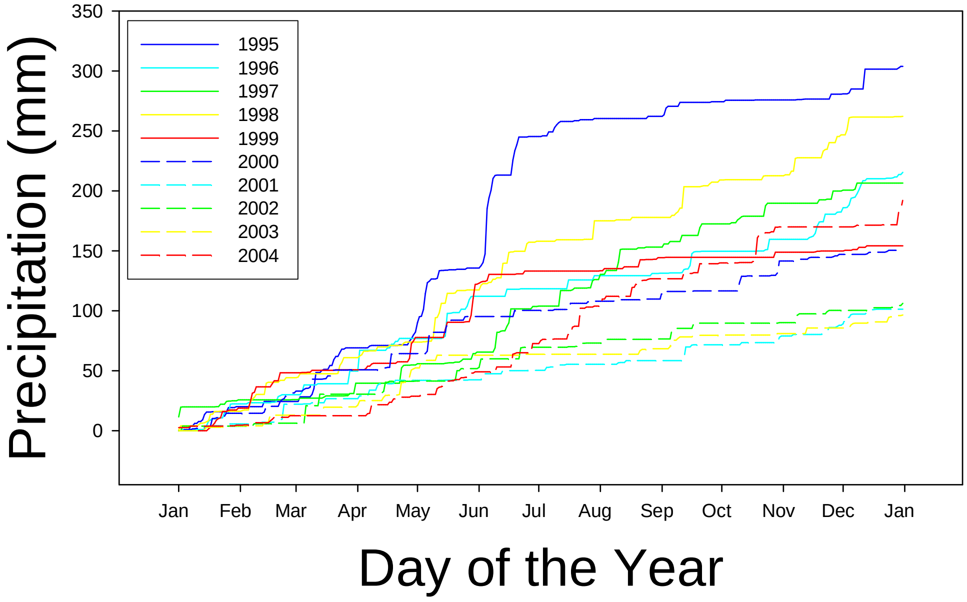


**Figure 5-7. Cumulative annual precipitation during the ten years between the occurrence of the fire on the oldest burn site sampled and the year the data were collected. Data are from INL Central Facilities Area (NOAA, unpublished data), located central to the study sites, about 8.1 km from the center of the study site on the closest burn (2000) and about 26.0 km from the center of the study site on the furthest burn (1996).**

## Discussion

Data collected for the purpose of characterizing natural sagebrush recruitment patterns on unseeded burns failed to clearly support the hypothesized exponential decay model. In three of the five unseeded burn sites studied, regression results did not indicate a statistically significant relationship between distance from the upwind unburned edge and seedling abundance. Low regression coefficients on the two burns that did exhibit statistically significant slopes indicated a very weak relationship. Furthermore, a weak negative distance/abundance relationship with regard to unburned islands was statistically significant on only one burn, and islands on that burn tended to be located close to the burn edge, further complicating the result. Additionally, results from the ANOVA and chi-squared heterogeneity analyses did not indicate any other potentially recognizable distance/abundance relationships with respect to unburned, upwind edges. Several factors including, remnant seed in the seed bank, long-distance dispersal events, and favorable microclimatic conditions may have influenced seedling recruitment patterns more than the exponential decay model used to describe seed dispersal patterns.

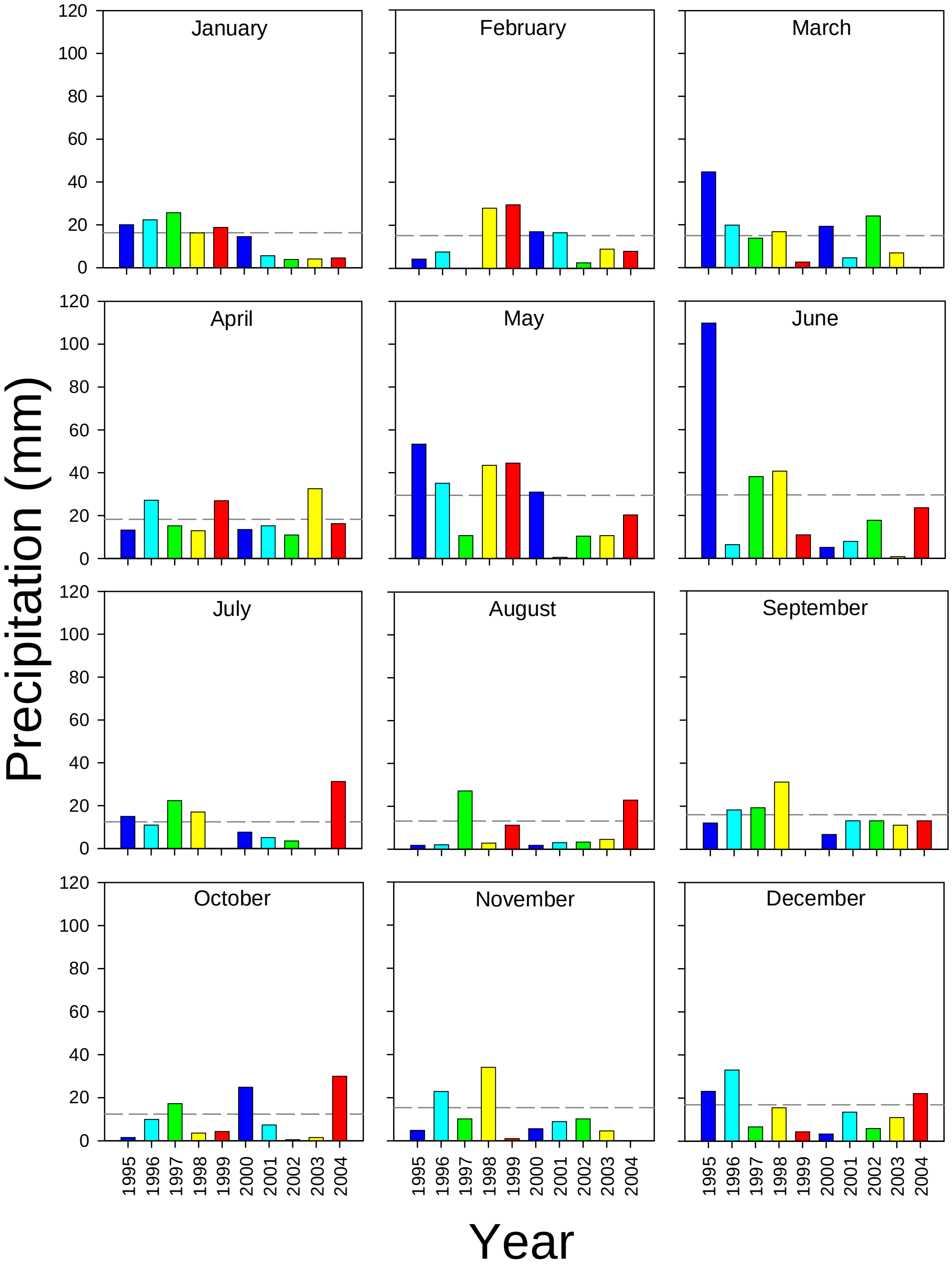


**Figure 5-8. Monthly precipitation during the ten years between the occurrence of the fire on the oldest burn site sampled and the year the data were collected. The dashed line represents average monthly precipitation from 1950 through 2004. Data are from INL Central Facilities Area (NOAA, unpublished data), located central to the study sites, about 8.1 km from the center of the study site on the closest burn (2000) and about 26.0 km from the center of the study site on the furthest burn (1996).**

Several researchers have addressed the persistence of big sagebrush seed in the seed bank (Mueggler 1956, Johnson and Payne 1968, Young and Evans 1989). Many studies indicate that big sagebrush seed is short-lived in the seed bank, usually less than one year (Meyer 1994). Additional studies have demonstrated that very little sagebrush seed persists in the seed bank following wildland fire (Boudell et al. 2002). However, other authors have documented patterns of natural sagebrush recruitment on burned sites that can only be explained by the presence of at least some seed at great distances from the unburned edge, and they have postulated that some of that seed may be residual in the seed bank (DiCristina et al. 2007). Because big sagebrush can produce an enormous amount of seed, even if only a small fraction of seed persisted in the seed bank, germination and recruitment of individuals from that seed may have affected recruitment patterns across the landscape (Chambers and MacMahon 1994, Meyer 1994). Likewise, if a very small proportion of the seeds produced by mature individuals in an upwind unburned edge or island dispersed great distances, they also would have an important impact on spatial patterns of recruitment.

Results from a handful of sagebrush recruitment studies indicate that microenvironmental conditions are extremely important for germination and establishment (Young et al. 1990, Boudell et al. 2002). Ecologists studying patterns of species invasions in other ecosystems have suggested microclimatic conditions may have a much greater influence on spatial patterns of recruitment than seed distribution curves (Robinson and Handel 2000). Thus, occasional dispersal events at relatively remote distances from the seed source may provide enough seed to sites with proper microenvironmental conditions to substantially impact spatial recruitment patterns. Hence, exceptional dispersal events in the tail of the seed dispersal curve may be disproportionately more important to spatial recruitment patterns than the vast amount of seed deposited within a few meters of the unburned edge (Cain et al. 2000, Hastings et al. 2005). Smith et al. (1997) observed that in arid and semiarid ecosystems, such as the sagebrush steppe, vegetation establishment responds to exceptional, rather than average, temporal events like precipitation. Results from our study suggest that exceptional events, like remote seed dispersal, or even residual seed in the seed bank, may also drive spatial patterns of vegetation establishment.

Yarranton (1974) described a nucleation pattern of species invasion in which scattered colonization foci become established, likely through distant dispersal events. These foci then coalesce as individuals mature, ameliorate microclimatic conditions, and produce seed. DiCristina et al. (2007) described recruitment patterns in nearby mountain big sagebrush that approximates spatial recruitment patterns suggested by the nucleation model. The significantly clumped nature of seedling occurrence within plots on our study sites, as indicated by chi-squared analyses of the Poisson distribution, may also lend some support to the nucleation model of invasion for big sagebrush. The aggregation of individuals within the spatial scale of the plot size sampled in this study is consistent with the colonization of favorable microsites, such as rock outcrops that trap snow. Our geospatial statistical results indicate significant clumped distributions among plots in only two of the five study sites, thus, the spatial scale at which recruitment “foci” occur is relatively small. Because these small groups of individuals on relatively young burned sites will eventually mature and produce seed, they likely represent the “foci” described by Yarranton (1974).

Finally, as mentioned above, weather-related events are extremely important to temporal patterns of species recruitment in sagebrush steppe ecosystems. The same weather patterns likely affect spatial patterns of species recruitment in these ecosystems. Precipitation was at, or above normal for many of the spring and summer months during post-fire years for all burns except the 2000 burn. Considerable drought conditions prevailed all of the years subsequent to the 2000 burn until the time of the study, with most spring and summer months well below monthly precipitation averages. Interestingly, the study site located on the 2000 burn was the only one for which we found reasonable statistical support for the negative exponential decay model of seedling abundance from the upwind burn edge.

Some sagebrush establishment research indicates that a substantial amount of seed germinates every spring, but very few of those individuals actually become established (Owens and Norton 1989). A successful establishment event depends on above average summer precipitation and favorable microclimatic conditions (Young et al. 1990, Boudell et al. 2002). Above average precipitation probably contributed to successful recruitment by creating favorable microsites at greater distances from the burn edge on the four older burns studied. Below average summer precipitation, may have limited the number of favorable microsites on the 2000 burn to the areas around large established shrubs at the unburned edge and around unburned islands, which also happened to be located near the unburned edge. The unburned edge and unburned islands, for example, may have created a snow accumulation effect that was more important for germination and establishment in drought years than it was in years with average or above average precipitation.

## Management Implications

Seedling recruitment mechanisms, other than the exponential decay model of the seed distribution curve, likely have an important effect on spatial patterns of sagebrush recovery. If recruitment patterns were closely tied to the seed distribution curve, natural sagebrush recovery would take thousands of years on large burns, based on maximum seed movements of thirty meters per year. Instead, several decades to a few centuries are currently being documented as the amount of time needed for natural recovery of Wyoming big sagebrush cover to pre-burn levels (Baker 2006, Colket 2002). It is important, therefore, to begin to understand and consider the mechanisms driving spatial recruitment patterns, beyond the exponential decay model of seed dispersal, in order to further accelerate sagebrush recovery in burned sagebrush steppe ecosystems.

The results of this study indicate that seed availability may not be as important a limiting factor for sagebrush recruitment as is appropriate microsites for establishment. Therefore, using management approaches that focus on overcoming establishment limitations may be a more effective approach to increasing recruitment than using approaches that emphasize overcoming seed dispersal limitations. Because weather patterns likely influence spatial patterns of seedling establishment and are highly variable in arid and semi-arid environments, planting sagebrush seed is a gamble at best. However, planting sagebrush seedlings circumvents the critical establishment period during which many recently germinated seedlings die, and may thereby increase successful establishment.

Additionally, planting seedlings in islands, as a means of overcoming seed dispersal limitations, may not be the most effective management strategy if seed availability is not the primary limiting factor. The nucleation hypothesis of invasion more accurately describes spatial recovery patterns on the burns we studied than did the exponential decay model. Moreover, “nucleation foci” tend to occur at the relatively small spatial scale of the plots used in this study, indicating that large islands of seedlings are not necessary for successful establishment. Therefore planting individuals or small groups of individuals spaced widely across a burned site, to effectively increase the number of colonization foci, may be the most effective use of resources for accelerating seedling establishment.

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# SECTION 6: ANALYSIS OF VEGETATION COVER SAMPLING METHODS USED ON THE TIN CUP PLOTS

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## Introduction

The abundance of individual plant species and the relative composition of those species within a plant community are often used as criteria in determining the resilience, condition, invasibility, and sustainability of rangelands (Elzinga et al. 1998). Vegetative cover estimation is one very common technique used to assess post-fire recovery of plant communities. Because precise and representative vegetation cover estimates are critical for hypothesis testing and making management recommendations, we considered our approach for estimating plant cover on this project very carefully. The project team engaged in numerous conversations about the experimental approach and sampling design of the portion of the study addressing short-term recovery trajectories on the Tin Cup fire with and without livestock grazing. These discussions were instrumental in focusing the data collection effort towards an ecologically meaningful, statistically sound experimental design.

Numerous statistical models, plot designs, and measurement methods are scientifically sound and well documented for use in sagebrush steppe systems and should be carefully considered with respect to the objectives of a given project. However, the analyses we used to address experimental design and vegetation sampling in this study may be applicable to research and monitoring efforts in other sagebrush steppe rangelands. The intent of this chapter is to document the experimental/sampling design choices made by the project team for the Tin Cup Fire Recovery Project and the analyses used to make and/or support those decisions. We fully recognize that approaches beyond the scope of this chapter or this project may be equally or more valid for other data collection efforts. A synopsis of the final design and the analytical process leading to that design are presented here simply as a tool for those sampling cover in sagebrush steppe ecosystems for the purpose of research or monitoring.

## Experimental Approach

The study plots on the Tin Cup burn and the vegetation cover estimates collected therein were initially to be used to address two study objectives, short-term, post-fire recovery trajectories and the effects of livestock grazing on those trajectories. In order to address both objectives simultaneously, we used an experimental design based on two-factor analyses of variance with both factors (grazing treatment and number of growing seasons following the fire) repeated. Sampling variability can be reduced and the power to detect significant changes in vegetative cover can be substantially increased by using a repeated measures design (Elzinga et al. 1998). We set up the experiment such that permanently located, paired plots would be sampled over three growing seasons. Each pair of plots included one 20 x 50 m plot centered within a 1 ha, square livestock grazing exclosure (native grazers could still gain access), and an adjacent 20 x 50 m plot that did not limit access to livestock. The plot that did not limit livestock grazing was also placed within a 1 ha buffer so that it was not located adjacent to a fence, road, or any other obvious feature that may have affected vegetation cover. Both plots within each pair were located such that variability in topography and soils within and between the plots was minimized. The paired nature of the plots allowed us to consider the grazing treatment factor of the statistical model “repeated,” which in turn controls for much of the variability inherent in the plant community across the large spatial scale at which we sampled. We also considered the factor “number of growing seasons post-fire,” repeated with regard to the statistical model because we resampled permanent plots every growing season.

Fire recovery studies have often been criticized with regard to experimental design. The primary design and sampling concern affecting these experiments is pseudoreplication (Hurlbert 1984). It has been argued that a post-fire, burned site is one experimental unit, or treatment, and that sampling several plots within that unit constitutes subsampling (Krebs 1999). Therefore, the degrees of freedom, or sample size, in a statistical model can only be increased by increasing the number of burns sampled rather than the number of plots sampled in one burn. This criticism may very well apply to manipulative experiments in which burned treatments are compared to unburned control plots. Of the two hypothesis tested on the Tin Cup burn, only one can be considered manipulative, the exclusion of grazing. The effects of grazing exclusion on post-fire recovery trajectory were addressed in this experiment using a randomized block design, with physically independent grazing exclosures. Thus the grazing factor of our experimental design can be considered statistically independent, so long as conclusions from this study don’t extend beyond the burn being investigated.

The hypothesis we tested concerning short-term recovery trajectories, did not involve the use of unburned control plots, as is often the case in post-fire recovery studies. Instead, we compared vegetation cover on burned plots from one growing season to the next. Hurlburt (1984) describes a mensurative experiment as one testing hypotheses about space or time. The recovery trajectory hypothesis tested in the Tin Cup project clearly falls into this category. Appropriate dispersion of replicate samples, or plots, with regard to the hypothesis being tested is the most important consideration in avoiding pseudoreplication in mensurative experiments (Hurlbert 1984). Variability in plant cover from one growing season to the next must be adequately characterized across the burn in order to make valid conclusions about recovery trajectories on the burn as a whole. Plot pairs in this experiment were arranged such that they were widely dispersed and captured much of plant community heterogeneity across the burn, minimizing the potential for pseudoreplication. The nature of the hypotheses being tested coupled with the criteria used for plot placement allowed us to consider the plot the experimental unit for this study.

The project team collectively determined that an exclosure size of 1 ha was the minimum necessary to buffer the study plots from any impacts of livestock grazing nearby. We also wanted each pair of plots to remain statistically independent of one another to minimize any potential pseudoreplication or spatial autocorrelation among plot pairs, as described above. Therefore, plot pairs were spaced about 1 km apart. The minimum exclosure size, amount of distance needed between plot pairs, and estimated annual sampling effort constrained the number of plot pairs to ten. Since statistical power is primarily a function of the number of plots sampled (Glantz 2002), we essentially defined our power to detect changes in vegetative cover (and all of the other parameters we sampled) by choosing to use ten plot pairs. Although much of the statistical power of a given experimental design results from sample size (number of plots sampled), power can be improved by minimizing or controlling for variation elsewhere. In the case of this project, we were able to control for landscape-scale variation in the plant community and the variability associated with sampling over multiple growing seasons by using the repeated measures design outlined here.

## Sampling Design

As with spatial and temporal variability related to plant community heterogeneity, sampling error is an additional type of variability that can affect the power of an experimental design and can be managed to improve the statistical power of an experiment. Sampling error is the variability associated with making vegetation cover estimates by subsampling vegetation within a plot (Elzinga et al. 1998). Because the number of plots on this project, as with most projects, was constrained by financial resources, and we had chosen the most powerful experimental design possible, the project team focused on increasing power by decreasing sampling error. Two vegetation cover sampling methods were considered. Both were objective, quantitative measures since subjective methods like ocular estimation don't reasonably lend themselves to precise, repeatable estimates by multiple field crews over multiple growing seasons (Elzinga et al. 1998). Choosing the method with the greatest precision and by consequence, the least sampling error, would improve our ability to accurately estimate vegetation cover and to detect real differences in cover statistically. The project team was also interested in determining which sampling method was most cost-effective.

Initially, we chose a point-intercept sampling method described by Floyd and Anderson (1987), in which points are sighted using a 0.5 x 1.0 m frame (henceforth referred to as "point sighting frame"). Thirty-six points are arranged on 10 cm centers and are sighted through two sets of crosshairs. Point sighting frames are typically located within a plot using a stratified random sampling design. A point intercept method described by Herrick et al. in a 2002 draft of a sampling manual published three years later (Herrick et al. 2005) was also considered. This method, as it was described in 2002, involves sampling 150 points along three transects using a pin as a sighting tool (henceforth referred to as "point intercept"). The three, 50 m transects radiate from a center stake to form a "spoke design."

Both plot layout and measurement method were considered in the comparisons between the two sampling designs. With regard to plot layout, we determined that we would locate the sample units using the stratified random plot layout rather than the radial plot layout, regardless of which measurement method was used. The stratified random approach to sample unit (point or frame) placement within a rectangular plot was chosen to alleviate concerns of oversampling the plot center and undersampling the periphery of the plot, as is common with radial sampling designs.

Figure 6-1 demonstrates the radial sampling problem. Half of the plot area is located inside the red circle and half is located outside of that circle. If sampling points are located randomly along three transects radiating from the middle, about 70 % of those points are located within the center half of the plot and about 30 % of those points are located within the outer half of the plot. Thus, points near the center of the circular plot tend to be autocorrelated and points near the periphery of the circular plot tend to be overdispersed. The spatial patterns inherent to circular plot designs, therefore, violate assumptions of random sampling (Zar 1999). Additionally, achieving the sample unit dispersion necessary to make accurate vegetation cover estimates is unlikely with circular plot designs (Hurlbert 1984).

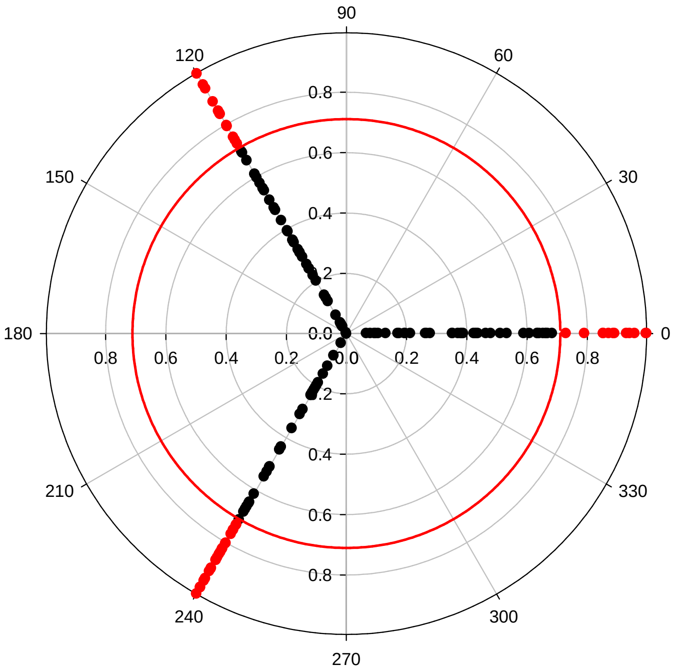

**Figure 6-1. Sample unit distribution using a circular plot with three transects radiating from a center point. One half of the plot area is contained within the red circle and one half of the plot area is contained between the red circle and the outer plot edge. Sample units within the central half of the plot area are represented by small black circles, while sample units within the peripheral half of the plot are represented by small red circles. When sample units are randomly located along the three transects, about 70% of the sample units occur within the centermost one half of the plot area, and about 30 % of the sample units occur within the peripheral one half of the plot area.**

In order to better understand the precision of each measurement method, we plotted cumulative mean and standard deviation curves for cover of several functional groups. The functional groups used for the initial analyses are a subset of those typically used for vegetation analyses on the INL. We compared the two methods using point sighting frame data from the 2001 INEEL Long-Term Vegetation Plots (see Anderson and Inouye 2001 for a description of these data). Out of the 100 plots in the data set, we selected data from only those from the “core” plots. These “core” plots were previously reported by Anderson and Inouye (2001) to be most similar and representative of the sagebrush steppe vegetation community type. For the analysis in which the frame was considered the sample unit (point sighting frame method), we randomly selected a frame from a randomly selected plot. We selected a random point from a random frame from a random plot for the analysis in which the point was considered the sample unit (point intercept method). As each point or frame was selected, mean cover and its standard deviation were calculated based on that data point and all preceding data points. A cumulative curve for mean cover and SD were plotted. The analysis included 50 frames and 300 points. We selected 50 frames initially as a target because that has been traditionally considered as sufficient to characterize a plot at the INL. Although sampling 150 points for point interception was recommended, we targeted 300 points because the method described by Herrick et al. (2005, as presented in the 2002 draft) is intended for use in monitoring plans and the project team felt that more precise estimates would be necessary for assessing grazing impacts and characterizing recovery trajectories, as was the goal of this study. We used a minimum of 2-meter point spacing based on data published for sagebrush resource island spatial patterns (Halvorson et al. 1994). Each time a frame was selected, that frame and the frames from either side of it were eliminated from further selections to minimize autocorrelation in this comparison.

The goal of this analysis was to determine how many sample points or frames were required for the mean to stabilize and to define the lowest possible variability, or standard deviation, for the plant community under consideration. The number of frames or points at which a vegetation cover mean stabilizes is the level of effort at which the most precise estimate can be achieved. Any frames or points sampled beyond that level will not improve the precision of the estimate. The magnitude of variation at which the standard deviation of the mean stabilizes is essentially a measurement of the amount of variation inherent in the sampling method and in the plant community. Because the area under the stabilized standard deviation curve reflects natural variation in vegetation cover, increasing sampling units will not further decrease the standard deviation and the error related to sampling has been reduced to the extent possible using the given sampling method.

For the point sighting frame analyses, the mean and standard deviation of most functional groups began to stabilize between 20 and 30 frames (Figure 6-2). The cumulative mean curve for all functional groups was relatively flat at 50 frames. Conversely, in the point intercept analyses, the mean and standard deviation curves only approached stabilization for three of the six functional groups analyzed, even at the maximum number of points used in the analysis. The functional groups for which mean cover began to stabilize were bare ground, litter, and green rabbitbrush. These functional groups were also the most abundant, in terms of cover, of the six functional groups considered. This result indicates that the point intercept method would not adequately characterize vegetation cover for less abundant functional groups, like bunchgrasses and native perennial forbs. Furthermore, the standard deviations calculated from the point intercept method tended to be much higher than the standard deviations calculated from the point sighting frame method (Figure 6-2), which suggested that we could further limit the amount of sampling-related variability in our sample design by using the point sighting frame method. It should be noted that the standard deviation was greater than the mean in all of the vegetation functional groups (not bare ground or litter) for both sampling methods considered. The standard deviation was greater than the mean for bare ground and litter cover using the point intercept method but not using the point sighting frame method. The relatively high standard deviation values resulting from these analyses reflect the heterogeneity of vegetation at the study site and indicate that a great deal of the variability detected by the statistical model is due to variation in the plant community, even at the scale of a plot.

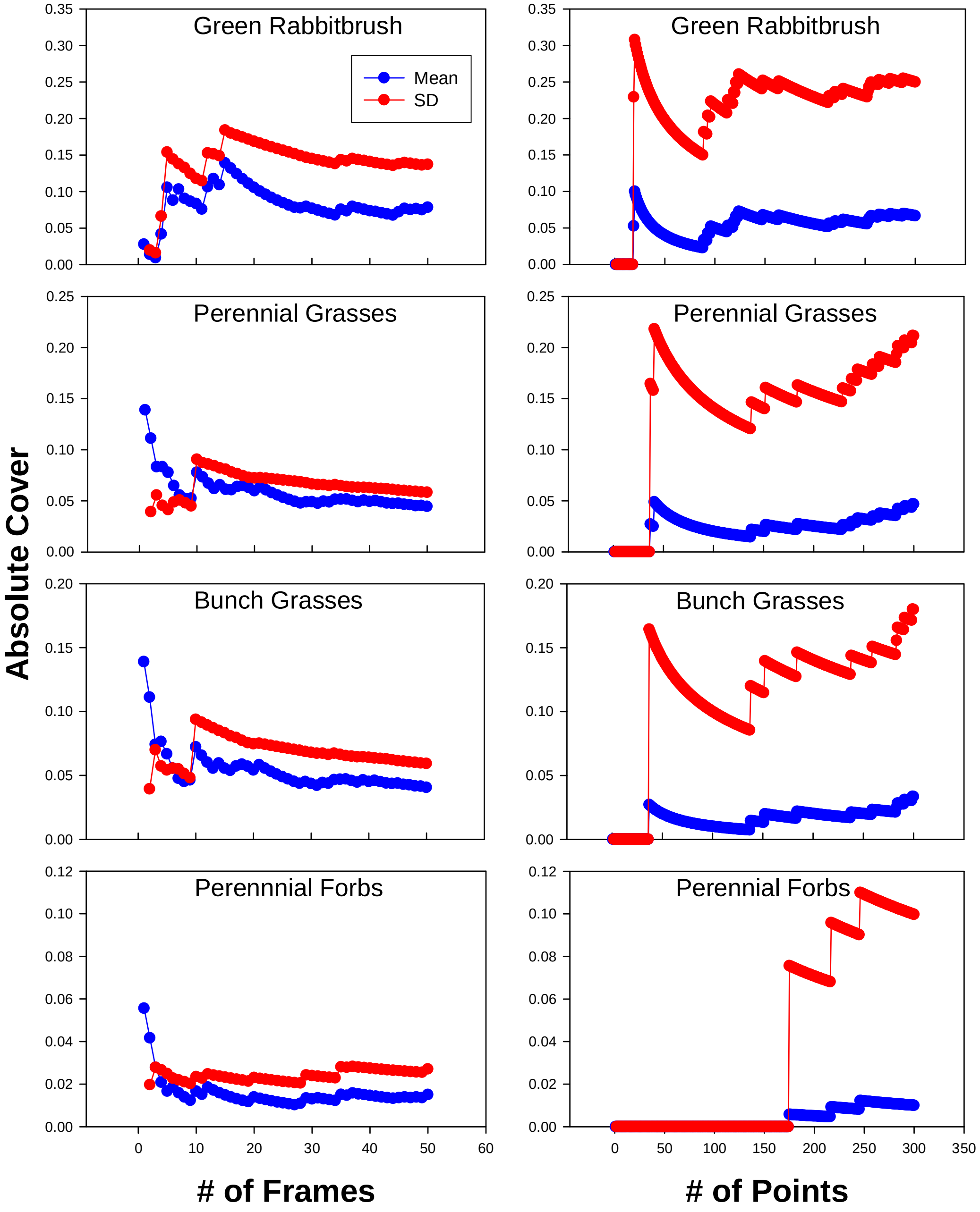

**Figure 6-2. Sample effort graphs used to assess the number of points needed to estimate vegetation cover in a plot using a point interception method and the number of frames needed to estimate vegetation cover in a plot using a point sighting frame sampling method. The graphs were created using 1995 data from the Long-term Vegetation Transects on the INL.**

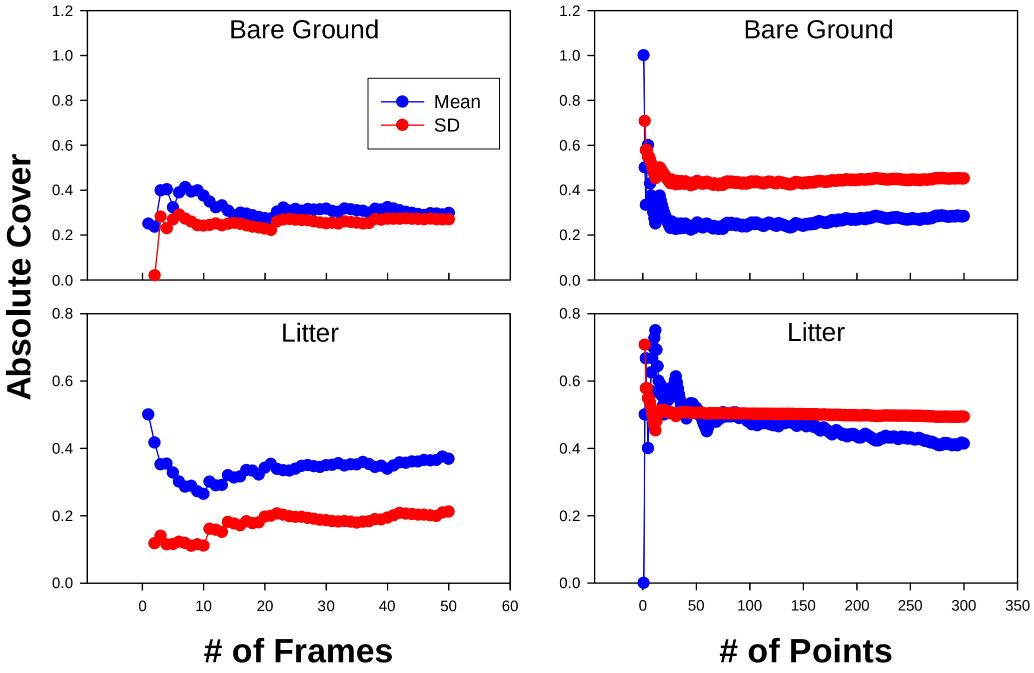

**Figure 6-2. (Continued)**

The amount of time required to sample a plot using the point intercept method is about half an hour for 150 points (one hour for 300 points, Herrick et al. 2005), compared to an estimated two hours for sampling 50 frames using the point sighting frame method. Consequently, some sampling time in the field could be reduced by choosing the point intercept method. In order to gain the level of precision achieved by using the point sighting frame method; however, the number of points in the point intercept method would have to be drastically increased. Increasing the number of points sampled using the point intercept method would offset any cost savings gained by choosing that method. Moreover, much of the sampling effort and related costs of sampling a plot are associated with locating that plot, setting up measuring tapes, measuring plant community characteristics other than cover, and in the case of point-based cover sampling, sighting a point. The effort of sighting a point is actually reduced using a point sighting frame since 36 points are sighted with one frame placement, as opposed to the effort of sighting each individual point using the point interception method. Nonetheless, compared to the sampling effort as a whole, neither sampling method would have been cost-prohibitive.

Based on the results of the precision analyses and the cost considerations described above, the project team decided to move forward with cover sampling using the point sighting frame method. We wanted to distribute the point sighting frames across the plots so as to achieve a good balance between randomness and dispersion. Additionally, we were concerned with avoiding autocorrelation or other artifacts of sampling related to the spatial patterns inherent to the plant community. To address all of these sampling issues, we chose a stratified, random sampling design. Using the 20 x 50 m sized plot described above, we established ten transects perpendicular to the long edge of the plot. To guarantee good point sighting frame dispersion, the transects were systematically spaced at 5 m intervals, starting 2.5 m from the short edge of the plot. Five contiguous point sighting frames were sampled from a random starting point along each transect (Figure 6-3). Point sighting frames were placed contiguously to address spatial autocorrelation issues. Halvorson et al. (1994) described a spatial pattern of about 2.0 m for sagebrush and associated resource islands in sagebrush steppe rangelands. Thus, individuals that are closer together than the minimum scale of pattern in the community are likely to be sampled using a single point sighting frame. This sampling issue is also of concern and has been addressed in line intercept sampling by ensuring the line is "long enough to include all phases of any mosaic pattern that may be present" (Greig-Smith 1983). We therefore determined that frames placed continuously over 5 m should adequately overcome the spatial pattern described for sagebrush steppe.

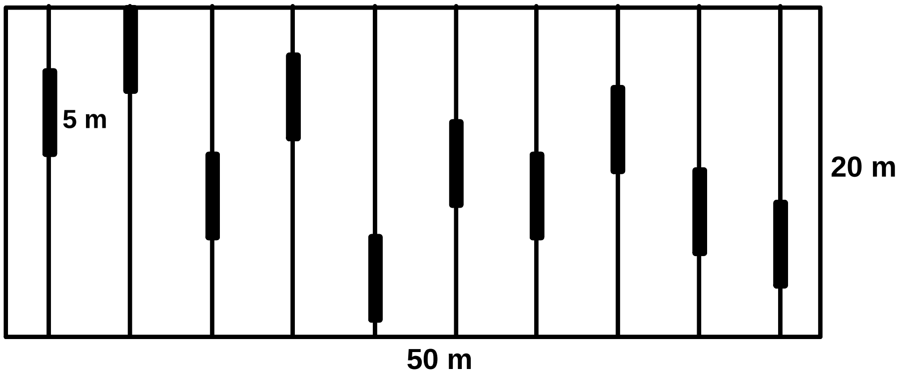

**Figure 6-3. An example of the layout used for sampling cover within each plot on the Tin Cup Fire Recovery Study using point sighting frames. The solid black rectangles represent 5 point sighting frames places contiguously, for a total of 50 frames within each plot.**

The initial functional groups chosen for the precision analysis to compare both point sampling methods were further refined for the final plant community analyses. The project team defined the functional groups used in the final analyses by carefully considering the ecological importance and statistical detectability of various groupings classified by growth habit, longevity, and nativity of individual species. The functional groups ultimately chosen for the final vegetation cover analyses include; total vascular plant cover, native resprouting shrubs, native perennial grasses, native perennial forbs, native annual and biennial forbs, and introduced species. The native perennial forbs functional group was further broken down into leguminous and non-leguminous perennial forbs. The introduced species functional group was also divided into introduced annual grasses and introduced annual and biennial forbs. As with the precision analyses, we also sampled bare ground and litter. The use of these functional groups allowed us to understand the relative importance and stability of major growth forms like shrubs, grasses, and forbs in a recovering plant community. The role of annual versus perennial species on post-fire recovery trajectories were also easily characterized using these functional groups. Finally, comparing the cover of non-native, and potentially invasive species, to the recovery trajectories of native species was straightforward with the functional groups defined by the project team. The team initially wanted to analyze cover data for rhizomatous and bunch grasses separately. However, the cover of grasses representing each growth form was so variable relative to total grass cover within each plot that our ability to detect differences in those growth forms among plots would have been nearly impossible. Likewise, non-resprouting shrubs and biological crusts did not occur with enough regularity or abundance to consider statistical analysis of them.

Vegetation cover data were analyzed using aerial cover for shrubs and forbs, excluding reproductive structures, and basal cover of grasses. We used aerial cover of shrubs and forbs because above ground biomass of individuals of those growth forms tends to remain relatively stable from one growing season to the next (Greig-Smith 1983). The above ground biomass of perennial grasses, on the other hand, is considered to be highly variable from one season to another, depending on soil moisture conditions. While measuring aerial cover of grasses can be an important indicator of the amount of biomass produced during a given growing season, we were more interested in the relative importance of native, perennial grasses as a component of the entire plant community over the long term. Although both aerial and basal cover were collected for grasses, we ultimately chose to analyze grass cover data using basal cover, which provided a much more stable metric for characterizing post-fire recovery trajectories.

## Internal Review of the Design

We established plots, built grazing exclosures, and sampled cover in 2002 according to the experimental design and sampling protocols described above. Using that vegetation cover data, we reassessed our approach for estimating plant cover on the Tin Cup burn in terms of the statistical power to detect significant differences. In order to determine whether the data collected were sufficient to address the study objectives, the project team identified a criterion of minimum detectable change (MDC) at 5% absolute cover for all functional groups. For example, if the average absolute cover of resprouting shrubs on grazed plots was 5%, we wanted to be able to detect an average absolute cover of 10% or greater on plots located within grazing exclosures with statistical significance.

We ultimately intended to address the effects of grazing treatments and changes in plant community composition over time by analyzing vegetation cover data using two-way repeated measures ANOVAs, as described above. However, we did not yet have data from multiple growing seasons with which to test for trends over time. Therefore, means and MDC’s of the functional groups used for this analysis were calculated assuming the data would be analyzed using paired t-tests to assess grazing effects (based on plot pairs). In effect, we assessed the magnitude of vegetative cover difference we could detect on one factor of the two factor statistical model. We calculated MDCs at $\alpha = 0.05$ and $\beta = 0.20$. MDCs ranged from less than one percent absolute cover for native perennial grasses to almost nine percent for litter (Table 6-1). MDCs tended to be higher for functional groups with higher average absolute cover. For two functional groups that consist primarily of annual species, native annual and biennial forbs, and introduced species, the MDCs were greater than the average mean cover. This is likely a consequence of the large variability in distribution of annual species. Overall, five of the eight functional groups identified for final cover analyses met the 5% MDC criterion set by the project team. The three functional groups that did not meet the criterion were; total vegetation, bare ground, and litter. Total vegetation and bare ground exceeded the 5% desired MDC by only 1%.

**Table 6-1. Means and Minimum Detectable Changes (MDCs) for plant cover by each functional group. Means* were pooled across grazing treatments and MDCs** were calculated at $\alpha = 0.05$ and $\beta = 0.20$.**

| Functional Group | *Mean** | *MDC*** |
|---|---|---|
| Native Shrubs | 4.63 | 3.44 |
| Native Perennial Graminoids | 1.53 | 0.67 |
| Native Perennial Forbs | 3.63 | 2.24 |
| Native Annual and Biennial Forbs | 0.55 | 1.10 |
| Introduced Species | 0.71 | 1.01 |
| Total Vegetation | 11.05 | 6.04 |
| Bare Ground | 72.98 | 5.83 |
| Litter | 15.89 | 8.96 |

Based on the MDC analyses, we largely considered the experimental design, including the vegetation cover sampling methodology, sufficient to address hypotheses about short-term, post-fire recovery trajectories and the effects of grazing on those trajectories. In some cases, the MDCs associated with the statistical model were small enough as to be unimportant ecologically. For example, while statistically significant, a difference of less than one percent absolute cover for native perennial grasses, probably doesn’t translate to notable differences in plant community structure or function. So, while we were confident that our chosen design was as statistically powerful as possible, given the constraints on the number of plots and the variation inherent in the plant community, we did have to remain cognizant of understanding whether statically significant differences were also ecologically meaningful.

The portion of the Tin Cup Fire Recovery Project discussed here was initially designed to test two hypotheses, one concerning recovery trajectories and the other addressing the effects of grazing on those trajectories. The grazing factor of the two factor design was ultimately dropped in the final report since utilization data indicated that utilization was generally less than a few percent, far below the moderate level of 30-40 percent outlined as a requirement in the study objectives. Both the recovery trajectory and grazing effects hypotheses were discussed in this chapter since they illustrate the experimental design issues the project team considered in this study, some of which may be applicable to other research or monitoring efforts.

## Outside Statistician Review of the Design

Some concerns about the sampling design of the project became apparent through internal discussions during and after the first year of data collection. These concerns could not be resolved with the analyses described above. Therefore, an independent statistician was consulted to verify the results of the analyses concerning the experimental approach, sampling methodology, and the resulting statistical power of the vegetation cover aspect of the Tin Cup Fire Recovery Study. Specifically, Charles D. Bonham, Ph.D. was tasked with 1) Conducting a review of the study design, 2) Assessing the ability of the design to meet the 5% MDC requirements outlined by the project team, 3) Making recommendations for improving the design of the study to meet the MDC requirements and, 4) Comparing the merits and errors inherent in both point sampling techniques considered for this study. A brief summary of the results of the statistician's review are presented here; a report detailing all of the results and recommendations from that review is included in Appendix A.

The first objective of the statistician review, reviewing the study design, was addressed both qualitatively and quantitatively. Results from the qualitative assessment indicated that the point sighting frame method is a sound sampling technique with which to estimate cover and that collecting cover data at the species level is important in case functional groups, or guilds, need to be redefined to meet specific statistical or ecological criteria at a later date. The quantitative assessment addressed the study design in terms of experimental design, measurement design, and sample design. The project team's approach to increasing statistical power by decreasing sampling error was considered appropriate, given the spatial and financial constraints on the number of plots in the study. The point sighting frame method selected for sampling vegetative cover provided very precise estimates of mean and variance for the functional groups considered and, the reviewer recommended maintaining the same sampling intensity to maintain the high levels of precision desired for the entire study. The method used for placing point sighting frames within the 20 x 50 m plot is commonly used and well accepted for vegetation field studies. Bonham's conclusion for task one was that the study design was adequate to address both study objectives.

The ability of the study design to meet a 5% MDC in absolute aerial cover of nine functional groups, or guilds, was the focus of the second task in the statistician's review. The reviewer concluded that the 5% MDC was achievable using the current study design for many of the guilds; however, no reasonable amount of increase in sampling effort would yield an MDC of 5% or less for a few of the functional groups, or guilds. Bonham also recommended measuring basal cover of grasses since it is a better indicator of plant community change over time. Averaged at the plot level, most of the functional groups met a normal distribution. But, if the project team were to decide to analyze data at the level of a transect line or of a frame within a plot, the binomial distribution would need to be considered. Overall, the data set resulting from the study design was considered to be very large, both spatially and statistically, and low statistical power for some functional groups resulted primarily from the limited number of plots and the variation inherent within the plant community, not the statistical model nor the sampling design.

Several recommendations concerning the study design and statistical approaches were made to address the third objective of the review. The first recommendation was to consider collapsing some of the guilds recommended by the project team into fewer functional groups. Greater power may be obtained by avoiding the use of functional groups with high spatial variability and very low absolute cover. The data were ultimately analyzed using slightly different functional groups that those initially recommended by the project team in response to this recommendation. Bonham also recommended that the project team consider using confidence intervals to assess plant cover changes rather than statistical models for those functional groups in which the 5% MDC was unrealistic or unachievable. The issue of having functional groups that were difficult to characterize statistically was mostly resolved by redefining the functional groups in response to the first recommendation, thereby making action with regard to the second recommendation unnecessary. The third recommendation from the reviewer related to the integrity of the Tin Cup Fire Recovery Study design was to continue collecting data using the same methodology used in 2002 (as described above) since the design was robust statistically and would easily meet most of the hypothesis testing goals outlined by the project team.

The fourth and final objective of the statistician's review was to compare the two point sampling methods considered for this project. Bonham highlighted the potential sampling errors and advantages of each method. Table 6-2 summarizes the weaknesses of each design and Table 6-3 summarizes the strengths of each design as outlined in the reviewer's report. All of the potential errors associated with the sample design chosen for the project were related to sighting the sample points. To address these issues, we used levels to ensure the line of sight was vertical and, we used relatively small diameter "crosshairs" to minimize problems with visualizing the point sighted through the "crosshairs." Given the advantages of the point-frame method, in terms of precision, repeatability and the amount of data collected as determined for this study, the project team sampled vegetation cover using the same sample design in all subsequent years of the study.

**Table 6-2. Potential sampling errors, as reported by an independent reviewer, related to both sampling methods considered for the Tin Cup Fire Recovery Study (from Bonham, Appendix A).**

| Errors in the point intercept method | Errors in the point sighting frame method |
|---|---|
| Pin placement may have low precision and/or low repeatability | Visual alignment of the point frame grids may depart from vertical |
| Pin drop may cause error by pushing a plant part out of the direct line of the pin | The two-directional intersection may be difficult to visualize |
| Misidentification of living or dead plant parts may be increased | May be difficult to sight beyond the first contact without changing subsequent contacts |
| Wind can cause motion in plant parts when the pin is dropped | The observer is required to maintain a rigid position |
| Species contact with the pin may be influenced by pin placement | Reading 36 intersections may become tedious, resulting in errors |
|  | It is difficult to discern contacts in canopies when foliage is dense |
|  | Shadows may make sighting intersections difficult |

Table 6-3. Potential advantages, as reported by an independent reviewer, related to both sampling methods considered for the Tin Cup Fire Recovery Study (from Bonham, Appendix A).

| Advantages of the point intercept method | Advantages of the point sighting frame method |
|---|---|
| Easy to locate placement of lines in sample area | Precise estimates of small scale patterns |
| Economical for broad estimates of cover | More precise estimates of cover |
| Short training period for observers | More likely to relocated points in the permanent plots |
| Quicker data synthesis because of fewer data points | More precise estimates of changes in cover over time |
| Variations in contacts is spread over a larger distance | Provides the opportunity to estimate cover in three-dimensional space |
| These advantages are based on the assumption that field instructions are easier to understand than the manual | Provides large data sets within and among plots |
|  | Easy to train observers to use |

## Implications for Monitoring Post-Fire Recovery and Livestock Grazing Impacts

Throughout the process of developing the experimental methodology and vegetation cover sampling design for the Tin Cup Fire Recovery Project, the project team identified several principles that may be useful for planning research and monitoring projects in sagebrush steppe ecosystems range wide. These principles are presented here as a set of recommendations to be considered as a critical component of the project planning process so that the appropriate data are collected with enough to precision to confidently address the hypotheses being tested. These recommendations encompass all aspects of experimental design as they relate to hypotheses testing and statistical power for detecting real differences in vegetation cover data.

### 1. Carefully consider the relationship between the experimental design and the hypotheses being tested.

An effort to very clearly define the hypotheses being tested, prior to sampling, can save land management agencies substantial resources throughout the life of a research or monitoring project. Accurately outlining hypotheses allows a project team to identify the level of difference in vegetation cover that is ecologically meaningful. It is important to collect enough data to detect meaningful differences; however, collecting more data than is required to address the project objectives is unnecessary. While determining the amount of data needed, it is also important to consider the quality of that data with respect to the hypotheses outlined for the project. Conclusions reached through statistical hypothesis testing are only as reliable the vegetation cover data used in the statistical models. Additionally, carefully defining the hypotheses to be tested for a given project can assist a project team in determining which type of cover data to collect. For example, both basal and aerial grass cover were collected on the Tin Cup Fire Recovery Project, but we only analyzed basal cover based on the hypothesis addressing changes in plant community composition through time. The sampling effort for this study could have been reduced by omitting aerial cover sampling for grasses, which didn't directly support any of the previously outlined research questions.

### 2. Chose an appropriate statistical model to address the study objectives.

Although vegetation cover data for most research and/or monitoring studies in sagebrush steppe rangelands can be analyzed using any one of several statistical models, some of those models will be more powerful than others. It is therefore very important to choose the model with the greatest statistical power for testing the hypotheses outlined for a given project. In general, parametric statistical tests, such as Student's t-tests or ANOVAs, tend to be more powerful than non-parametric tests (Elzinga et al. 1998). However, vegetation cover data from sagebrush steppe ecosystems often don't meet the normality assumptions of parametric tests and can't easily be transformed to do so. Some statisticians have argued that ANOVA and similar models are relatively robust and aren't likely affected by deviations from assumptions about normality and equal variance (Elzinga et al. 1998). Others feel strongly that non-parametric tests should be used when a normal distribution cannot be approximated using transformations (Bonham 1989). Thus, the project team should develop a strategy for dealing with non-normal data in such a manner that it doesn't compromise the statistical power of the model chosen for the project.

Paired or repeated measures designs help control for the variability inherent within plant communities in both space and time, thereby increasing the power of a statistical model (Zar 1999). The use of permanent plots can greatly improve the likelihood of detecting real changes in vegetation cover through time and should be an important consideration in the statistical approach to a study, especially in monitoring projects where the primary hypothesis being tested is about changes through time (Greig-Smith 1983). Likewise, when addressing hypotheses about a treatment (fire, grazing, etc.) the use of paired plots can increase the power of the chosen statistical model (Zar 1999). Permanent, paired plots were used to test the hypotheses outlined for the Tin Cup Fire Recovery Project, making the statistical design "repeated" both spatially and temporally. This proved to be a very powerful statistical approach, as indicated by the low MDCs, for detecting differences in vegetative cover among sample years and between grazing treatments.

### 3. Ensure that the sample design adequately characterizes variability in plant cover.

Pseudoreplication often occurs as a result of testing statistical hypotheses using data that do not adequately characterize the study site about which conclusions are being drawn (Hurlbert 1984). Testing hypotheses comparing burned areas to control plots may necessitate the use of several burn treatments to avoid pseudoreplication. To address pseudoreplication concerns in fire recovery studies testing hypotheses about time, plots should be placed such that they sufficiently characterize variability in vegetative cover across the study site. Thus, the distribution of sample units within a plot should be considered an integral part of the experimental design. For example, a radial sampling approach was considered for estimating vegetation cover on the Tin Cup Fire Recovery Project, but was later abandoned since radial designs tend to oversample the center of a plot, which would not result in adequate dispersion of sample units to accurately characterize that plot. Finally, spatial patterns inherent to the plant community need to be considered when addressing sample unit dispersion. If the spatial pattern of a plant community is two meters, then the placement of the sample units within a plot should capture all of the variability within at least two meter intervals (Greig-Smith 1983).

### 4. Define the relationships between sampling error, plant community variability, and statistical power.

A thorough understanding of the error associated with a specific sampling design in a specific plant community allows a project team to both minimize that error when possible and gain a realistic perspective of which statistical conclusions can or can't be made as a result of the research or monitoring effort. Some sources of statistical error, such as those associated with sampling vegetation cover, can be minimized by increasing the number of plots sampled, improving the precision of the cover estimates, and choosing sampling techniques with the lowest possible associated error (Elzinga et al. 1998). Using point sighting frames as a sample unit rather than individual points may be one approach for reducing sampling-related error and increasing precision, as was demonstrated during the course of designing the project described above.

Some sources of statistical error, such as those related to the spatial and temporal variability inherent in a plant community, can't be minimized, but can affect the power of a statistical model. It is important to characterize the statistical error associated with plant community variability because it allows a project team to quantify statistical power, and therefore the ability of the statistical model to detect real differences in plant cover. The limitations on statistical power due to heterogeneity in vegetation aren't easily remedied, but defining these limitations can help a land management agency understand which conclusions can be made with a given study and which can not. Some statistical techniques, such as power analysis and sample effort curves, can be useful for defining the statistical error related to sampling and the error related to plant community variability. When plant community variability precludes hypothesis tests, other metrics like confidence intervals, should be considered for guiding management decisions (Krebs 1999).

### 5. Use functional groups that are ecologically informative and statistically meaningful.

Collecting vegetation cover data at the species level is important as it allows functional groups or guilds to be redefined throughout the life of a monitoring or research project. Depending on which hypotheses are being tested, the cover of a single species may be of interest to a project team. However, several species can often be lumped into functional groups to address study objectives relevant to general plant community change. For instance, defining a functional group of resprouting shrubs may be ecologically meaningful in a fire recovery study, while using a functional group that includes all shrub species may be more pertinent to a grazing effects study.

The feasibility of detecting statistically significant changes in various functional groups should also be considered when assigning species to those groups. Rhizomatous and bunch grasses were initially considered separately in the Tin Cup Fire Recovery Project; however, the absolute cover of each group was so low and the statistical variability was so high that there was very little power to detect significant differences. Combining both groups into one functional group of native perennial grasses yielded a guild for which detecting statistical differences was possible. High variability and low statistical power will likely affect species or functional groups with low absolute cover across the sagebrush steppe. A project team must therefore retain some flexibility in assigning ecologically meaningful functional groups to achieve statistically meaningful results.

### 6. Balance the amount of sampling necessary to achieve results with the resources available.

Unless unlimited project funding is available, the amount of data collected for a given research or monitoring project will often be less than ideal for most statistical models. The project team must therefore consider very carefully how sampling effort tradeoffs affect the experimental design. If, for example, the project team decides that higher MDCs are still ecologically meaningful, the number of plots planned for a project may be reduced. Additionally, using fewer functional groups, each containing more species, may be a useful approach to reducing sampling effort. Some approaches to reducing sampling effort, however, should not be considered if they compromise the quality of the data, in terms of the dispersion and precision needed for statistical hypothesis testing. Confining the distribution of the plots to a small portion of a burn, for instance, reduces a project team's ability to characterize the variability of the plant community across that burn and compromises the ability of the team to make statistically valid conclusions about the entire burn. In many cases, the cost of sampling a plot is quite minor when compared with other data collection and data analysis costs of a project, such as travel to the study site, setting up the plots, data entry, performing descriptive statistical summaries, etc. Thus, minimizing sampling effort within plots isn't usually the most effective means of reducing project costs though it is often the first cost-savings measure to be considered by a project team. Ultimately, the utility of a project is determined by the soundness of the experimental design, so cost-saving measures should never impair the overall integrity of the design.

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# SECTION 7: SUMMARY

***Roger D. Blew and Amy D. Forman***

A series of studies was conducted to investigate the vegetation changes that occur following fire in high quality sagebrush steppe rangeland in southeast Idaho. The studies had two primary objectives. The first was to describe post-fire trajectories in community composition and structure and the second was to examine the return of Wyoming big sagebrush onto burned areas though both natural processes and following artificial seeding.

The first objective was addressed using two studies. One addressed short-term (up to four years post-fire) community changes and the second addressed community changes over the long-term (up to 90 years post-fire).

The first study was conducted on the Tin Cup Fire that burned in 2000 on the Idaho National Laboratory (INL). The specific study goals were to understand whether the post-fire community had adequate natural recovery without seeding and whether fire followed by grazing exclusion resulted in an unacceptable trajectory. That study demonstrated that short-term response to fire is by resprouting of perennial grasses, forbs and shrubs. Similarly, exclusion from livestock grazing did not result in an undesirable plant community.

The study of long-term community change used a chronosequence of fires that burned in 2000, 1999, 1996, 1995, 1994, 1983, 1980, 1930s, and 1910. The specific goals of this study were to consider the role of fire in the overall dynamics of sagebrush steppe communities, examine how fire and recovery from fire function as transitions mechanisms between states and the potential for fire to result in transitions across a threshold to undesirable permanent states.

The results of this study showed little evidence for seral replacement in communities following fire. Differences in plant communities were related to more to local conditions (likely soils) than to time since fire. The only exception was the increase in big sagebrush with time since fire leading from communities dominated by native perennial grasses to communities co-dominated by sagebrush and native perennial grass. This study did not show evidence of transitions leading to undesirable states that might require management intervention. There was no evidence of cheatgrass or other annual species dominating a community following fire. The trend toward increasing big sagebrush with time since fire indicates no obvious threshold preventing its eventual return as a co-dominant.

These results indicate that native perennial grasses respond to fire through vegetative regrowth and not by recruitment. Improving condition for seed germination is likely not an important management consideration on healthy sagebrush steppe rangelands in the upper Snake River Plain. Managing for vigor of the perennial grasses should be the higher priority on burned areas. Areas with vigorous native perennial plant communities are less at risk to post-fire invasions and are less likely to require active restoration activities to establish a healthy plant community following fire. A healthy pre-fire plant community can increase the ability of a site to recover even under very adverse conditions such as severe drought, much like that experienced during our study period

The second major objective for this project was to examine the return of big sagebrush onto burned areas through both natural processes and following artificial seeding during the first ten years following fire. This objective was addressed in two studies.

The first of these was to measure the success of aerial seeding on a recent burn and on one that burned in 1994. The Tin Cup Fire which burned in 2000 was aerially seeded with Wyoming big sagebrush in February of 2001. In addition, a portion of a fire that burned in 1994 was seeded at the same time. The 1994 fire was studied to determine if sagebrush establishment could be improved by seeding onto burns on which the native perennial grasses, forbs and resprouting shrubs we already well established.

We surveyed a total of 24 transects each 1000 m (3208 ft) in length. There were six transects in each of four planting treatments: 1) burned in 2000 and planted in 2001, 2) burned in 2000 and not planted, 3) burned in 1994 and planted in 2001, 4) burned in 1994 and not planted.

Very few seedlings were found along the transects in the 2000 fire during surveys in 2002 and 2003. In the 1994 burn, fewer seedlings were found in the planted transects than in the transects in the area not planted. On these burns and with this planting, aerial planting had not been effective in accelerating big sagebrush recovery after fire.

The second study to look at sagebrush return following fire addressed the issue of sagebrush seed availability on the burned area as primary factor limiting recovery on large burns. Because sagebrush seed is not carried far by wind, the distribution of seed on the burn should decline with distance downwind from the upwind edge of the burn. If seed is the limiting factor, then the density of seedlings should show this same decline with distance from the burn edge.

We tested this by surveying plots for sagebrush seedlings. The plots started near the upwind edge and were spaced at 50 m intervals on transects extending 1000 m downwind. The surveys were conducted during 2004 in fires that burned in 1994, 1995, 1996, 1999, and 2000. None of these burned areas had been seeded.

The results of the surveys did not show a consistent, recognizable pattern of seedling distribution that could be attributed to seed dispersal by wind. The results suggest that factors other than seed dispersal are likely important for sagebrush recruitment and restoration strategies should focus on overcoming limitations to establishment in the seed bed rather than increasing seeding rate. Planting sagebrush seedlings rather than seeds circumvents limitations due to conditions in the seed bed and the critical establishment period during which many recently germinated seedlings die due to adverse environmental conditions.

Planting sagebrush seedlings in islands to disperse seeds downwind may not be the most effective management strategy. Planting individual seedlings or small groups of seedlings widely across the landscape to serve as colonization foci may be the most effective and most efficient use of resources for accelerating big sagebrush return to burned areas.

# Appendix A

# Review of the Sampling and Experimental Design of the Tin Cup Fire Recovery Project for Ability to Meet Certain Minimum Detection Levels