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The Idaho National Laboratory Site Long-Term Vegetation Transects: Understanding Change in Sagebrush Steppe

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January 2013



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The fourth chapter of this report addresses sagebrush population structure using data collected in 2006 with a field crew led by Kurt Jensen. Lance Kosberg and Rory O'Connor meticulously prepared sagebrush cross-sections and counted rings from nearly one thousand samples. The premise of investigating sagebrush stand condition and die-off in terms of population structure was conceptualized by numerous discussions between the authors and other scientists; Jay Anderson, Richard Inouye, and Sue Vilord, in particular.

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Executive Summary

The Long-Term Vegetation (LTV) Transects and associated permanent vegetation plots were established on what is now the Idaho National Laboratory (INL) Site in 1950. Vegetation abundance data, including density and cover, have been collected periodically once every two to ten years from plots located along two macro-transects, which are perpendicular to one another and intersect near the center of the INL Site. The data generated from the LTV Transects comprises one of the oldest, largest, and most comprehensive vegetation data sets for sagebrush steppe ecosystems in North America. LTV data are generally used to monitor vegetation condition and change in sagebrush steppe communities across the INL Site, while specific uses range from support for NEPA to conservation management planning. Eighty-nine plots are still active and were sampled in 2011, which represents the twelfth LTV sample period. Eleven plots were sampled again in 2012 because they burned just a few weeks subsequent to sampling in 2011, providing a unique opportunity to assess post-fire condition in terms of pre-fire vegetation composition. Additionally, results from 14 ancillary plots which were established and sampled in 2006 to address ongoing declines in big sagebrush, first detected in the LTV data, are included in this report as well.

Abiotic and biotic conditions across the entire region have been characterized by rapid change over the past decade. These changes include shifts in land cover, land use, and weather. Several large wildland fires have removed sagebrush from a large portion of the Upper Snake River Plain over the past ten to twenty years. On the INL Site specifically, nearly 60,000 ha have burned in the past five years alone. Soil disturbance associated with fighting wildland fires and soil disturbance associated with general increases in the use of remote backcountry areas are notable at the INL Site and across the region. Finally, some of the hottest and driest years during the 60-year weather record occurred during the past decade. All of these factors contribute to increasing stress on native plant communities and potentially set the stage for a period of dramatic change in vegetation composition across the region.

Some of the more important vegetation composition patterns resulting from incorporation of the 2011 LTV data into the long-term trend analyses were related to the abundance and distribution of non-native species. Crested wheatgrass abundance has continued to increase linearly since about 1990 and is of particular concern because it was not planted in the plots where it is now found, it continues to increase in the plots it occupies, and where present, it occurs with much greater mean abundance than comparable native, perennial bunchgrass species. Cheatgrass distribution increased between the 2006 and 2011 sample periods, and cheatgrass abundance, which historically fluctuates, increased significantly over the past five years. Introduced annual forbs, primarily desert alyssum, continued along a trajectory of exponential increase which began in the mid-1990s. Precipitation appears to be a poor indicator of non-native abundance. Native species trends were much easier to interpret within the context of precipitation and were within historical ranges of variation. Shrub cover from big sagebrush appears to be stabilizing, though it remains at the low end of its historical range.

In terms of composition, the abundance of non-native herbaceous species was nearly equivalent to that of native herbaceous species when averaged across the LTV plots. It does not appear as though increases in non-native species were at the expense of the native herbaceous understory, but these trends do mark a departure from ranges of historical herbaceous composition. Though native plant communities are still largely intact, their resistance and resilience may be tested over the next few decades.

The 2011 LTV surveys provided an opportunity to assess the relationship between pre-and post-fire vegetation condition when the T-17 Fire burned 11 of the LTV plots only a few weeks after they were sampled. The burned plots were surveyed again in 2012 to characterize vegetation abundance and composition during the first growing season after fire. Often, vegetation monitoring data are collected on burn scars to assess recovery, but pre-fire condition of the plant community is usually inferred or extrapolated from surrounding areas. Eventually, these data

will be used to develop a better understanding of how pre-fire condition affects post-fire recovery and to help identify indicators of potential post-fire recovery issues in the first few years after a wildland fire.

In earlier post-fire recovery studies at the INL Site, we have observed that the native species composition of post-fire communities generally reflect that of pre-fire communities, with exception of shrubs. Many of the changes identified in vegetation on the LTV plots after the T-17 Fire were consistent with those earlier observations. Despite the extremely dry conditions during the first growing season following the T-17 Fire, recovery of native perennial grasses, was noteworthy. Results from this limited data set indicate a striking post-fire decline in introduced annual species such as cheatgrass. These results suggest a different post-fire response of introduced annuals than may be otherwise expected. Introduced annual species are of particular concern in post-fire sagebrush steppe communities as they can, under some conditions, dominate entire regions. Post-fire decreases in the abundance of cheatgrass and desert alyssum on the LTV plots affected by the T-17 Fire were also counter to recent trends for these species at the INL Site based on the LTV data collected in 2011. Low cover values for all annual species in 2012 may be due in part to the below normal precipitation during the spring and summer period prior to the 2012 data collection. However, other recent research from the INL Site established patterns of post-fire decreases in cheatgrass abundance during a year with above normal precipitation.

Previous reports from the LTV data have demonstrated a decline in big sagebrush cover that is not associated with loss due to fire. Between 1975 and 2006 average big sagebrush cover on the unburned, core LTV plots declined from more than 20% to less than 10%. The cause of this trend is unknown, but earlier LTV reports suggested that it began as part of a widespread big sagebrush die-off in the mid- 1970's and has persisted, at least in part, because of a general lack of vigor. In order to better understand the losses of big sagebrush cover and the declines in stand condition at the INL Site, we initiated an investigation to 1) determine the typical range of stand age structure of mature big sagebrush stands, 2) investigate how stand age structure relates to shrub die-off and stand condition for big sagebrush, and 3) examine the dynamics of sagebrush stand replacement in the absence of wildland fire.

Based on our results, mechanisms controlling big sagebrush stand replacement appear to be related to a combination of general precipitation patterns and fine-scale microsite conditions. Generalized across the study site, annual recruitment patterns are cyclic and patterns in annual age class size reflect patterns in annual precipitation, but recruitment in some stands appears to be more affected by annual precipitation than in others. Big sagebrush stands are often managed by physical manipulation (fire, mowing, chaining, etc.) which are intended to reduce the abundance of older, "decadent" individuals and create space for younger, "healthier" shrubs. Our results suggest that disturbance is not required for stand replacement. In fact, all stands sampled for this project had a mean age of living individuals of less than 25 years and a mean age of individuals at death of less than 50 years indicating that natural rates of turnover at the INL Site are much higher than expected.

The LTV plots continue to provide an important data source for documenting and understanding the changes that are occurring in the sagebrush steppe ecosystem. There is no other comparable data set describing sagebrush steppe vegetation anywhere. These data have shown a broad scale decline in cover of big sagebrush not directly linked to broad scale disturbance like fire. The LTV data provide the only long-term documentation of invasion of crested wheatgrass into areas previously occupied by native sagebrush steppe plant communities in the absence of obvious disturbance. They provide the only opportunity to follow the dynamics of non-native annual, invasive species like cheatgrass and desert alyssum in native plant communities. Results from the most recently collected LTV data set suggest growing concern for the ability of sagebrush steppe plant communities to provide quality habitat for sagebrush-obligate wildlife species like Greater Sage-grouse, but also provide information that could guide management of those lands to prevent further decline.

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CHAPTER 1. INTRODUCTION: CHANGES IN FACTORS AFFECTING PLANT COMMUNITIES AND UPDATES TO VEGETATION SCIENCE AT THE INL SITE

1.1 The Long-Term Vegetation Transects

The Long-Term Vegetation (LTV) Transects and associated permanent vegetation plots (Figure 1-1) were established on what is now the Idaho National Laboratory (INL) Site in 1950 for the purpose of assessing the impacts of nuclear energy research and production on the surrounding ecosystem (Singley et al. 1951). Vegetation abundance data, including density and cover, have been collected periodically once every two to ten years from plots located along two macro-transects, which are perpendicular to one another and intersect near the center of the INL Site. The data generated from the LTV Transects comprises one of the oldest, largest, and most comprehensive vegetation data sets for sagebrush steppe ecosystems in North America. A detailed account of the history of the LTV Transects can be found in Forman et al. (2010).

Since their establishment, the LTV Transects have been used extensively to support the INL Site mission. They have been used for; supporting NEPA processes, making appropriate site-specific land management recommendations, and developing locally appropriate revegetation guidelines (e.g. Anderson and Shumar 1989, Blew et al. 2002). Data generated from the LTV Transects have also been the basis for major milestones in understanding practical and theoretical ecology of sagebrush steppe vegetation, which also benefits land stewardship efforts at the INL Site. Pertinent topics include; plant community classification and mapping, assessing effects of drought and livestock grazing, understanding fire history and recovery, characterizing species invasion patterns, testing theories of vegetation succession and change, and as a starting point for habitat suitability modeling for sensitive species. More recently, the LTV data have been used to support conservation management planning on the INL Site. Specifically, the LTV data were utilized, along with other vegetation data sets, to generate information about key ecological attributes and status indicators for four vegetation-related conservation targets in the Conservation Management Plan for the INL Site (Shurtliff et al. In Prep). Analyses of data from the LTV Transects have also been important for understanding conservation threats like habitat fragmentation, climate change, and the effects of stressors on the resilience of native plant communities.

Eighty-nine plots are still active and were sampled in 2011, which represents the twelfth LTV sample period. Eleven plots were sampled again in 2012 because they burned just a few weeks subsequent to sampling in 2011. Data from both 2011 and 2012 are considered part of the same sample effort and are summarized in this report.

1.2 Overview of the Local INL Site Environment

The INL Site is positioned at the northern extent of the Great Basin and is characterized by cold desert sagebrush steppe vegetation. Over the past sixty years, Plant communities on the INL Site have been classified into between eight and twenty-six distinct vegetation types (McBride et al. 1978, Anderson et al. 1996, Shive et al. 2011). Mean elevation of the INL Site is 1500 m. Surficial geology is strongly influenced by volcanic activity and soils include wind-blown 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. Shive et al. (2011) provides a more thorough description of the local environment.

Annual precipitation at the INL Site averages 210 mm (Figure 1-2), with May and June typically being the wettest months (Figure 1-3). Snow cover may persist from a few weeks to several months in the winter. Mean annual temperature for the INL Site is 5.6 °C; however, high diurnal and seasonal temperature fluctuations are normal (Anderson and Inouye 2001). Windy conditions occur frequently and wind direction is predominately from the southwest, but changes to the northeast for a few early morning hours daily.

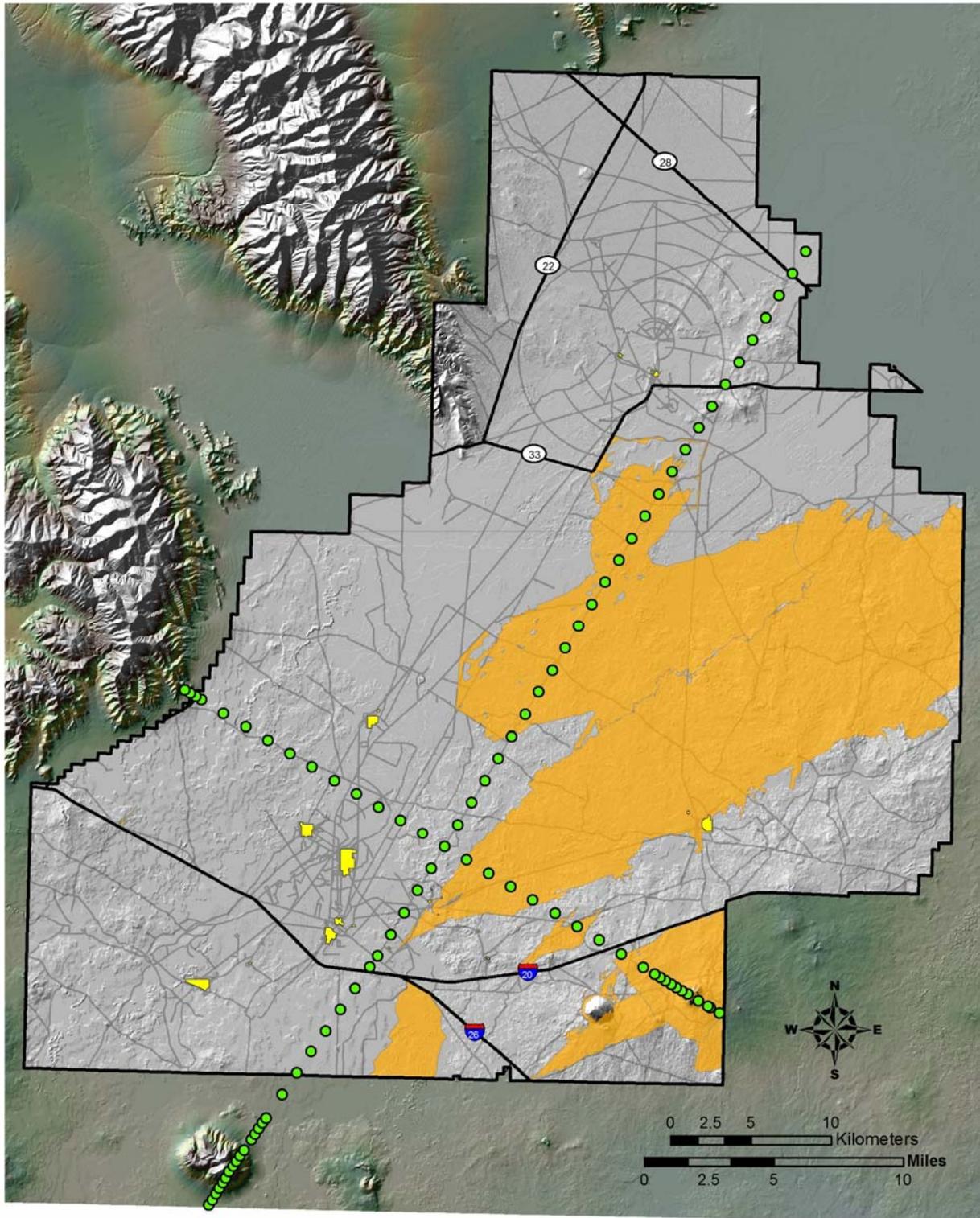


Figure 1-1. Long-Term Vegetation Plots on the Idaho National Laboratory Site. Wildland fires depicted in orange include only those which occurred after 2006.

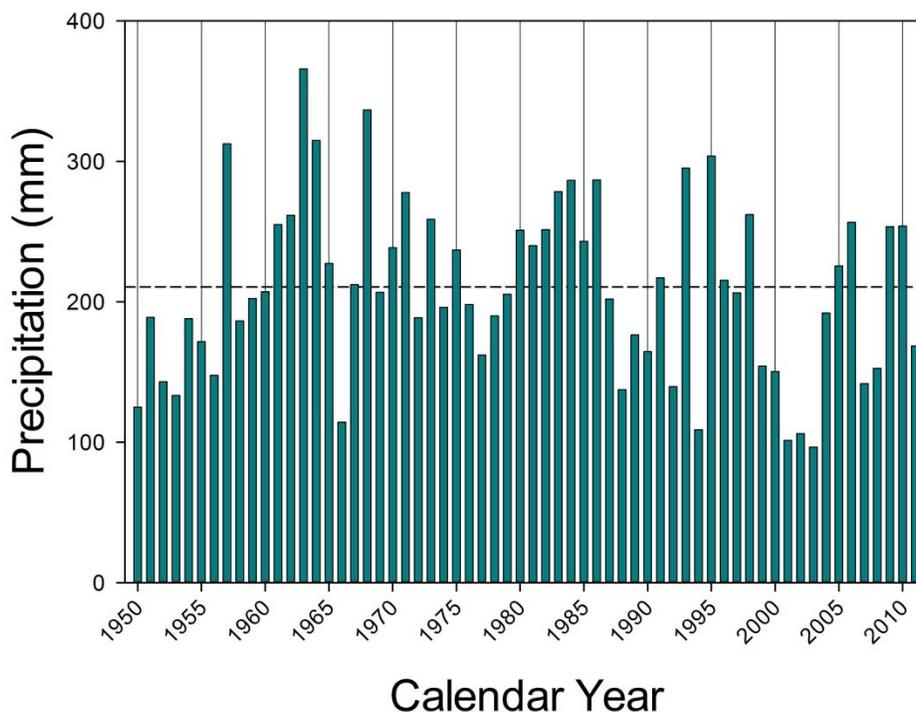


Figure 1-2. Total annual precipitation from 1950 through 2011 at the Central Facilities Area, Idaho National Laboratory Site. The dashed line represents mean annual precipitation.

1.3 Updates since the 2006 Sample Period

1.3.1 Summary of Recent Vegetation Monitoring and Applied Research

Two large-scale vegetation projects, independent of the LTV, were completed during the five years between the 2006 and 2011 sampled periods. The first project to be completed during this time period was a new plant community classification and vegetation map of the INL Site; the second addressed patterns of non-native species invasion and the factors driving plant biodiversity. Results from both projects increased our understanding of vegetation distribution and change at the INL Site, and both help inform interpretations of abundance and distribution trends apparent in the LTV data.

The plant community classification and vegetation map was completed in 2011 (Shive et al. 2011), and it represents the most thorough statistical classification of plant communities and the most detailed map of vegetation classes ever prepared for the INL Site. It also includes a comprehensive accuracy assessment, which hasn't been available with previous mapping efforts. Results of the plant community classification are consistent with other INL Site classification efforts (Anderson 1991), which conclude that most species' distributions are continuously variable in space and plant communities are typically only unique from one another in the relative abundance of regionally common species. Of the 89 active LTV plots, 70 are within the INL Site boundary and can be cross-walked with the most recent vegetation map (Table 1-1). The vegetation map is considered to be current through 2009; as of that time, 38 plots were in sagebrush shrublands (*Artemisia tridentata*), 29 were in green rabbitbrush (*Chrysothamnus viscidiflorus*) shrublands, two were in grasslands, and one was in a Utah juniper (*Juniperus osteosperma*) woodland. In terms of dominance by non-native species, seven were in vegetation classes defined as being dominated or co-dominated by crested wheatgrass (*Agropyron cristatum*) and two were in a class characterized by an abundance of desert alyssum (*Alyssum desertorum*).

Table 1-1. Cross-walk of the Shive et al. (2011) vegetation classes to the LTV plots. Wildland fires have burned through several plots since the vegetation map was completed, so many classes have likely changed.

Vegetation Class or Class Complex	Number of Plots
(2) Big Sagebrush Shrubland	19
(4b) Green Rabbitbrush/Bluebunch Wheatgrass Shrub Herbaceous Vegetation and (11ab) Bluebunch Wheatgrass - Sandberg Bluegrass Herbaceous Vegetation	12
(1/9) Green Rabbitbrush/Streambank Wheatgrass (Western Wheatgrass) Shrub Herbaceous Vegetation and (3) Needle and Thread Herbaceous Vegetation	5
(2) Big Sagebrush Shrubland and (4b) Green Rabbitbrush/Bluebunch Wheatgrass Shrub Herbaceous Vegetation	5
(7) Wyoming Big Sagebrush Shrubland	5
(2) Big Sagebrush Shrubland and (10) Crested Wheatgrass Semi-natural Herbaceous Vegetation	4
(4a) Green Rabbitbrush Shrubland and (1/9) Green Rabbitbrush/Streambank Wheatgrass (Western Wheatgrass) Shrub Herbaceous Vegetation	3
(4a) Green Rabbitbrush Shrubland and (12) Indian Ricegrass Herbaceous Vegetation	3
(7) Wyoming Big Sagebrush Shrubland and (1/9) Green Rabbitbrush/Streambank Wheatgrass (Western Wheatgrass) Shrub Herbaceous Vegetation	3
(10) Crested Wheatgrass Semi-natural Herbaceous Vegetation	2
(2) Big Sagebrush Shrubland and (4a) Green Rabbitbrush Shrubland	2
(4a) Green Rabbitbrush Shrubland and (4b) Green Rabbitbrush/Bluebunch Wheatgrass Shrub Herbaceous Vegetation	2
(11d) Utah Juniper Woodland and (11ab) Bluebunch Wheatgrass - Sandberg Bluegrass Herbaceous Vegetation	1
(4a) Green Rabbitbrush Shrubland	1
(5) Green Rabbitbrush - Winterfat Shrubland and (1/9) Green Rabbitbrush/Streambank Wheatgrass (Western Wheatgrass) Shrub Herbaceous Vegetation	1
(8) Green Rabbitbrush/Desert Alyssum Shrub Herbaceous Vegetation and (1/9) Green Rabbitbrush/Streambank Wheatgrass (Western Wheatgrass) Shrub Herbaceous Vegetation	1
(8) Green Rabbitbrush/Desert Alyssum Shrub Herbaceous Vegetation and (10) Crested Wheatgrass Semi-natural Herbaceous Vegetation	1

Concurrently, an investigation into non-native species invasion and native biodiversity was undertaken by Montana State University (Rew et al. 2012). Two important conclusions regarding non-native species abundance and distribution were made based on data collected to support this project. The first is that cheatgrass abundance tends to decrease in the years immediately post-fire, indicating that annual grass/fire cycle may be more complicated than

is reported elsewhere. The second is that crested wheatgrass invasion patterns indicate it may be considered one of a few truly invasive species on the INL Site. Similar, but much less detailed observations have been reported previously from analysis of the LTV data (Forman et al. 2010). Data from the biodiversity assessment portion of the investigation indicate that wildland fire does not considerably impact plant diversity and does not cause the same abrupt decline in vegetation condition as soil disturbance. Soil disturbance causes the greatest impact on native biodiversity at the INL Site followed by seasonality of weather-related factors, like precipitation. These results also substantiate patterns observed previously on the LTV plots (Colket and Bunting 2003) and from other vegetation data sets at the INL Site (Blew and Forman 2010).

1.3.2 Changes in Land Cover since 2006

Impacts caused by wildland fire and soil disturbance are abrupt and distinct enough in both time and space to be readily detectable over a period of five years. Other changes in vegetative land cover have also occurred during this time period, but tend to be more gradual in nature and difficult to assess over short time periods. Twelve wildland fires burned a total 58,992 ha on the INL Site between the 2006 and 2011 sample periods (Figure 1-1). Twelve plots were burned prior to the 2011 sample effort and two of those plots burned twice. An additional 11 plots burned within a few weeks of completing sampling in 2011. Of the 89 active LTV plots, a total of 40 have burned and several of those have burned more than once since 1994. Prior to 1994, the last fires known to affect LTV plots occurred at an unknown time before 1949 and before the plots were established.

Though the total spatial extent of soil disturbance since 2006 has not been quantified, it has certainly increased. The most extensive cause of soil disturbance over the past five years is from containment lines and fire breaks associated with firefighting efforts. Many core LTV plots are now within a few hundred meters of soil disturbance associated with firefighting. The core plots are 43 centrally located plots which are generally excluded from livestock grazing. While less widespread, there has also been soil disturbance related to INL Operations since 2006 (e.g. internal roads, explosives test range, gravel pit expansions, etc.)

1.3.3 Recent Precipitation Patterns

During the five years between the 2006 and 2011 sample periods, total annual precipitation was below average in three years (Figure 1-2). Total precipitation in 2011 was below average and cumulative precipitation through the completion of sampling in early August was well below average (Figure 1-3). Compared to 2006, the most recent

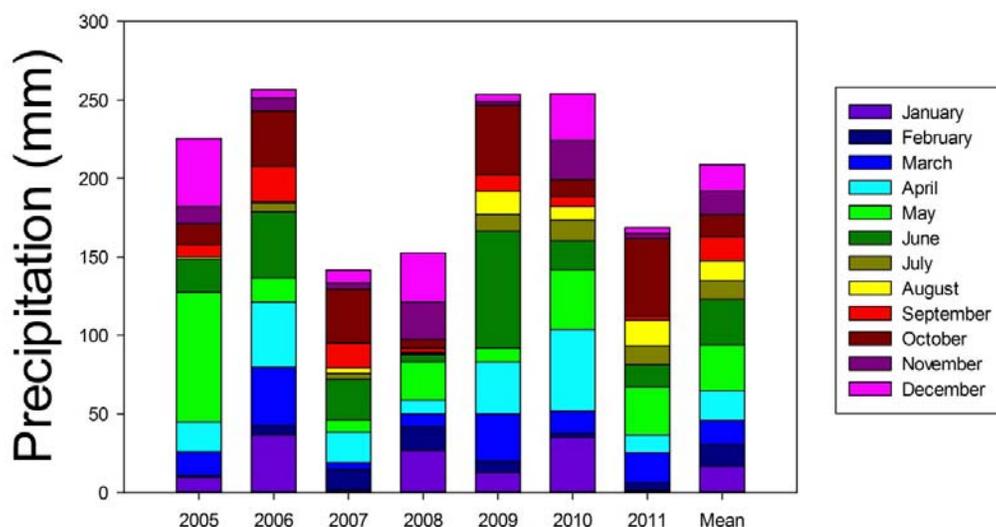


Figure 1-3. Annual precipitation by month from the Central Facilities Area, Idaho National Laboratory Site. Mean monthly precipitation includes data from 1950 through 2011.

previous sampling effort, 2011 was drier and had much lower spring precipitation as well (Figure 1-3). In addition, the seasonality of precipitation during the past five years appears to deviate from historical monthly means. Precipitation during late fall and early winter, October through January, tended to be much higher than average for one or more months in all years. In dry years like 2007 and 2008, nearly half of the total annual precipitation occurred during these winter months. Long-term monthly means indicate that only about one quarter of total annual precipitation historically fell during this fall/winter time period (Figure 1-3).

1.4 Objectives Associated with the 2011 Sample Period

Sampling for the twelfth LTV data set occurred between early June and mid-August of 2011. Data were collected according to the protocols outlined in 2006, which formalized data collection procedures and processes used on the LTV Plot since 1950 (Forman et al. 2010). Field data were processed daily and were integrated into a 2011 project database, where they were subject to a number of Quality Assurance/Quality Control routines. Upon final verification/validation of the 2011 project database, all 2011 data were migrated into the comprehensive LTV project database. Forman et al. (2010) provide a description of the LTV database and its substructures. Data were collected again on 11 plots in 2012 and were subject to the same data process. This report documents the current LTV sample period and is organized into four chapters. This chapter provides a brief overview of the LTV project and summarizes changes since the previous LTV sample period.

The second chapter is an update of long-term trend analyses for major vegetation functional groups across the INL Site. It includes information about native shrub, grass, and perennial forb abundance, as well as distribution and abundance patterns of non-native species. The results of analyses presented in this chapter provide an indication of general vegetation condition on the INL Site.

Very little empirical information is available on the relationship between pre-fire condition and post-fire recovery of plant communities in the sagebrush steppe. Often, vegetation monitoring data are collected on burn scars to assess recovery, but pre-fire condition of the plant community is usually inferred or extrapolated from surrounding areas. The T-17 fire burned 11 LTV plots just a few weeks after data collection had been completed in 2011, providing a unique opportunity to monitor post-fire recovery on well-characterized sites. We resampled the 11 burned plots in 2012, and Chapter 3 compares vegetation abundance and composition immediately pre- and post-fire. Eventually, these data will be used to develop a better understanding of how pre-fire condition affects post-fire recovery and to help identify indicators of potential post-fire recovery issues in the first few years after a wildland fire.

Chapter 4 includes INL-specific data and discussion about big sagebrush population biology and its effects on sagebrush steppe plant communities. In the late 1990's and early 2000's it became evident in the LTV data that big sagebrush was undergoing a dramatic and prolonged period of decline. Though researchers had speculated about its cause (Anderson 1986, Colket and Bunting 2003), we lacked sufficient information about the population biology of the species to understand the processes involved. The nature of big sagebrush decline on the INL Site coupled with increasing conservation pressures for sagebrush-obligate species, made obvious the need for an investigation into big sagebrush ecology. Because destructive sampling on or around the LTV plots would render them useless for future trend analyses, 14 independent plots were established in the center of the INL Site for the purposes of characterizing big sagebrush population dynamics. Results of the big sagebrush population study, as well as the implications of these results on conservation measures and land management strategies at the INL Site are also included in this chapter.

1.5 Literature Cited

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CHAPTER 2. LONG-TERM TRENDS: DISENTANGLING MEANINGFUL CHANGE FROM STOCHASTIC NOISE

2.1 Introduction

Although the Long-Term Vegetation (LTV) Transect plots have had limited human impact since the 1950s (Forman et al. 2010), they are located in increasingly close proximity to disturbance. They are also experiencing more variability in seasonal weather patterns, as noted in the previous chapter. The LTV plots are becoming more important for understanding changes in native sagebrush steppe under increasing environmental stress. The LTV plots are particularly pertinent for monitoring change in vegetation condition, mechanisms driving local changes in species abundance and distribution, and understanding resistance and resilience inherent to local native plant communities.

Over the past few decades, the local Idaho National Laboratory (INL) Site and greater regional landscape have undergone some dramatic changes. Wildland fires have affected plant communities across about 38% of the INL Site since 1994. Several large fires have altered much of the sagebrush steppe vegetation to the south and to the east of the INL Site during the same time period. As noted in Chapter 1, the total extent of disturbed soil outside of facility footprints at the INL Site is greater now than it was a few decades ago and programmatic use of remote areas beyond facilities fences has also increased. Impacts from livestock operations in grazing allotments, such as vegetation damage due to off-road travel, have also been noted to be increasing in recent years. The increase in anthropogenic disturbance on the INL Site is consistent with trends in land-use across the western U.S. As direct disturbances on plant communities within the INL Site boundaries increase and as disturbances across the region influence vegetation composition on the INL Site, plant communities will likely be affected as well.

The LTV Plots were sampled for the twelfth time in 2011. Analysis of the data collected during the 2011 sample period entails updating functional group trends which have been reported for several decades (Forman et al. 2010). Trend analysis of non-native species abundance and distribution, first conducted in association with the 2006 sample effort, will also be updated with the most current data set and discussed here.

2.2 Methods

All 89 active LTV plots were sampled between early June and mid-August of 2011. The plots are located along two macro-transects which bisect the INL Site. Plots are located approximately 1 mile apart, except on steep slopes, where they are in closer proximity to one another to capture the environmental gradient associated with changes in elevation. Forty-three plots, which are located in the centermost part of the INL Site, have been defined as “core plots” and have generally been considered to be representative of good-condition sagebrush steppe plant communities. The remainder of the plots are in vegetation types typically associated with sagebrush steppe, but communities are much more variable in the “peripheral plots” due to elevation, livestock grazing management, topography, soils, etc. For a more thorough description of the LTV plots, see Forman et al. (2010). Shive et al. (2011) and Forman et al. (2010) provide good discussions of the study site in terms of local biotic and environmental conditions.

The original LTV plots consisted of two, 15.24-m transects that were placed parallel to one another and about 4.5 m apart. An additional transect, 20 m in length, was added to many of the plots in 1985; it was placed parallel to the original transects and about 4.5 m from the second transect. Cover, density, and frequency were measured on all of the accessible LTV plots in 1950, 1957, 1965, 1975, 1985, 1995, 2001, 2006, and most recently, 2011. A subset of plots and/or abundance metrics were sampled in 1978, 1983, and 1990.

Density and frequency were measured using twenty, 0.3 x 1.0 m quadrats located along the original two transects of each plot. Cover was estimated using line interception in all study years and point interception was used on most of the plots during the 1985 sample year and all sample periods thereafter. Line intercept data were collected along both original transects and point intercept data were collected using 50 point frames placed along all three transects.

Density and frequency data and point interception data were collected for all taxa, while line interception data were collected for only perennial grasses and shrubs. Line interception measurements follow guidelines suggested by Canfield (1941) and point interception data were collected using a method developed by Floyd and Anderson (1982). See Forman et al. (2010) for detailed sampling methodology.

Data were analyzed using regressions and one- and two-way ANOVAs (Zar 1999). Although a repeated measures design is the most appropriate statistical model given the permanent nature of the LTV plots, we were only able to use repeated measures ANOVAs for some of the analyses. Repeated measures tests are sensitive to balanced experimental designs, and in many cases data missing from the historical archives and inconsistencies in which plots were sampled from one sample period to the next led to unbalanced designs that were not amenable to repeated measures tests.

Updates to analyses characterizing trends in native species abundance and community composition were addressed using both point- and line-interception cover data on the core plots. An updated cover by species table for 2011 was compiled using the point-interception data, as it is the only cover metric that samples all taxa on a common scale. Long-term trends in functional groups of perennial species were characterized using line-interception data. Cover data for annual species were not collected prior to the initiation of point-interception methods in 1985, precluding cover analyses of those functional groups for the entire study period (1950-2011). The consistency with which line intercept data were collected in the core plots allowed us to analyze long-term trends of the perennial functional groups using repeated measures ANOVA's. Significance was determined at the $\alpha = 0.05$ level and the Holm-Sidak method (Sidak 1967) was used for multiple comparisons.

Analyses of non-native annual species' distribution and abundance patterns, from 1950 through the current study period, were conducted using density/frequency data on all of the LTV plots that were sampled during each of nine sample periods in which all available plots were sampled. The three sample periods during which only the core plots were sampled were omitted from the non-native species analyses because we were interested in invasion patterns over as large a spatial scale as possible. The distribution of plots in which non-native species and functional groups occurred was mapped for the two most recent sample periods. We further analyzed changes in the density and frequency of annual species over the nine sample periods using one-way ANOVAs. Repeated measures designs could not be used due to inconsistencies among years in terms of which plots were sampled and represented in the historical data archives. The Holm-Sidak method was again used for multiple comparisons and significance was determined at the $\alpha = 0.05$ level.

2.3 Results and Discussion

2.3.1 Cover by Species – Changes Since 2006

Point interception data from 2011 were used to summarize cover by species across the 43 core plots (Table 2-1). Compared with the 2006 data (Table 2-2), absolute total shrub cover increased slightly. Big sagebrush (*Artemisia tridentata*) cover was about 2% higher in 2011 and green rabbitbrush (*Chrysothamnus viscidiflorus*) cover was nearly 1% higher. These results are somewhat unexpected because five of the core plots burned between the 2006 and 2011 sample periods and three of those five plots were in a big sagebrush shrubland vegetation class prior to the fire. Because big sagebrush cover likely decreased precipitously in at least three plots between the two sample periods, the overall increase in cover when averaged across the core plots is notable.

Total cover by perennial graminoids was about 2% lower in 2011 than in 2006. Needle and thread (*Hesperostipa comata*) was the most abundant perennial graminoid in both sample periods, but mean absolute cover of this species declined by a little more than 1% over five years (Table 2-1 and Table 2-2). In 2011, crested wheatgrass (*Agropyron desertorum*), an introduced species, was the third most abundant perennial grass across the core LTV plots.

Table 2-1. Mean percent cover of vascular plants sampled on 43 Long-Term Vegetation Transect core plots using point-intercept methods during the 2011 sample period. Species are listed in order of descending cover values within each functional group. Cover is reported for each species having an absolute cover value > 0.1%. Constancy indicates the number of plots in which a species occurred and cover normalized by constancy indicates the mean cover of a species averaged across only the number of plots in which it occurred.

Plant Species	Absolute Cover (%)	Constancy	Cover (%) Normalized by Constancy
Shrubs			
<i>Artemisia tridentata</i>	9.55	33	12.45
<i>Chrysothamnus viscidiflorus</i>	7.88	39	8.68
<i>Linanthus pungens</i>	0.67	16	1.80
<i>Grayia spinosa</i>	0.60	6	4.28
<i>Krascheninnikovia lanata</i>	0.40	6	2.85
<i>Tetradymia canescens</i>	0.17	7	1.05
Others (n = 4)	0.22		
Total Shrub Cover	19.48		
Perennial Graminoids			
<i>Hesperostipa comata</i>	2.06	28	3.16
<i>Elymus lanceolatus</i>	1.58	31	2.20
<i>Agropyron desertorum</i>	1.13	6	8.08
<i>Achnatherum hymenoides</i>	0.45	29	0.66
<i>Poa secunda</i>	0.43	24	0.77
<i>Pascopyrum smithii</i>	0.31	5	2.63
<i>Elymus elymoides</i>	0.23	24	0.41
<i>Pseudoroegneria spicata</i>	0.15	5	1.32
<i>Aristida purpurea</i>	0.13	2	2.77
Others (n = 3)	0.07		
Total Perennial Graminoid Cover	6.53		
Perennial Forbs			
<i>Crepis acuminata</i>	0.34	11	1.32
<i>Phlox hoodii</i>	0.32	25	0.55
<i>Astragalus filipes</i>	0.22	13	0.71
<i>Eriogonum ovalifolium</i>	0.21	12	0.76
<i>Comandra umbellata</i>	0.21	4	2.24
<i>Erigeron pumilus</i>	0.14	13	0.47
Others (n = 29)	0.93		
Total Perennial Forb Cover	2.36		
Succulents			
<i>Opuntia polyacantha</i>	0.17	23	0.32
Native Annuals and Biennials			
<i>Gayophytum diffusum</i>	0.22	11	0.85
<i>Cryptantha scoparia</i>	0.18	12	0.65
<i>Descurainia pinnata</i>	0.14	13	0.47
Others (n = 11)	0.29		
Total Native Annual/Biennial Cover	0.84		
Introduced Annuals and Biennials			
<i>Alyssum desertorum</i>	7.37	27	11.73
<i>Bromus tectorum</i>	4.61	30	6.61
<i>Halogeton glomeratus</i>	0.63	2	13.52
<i>Descurainia sophia</i>	0.11	12	0.41
<i>Sisymbrium altissimum</i>	0.10	5	0.88
Others (n = 2)	0.01		
Total Introduced Annual/Biennial Cover	12.83		
Total Vascular Plant Cover	42.21		

Table 2-2. Mean percent cover of vascular plants sampled on 43 Long-Term Vegetation Transect core plots using point-intercept methods during the 2006 sample period. Species are listed in order of descending cover values within each functional group. Cover is reported for each species having an absolute cover value > 0.1%. Constancy indicates the number of plots in which a species occurred and cover normalized by constancy indicates the mean cover of a species averaged across only the number of plots in which it occurred.

Plant Species	Absolute Cover (%)	Constancy	Cover (%) Normalized by Constancy
Shrubs			
<i>Chrysothamnus viscidiflorus</i>	6.90	41	7.24
<i>Artemisia tridentata</i> ssp. <i>wyomingensis</i>	4.89	30	7.00
<i>Artemisia tridentata</i> ssp. <i>tridentata</i>	2.50	11	9.75
<i>Grayia spinosa</i>	1.00	9	4.77
<i>Linanthus pungens</i>	0.81	18	1.93
<i>Tetradymia canescens</i>	0.42	7	2.55
<i>Krascheninnikovia lanata</i>	0.37	5	3.14
<i>Ericameria nauseosa</i>	0.10	5	0.87
Others (n = 5)	0.13		
Total Shrub Cover	17.10		
Perennial Graminoids			
<i>Hesperostipa comata</i>	3.24	31	4.49
<i>Achnatherum hymenoides</i>	1.16	38	1.31
<i>Elymus lanceolatus</i>	1.00	31	1.39
<i>Agropyron desertorum</i>	0.95	6	6.82
<i>Elymus elymoides</i>	0.94	29	1.39
<i>Pascopyrum smithii</i>	0.82	10	3.54
<i>Poa secunda</i>	0.45	28	0.69
Others (n = 6)	0.20		
Total Graminoid Cover	8.76		
Perennial Forbs			
<i>Schoenocrambe linifolia</i>	0.78	32	1.04
<i>Phlox hoodii</i>	0.45	26	0.75
<i>Eriogonum ovalifolium</i>	0.26	16	0.71
<i>Astragalus lentiginosus</i>	0.19	15	0.54
<i>Stephanomeria spinosa</i>	0.12	5	1.02
Others (n = 24)	0.84		
Total Perennial Forb Cover	2.64		
Succulents			
<i>Opuntia polyacantha</i>	0.40	33	0.52
Native Annuals and Biennials			
<i>Lappula occidentalis</i>	0.36	24	0.64
<i>Eriogonum cernuum</i>	0.25	16	0.67
<i>Gayophytum diffusum</i>	0.23	16	0.62
<i>Eriastrum wilcoxii</i>	0.17	19	0.38
<i>Cordylanthus ramosus</i>	0.12	6	0.87
Others (n = 13)	0.38		
Total Native Annual/Biennial Cover	1.50		
Introduced Annuals and Biennials			
<i>Alyssum desertorum</i>	2.67	26	4.42
<i>Bromus tectorum</i>	0.77	27	1.23
<i>Salsola kali</i>	0.53	10	2.26
<i>Descurainia sophia</i>	0.14	11	0.55
<i>Sisymbrium altissimum</i>	0.14	7	0.84
<i>Halogeton glomeratus</i>	0.12	6	0.87
<i>Tragopogon dubius</i>	0.01	3	0.09
Total Introduced Annual/Biennial Cover	4.37		
Total Vascular Plant Cover	34.77		

However, it only occurred in six of the 43 plots. Cover normalized by constancy values for this species indicate that it is very abundant in the plots where it occurs, much more so than any single native species (Table 2-1). Increases in crested wheatgrass are of particular concern because it has invaded the plots where it is found (i.e. it wasn't intentionally planted), it continues to increase in the plots it occupies, and where it is present it occurs with much greater mean abundance than comparable native, perennial bunchgrass species.

Total perennial forb cover was similar between the 2006 and 2011 sample periods. All species with more than 0.1% absolute cover, when averaged across the core plots, were native. The species composition of perennial forbs was highly variable from plot to plot and when averaged across the core plots, the complement of species with greater than 0.1% cover changed considerably over the five years between sample efforts (Table 2-1 and Table 2-2). In fact only two species with more than 0.1% mean cover in 2006 were present with similar cover values in 2011; those species are Hood's phlox (*Phlox hoodii*) and cushion buckwheat (*Eriogonum ovalifolium*). Native forbs continue to be diverse and variable, with the potential to respond to changing environmental conditions rapidly from one year to the next.

Native annuals and biennials were about half as abundant in 2011 when compared to 2006 values, while introduced annuals and biennials were nearly three times more abundant in 2011 than in 2006. Desert alysium (*Alyssum desertorum*) was the most abundant introduced annual in 2011, when averaged across the core LTV plots. Cheatgrass (*Bromus tectorum*) was the second most abundant introduced annual, with a mean absolute cover of just under 5%. For the 2011 sample period cheatgrass was more abundant than any single perennial grass species, but as a functional group introduced annual grasses were not more abundant than total cover of perennial grasses.

Total annual precipitation in 2006 was about 40 mm above average while total precipitation for 2011 was about 40 mm below average (Chapter 1, Figure 1-3). Thus, cover of native annuals and biennials during the sample periods associated with those years was reflective of gross precipitation patterns. Cover of introduced annuals and biennials aren't as easily explained by total precipitation values, though the relatively wet November and December prior to the 2011 sample effort may have favored introduced annuals and the wet spring of 2006 may have increased the abundance of native annuals and perennials during the growing season of that sample period. Temperatures probably also contributed to the abundance patterns of native and introduced annuals from the 2006 and 2011 sample periods as well.

In summary, total vascular vegetation cover was almost 8% higher in 2011 than in 2006 despite the drier environmental conditions which prevailed before the more recent sample period. Nearly all of the increase in absolute total cover was from introduced annual grasses and forbs. Mean cover for crested wheatgrass and cheatgrass combined was roughly equivalent to total cover from native grasses and more than half of the total cover from herbaceous species across the core plots was from introduced grasses and forbs. Similar cover by species tables, summarized from the point interception data, were first presented using 1995 data (Anderson and Inouye 1999). Although the relative cover by introduced species decreased between the 1995 and 2006 sample periods, it was much higher during the 2011 growing season than in either 2006 or 1995, and 2011 is the first sample period during which introduced herbaceous species were as abundant as native herbaceous species.

2.3.2 Cover by Functional Group – Trends Since 1950

Line interception data were used to update functional group trend analyses across the core LTV plots, which include all sample periods since 1950. Perennial grass and shrub cover from 2011 were both within the historical range of variability for the 61 year-old data set (Figure 2-1). Mean shrub cover in 2011 wasn't significantly higher or lower than highest or lowest means from previous sample periods (Appendix A, Tables A-1a and A-1b). Mean perennial grass cover was significantly higher in 2011 than in the sample years with the four lowest means, and it was significantly lower than in 1975, the year with the highest reported mean perennial grass cover (Appendix A, Tables A-1a and A-1b). Generally, shrub cover was up slightly and perennial grass cover was down slightly from the previous sample period, but neither difference was significant. The trends for mean shrub and perennial grass cover

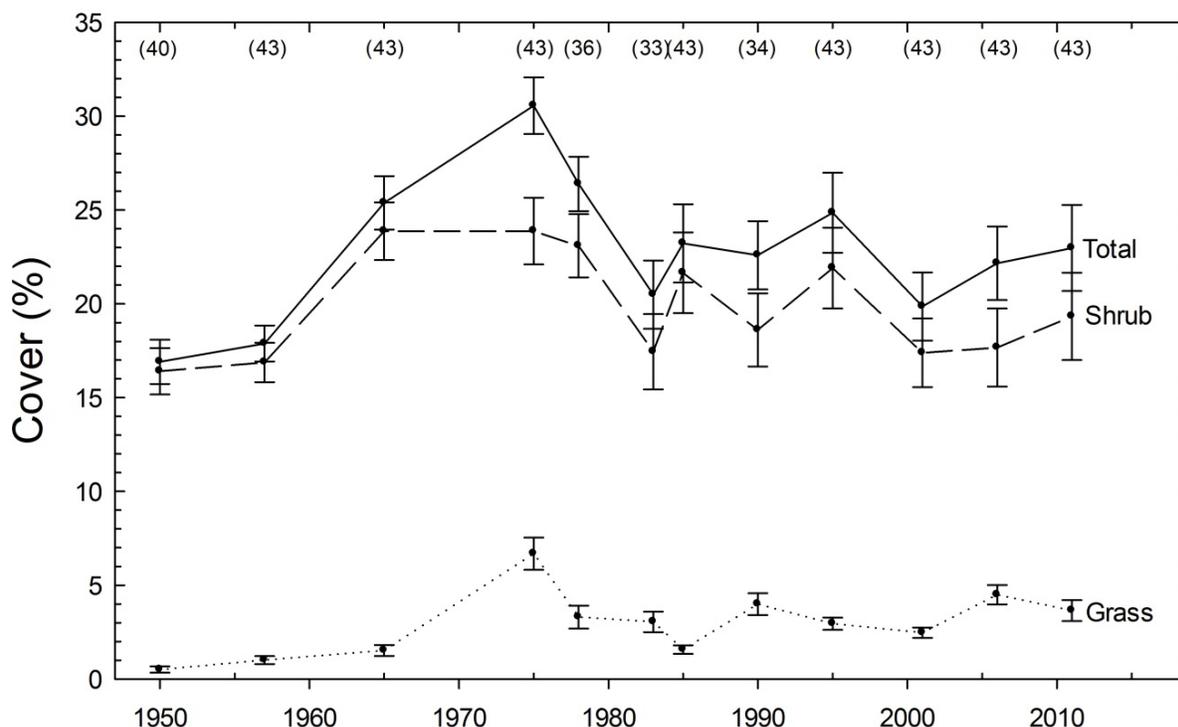


Figure 2-1. Trends in shrub cover, native perennial grass cover, and total combined perennial grass and shrub cover from 1950 to 2011 for the core subset of plots on the Long-Term Vegetation Transects at the Idaho National Laboratory Site. Data were collected using line-interception methods and are represented here as means \pm 1 SE. Numbers in parentheses at the top of the frame indicate the number of plots for which data were available in each sample year.

between the 2006 and 2011 sample periods were consistent between the point intercept (Tables 2-1 and 2-2) and the line intercept data sets (Figure 2-1).

Within the shrub functional group, mean green rabbitbrush cover values for 2011 remained similar to those reported for the 2006 sample period. Green rabbitbrush cover values have generally been trending upward over the 61 year-old data set (Figure 2-2) and mean cover for the two most recent sample periods was significantly higher than mean cover for the two earliest sample periods (Appendix A, Tables A-2a and A-2b). Big sagebrush cover increased slightly between the 2006 and 2011 sample periods (Figure 2-2). The increase was not statistically significant (Appendix A, Tables A-2a and A-2b), however, 2011 represents one of the few sample periods since 1965 for which big sagebrush cover did not decline over the mean from the previous sample period. Mean big sagebrush cover for 2011 would have been higher than is reported had three sagebrush-dominated core plots not burned between the 2006 and 2011 sample periods, as noted in the previous section. Spring and summer precipitation were well above average in 2008 and 2009 (Chapter 1, Figure 1-3) which may have improved big sagebrush recruitment in those years (see Chapter 4 for discussion) and resulted in slightly higher cover values in 2011. Although mean big sagebrush cover appears to have at least stabilized somewhat from 2001 through 2011, it remains significantly lower than it was from 1950-1978 and in 1985 (Appendix A, Tables A-2a and A-2b). For the third sample period in a row, mean green rabbitbrush cover and mean big sagebrush cover across the core plots were statistically indistinguishable from one another (Appendix A, Tables A-2a and A-2b).

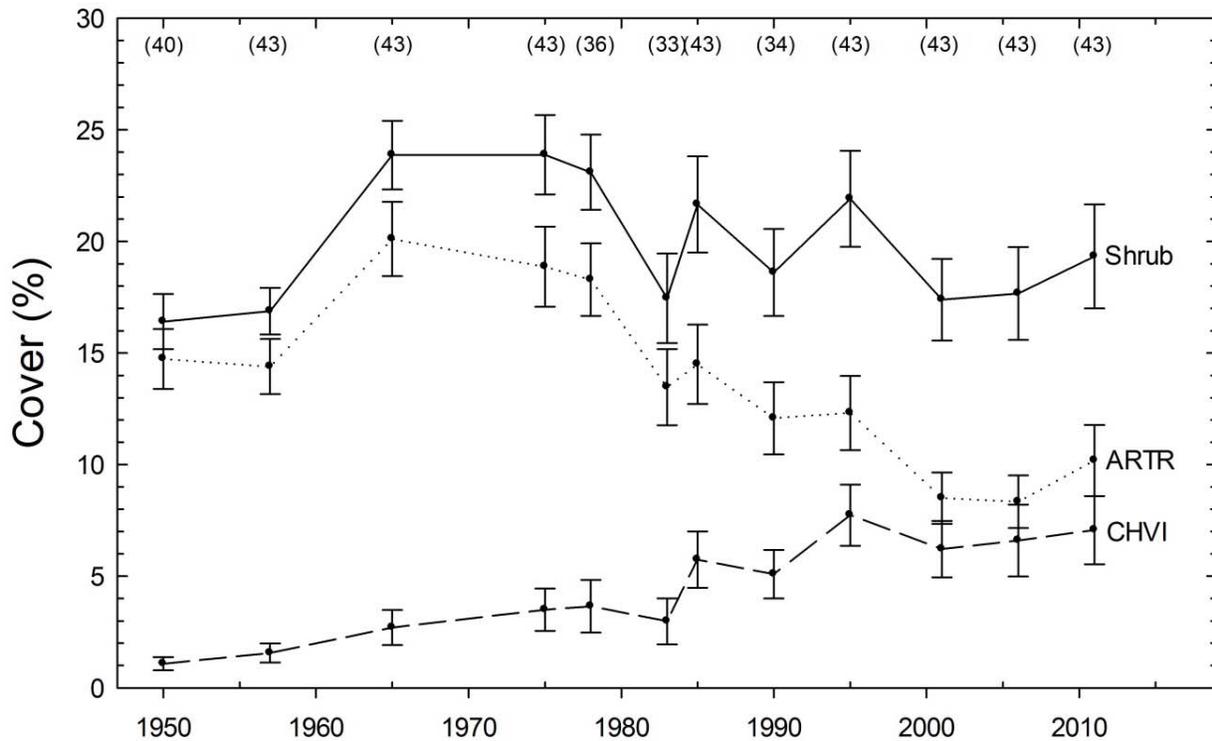


Figure 2-2. Trends in total shrub cover, *Artemisia tridentata* cover, and *Chrysothamnus viscidiflorus* cover from 1950 to 2011 for the core subset of plots on the Long-Term Vegetation Transects at the Idaho National Laboratory Site. Data were collected using line-interception methods and are represented here as means \pm 1 SE. Numbers in parentheses at the top of the frame indicate the number of plots for which data were available in each sample year.

Absolute cover of native grasses was near the middle of the historic range of variability in 2011, with a mean of about 4% (Figure 2-3). Total grass biomass is typically underestimated by this cover value, as cover is sampled basally. Bunchgrasses decreased and rhizomatous grasses increased slightly since the last sample period (Figure 2-3), but neither change was statistically significant (Appendix A, Tables A-3a and A-3b). Compared to historical values, mean rhizomatous grass cover in 2011 was not significantly different from the highest or lowest reported values (Appendix A, Tables A-3a and A-3b). Bunchgrass cover was significantly higher in 2011 than in was in the sample periods with the five lowest mean values, which occurred prior to the mid-1960s and during the mid-1980s (Figure 2-3). During the 2011 sample period, mean bunchgrass cover was significantly higher than mean rhizomatous grass cover, a trend which has been consistent since 1990 (Appendix A, Tables A-3a and A-3b).

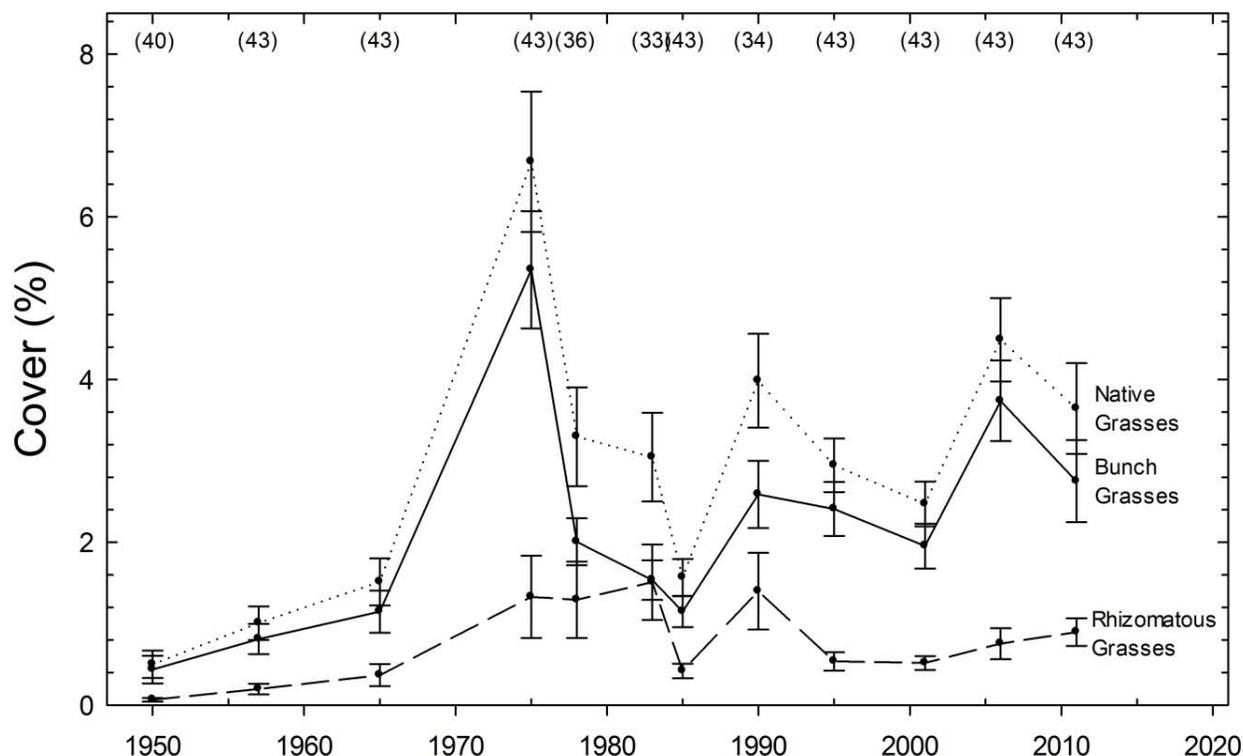


Figure 2-3. Trends in total native perennial grass cover, native bunchgrass cover, and native perennial rhizomatous grass cover from 1950 to 2011 for the core subset of plots on the Long-Term Vegetation Transects at the Idaho National Laboratory Site. Data were collected using line-interception methods and are represented here as means \pm 1 SE. Numbers in parentheses at the top of the frame indicate the number of plots for which data were available in each sample year.

Mean cover of native perennial grasses decreased between the 2006 and 2011 sample periods, as would be expected from precipitation trends during the corresponding time periods (Figure 2-4). Precipitation during the spring prior to the 2006 sample effort was much higher than average, while precipitation during the spring prior to the 2011 sample period was slightly below average (Chapter 1, Figure 1-3). The mean cover of introduced perennial grasses, entirely from crested wheatgrass, did not track precipitation as predictably between the two most recent sample periods (Figure 2-4). In fact, crested wheatgrass doesn't appear to track precipitation at all. Unlike native grass cover, which fluctuates from one sample period to the next, crested wheatgrass cover has been steadily increasing, in a very linear fashion since about 1990 (Figure 2-4). As mentioned in the previous section, this trend is driven by only six of the 43 core plots in which crested wheatgrass currently occurs and it wasn't planted in any of those plots. Thus, the risk of crested wheatgrass introduction to a local area via seed dispersal is high and subsequent linear increases in abundance appear to be highly predictable but irreversible under current conditions. Although native perennial grass cover is still significantly greater than introduced perennial grass cover (Appendix A, Tables A-4a and A-4b) when averaged across the entire core LTV plots, the introduction of crested wheatgrass into just a handful of additional plots could eliminate that difference within about a decade at the current rate of increase.

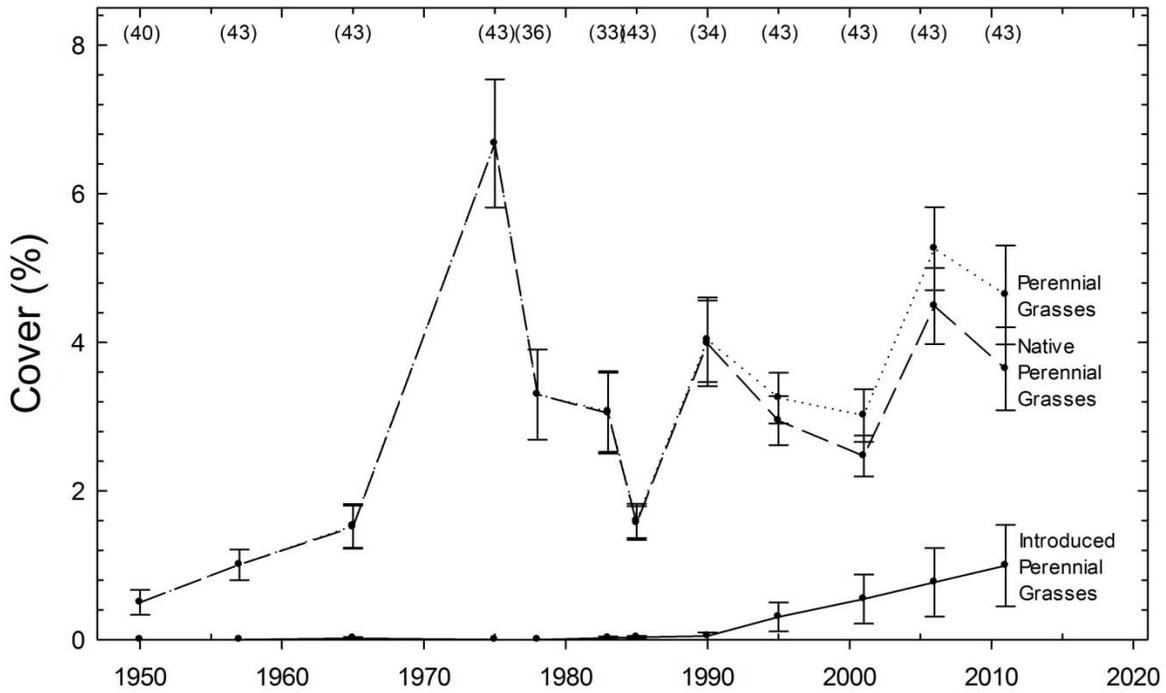


Figure 2-4. Trends in total perennial grass cover, native perennial grass cover, and introduced perennial grass cover from 1950 to 2011 for the core subset of plots on the Long-Term Vegetation Transects at the Idaho National Laboratory Site. Data were collected using line-interception methods and are represented here as means \pm 1 SE. Numbers in parentheses at the top of the frame indicate the number of plots for which data were available in each sample year.

2.3.3 Annual Species Abundance – Trends Since 1950

Trend analyses of abundance and distribution of annual species were first conducted subsequent to the 2006 sample period (Forman et al. 2010) and are updated here with the 2011 data. Density/frequency data are used for long-term analyses of annual species because it is the only dataset available with annual species abundance prior to 1985. During the 2011 sample period, cheatgrass was detectable in 56 out of 89 active LTV plots (Figure 2-5a), which is an increase over the 2006 sample period when cheatgrass was detectable in a total of 48 plots (Figure 2-5b). On a plot by plot basis, increases were not necessarily directional or linear. Plots 79 and 85, which were tracked individually during previous analyses, and for which cheatgrass cover was below detectable levels in 2006, both experienced density increases of varying magnitudes by 2011. Conversely, between the 2006 and 2011 sample periods, cheatgrass abundance decreased to undetectable levels in Plots 23 and 24 (Figures 2-5a and 2-5b). As noted from previous reports (Forman et al. 2010), the overall macro-scale distribution of cheatgrass continues to trend upward, but the pattern of increase is remarkably sporadic from plot to plot and year to year. At the scale of an individual plot, it remains difficult to predict whether density will increase or decrease from one sample period to the next based solely on historical trends in cheatgrass abundance.

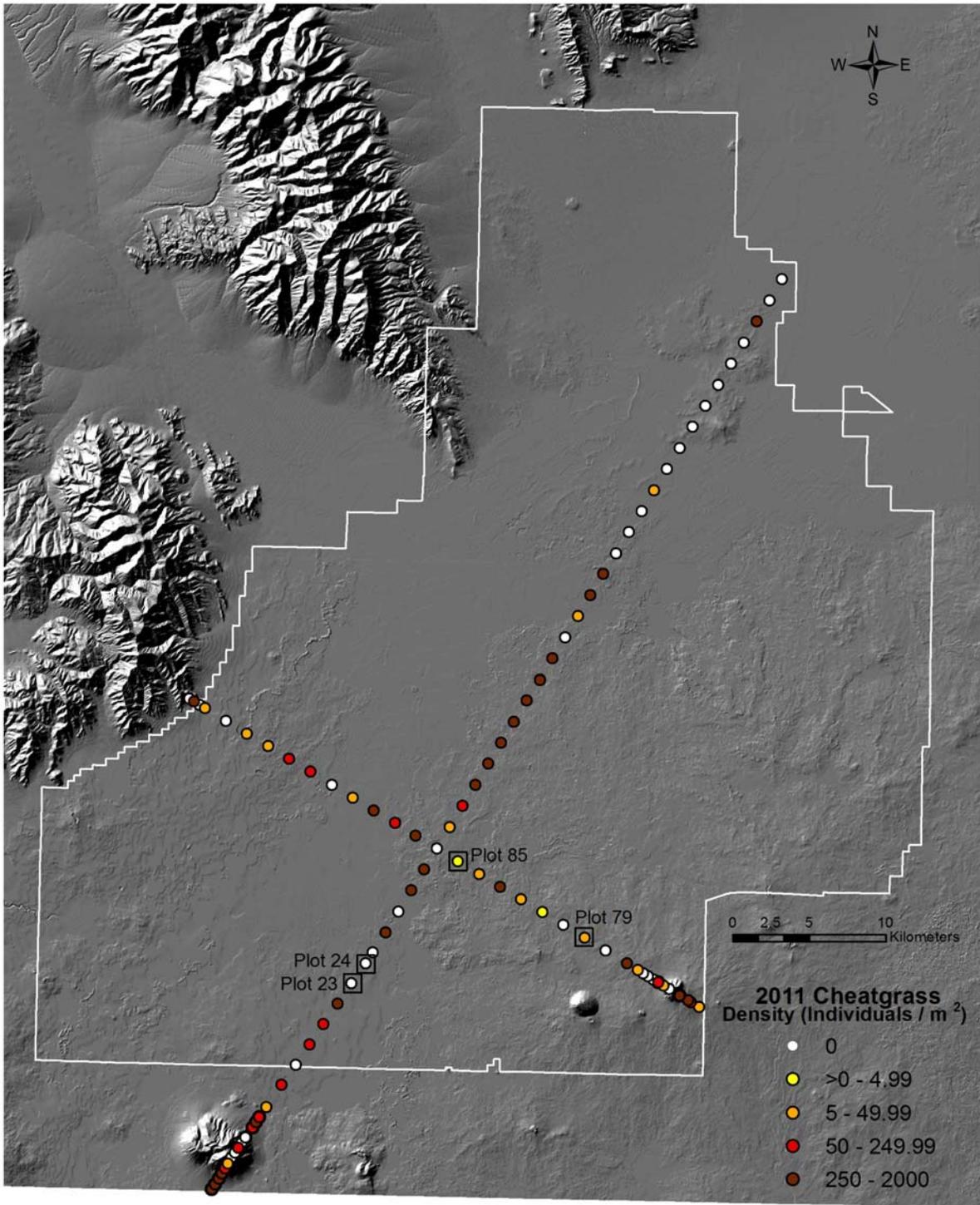


Figure 2-5a. *Bromus tectorum* distribution by density class on the Long-Term Vegetation Transect permanent plots at the Idaho National Laboratory Site in 2011.

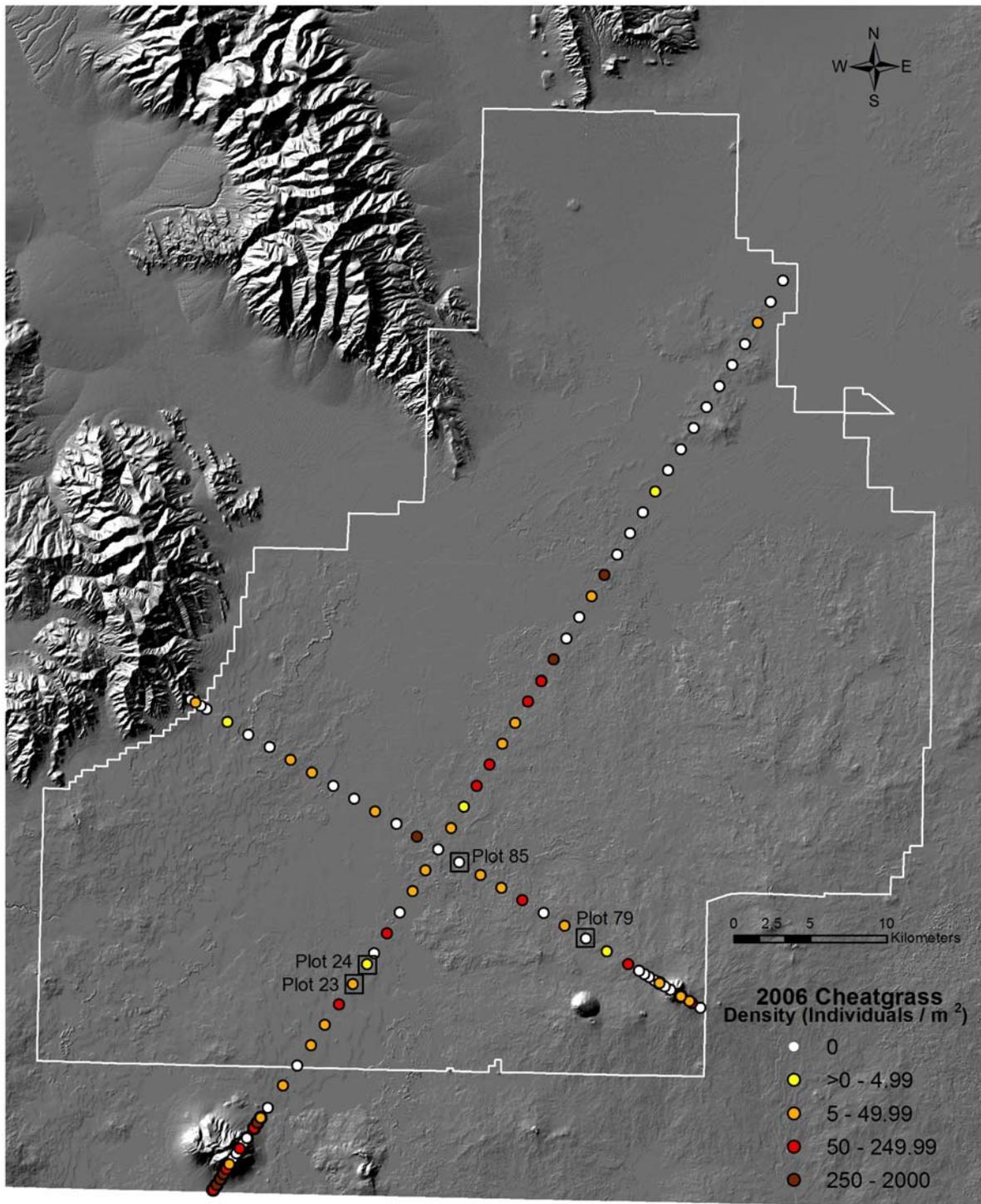


Figure 2-5b. *Bromus tectorum* distribution by density class on the Long-Term Vegetation Transect permanent plots at the Idaho National Laboratory Site in 2006.

Of greater concern than the general but modest increase in distribution of cheatgrass across the LTV Transects, is the dramatic increase in cheatgrass density when averaged across all active plots. Mean cheatgrass density during the 2011 sample period was more than double the mean of the next highest sample period (Figure 2-6) and when compared to each previous sample period, mean cheatgrass density was significantly higher in 2011 (Appendix A, Tables A-5a and A-5b). Prior to 2011, mean cheatgrass density of the most recent sample period did not significantly differ from that of the earliest sample period in 1950 (Forman et al. 2010). Between the 2006 and 2011 sample periods, cheatgrass frequency also increased and mean frequency was significantly higher in 2011 than all but the 2001 sample period (Appendix A, Tables A-5a and A-5b). Frequency is calculated as the percentage of density frames per plot in which cheatgrass is present; therefore, an increase in this metric indicates an increase in the small-scale, or local, distribution of the species.

The upsurge in cheatgrass abundance between the 2006 and 2011 sample periods marks a departure from historical patterns, which are generally characterized by fluctuations from one sample period to another. Forman et al. (2010) concluded that cheatgrass is widely distributed and seed is likely available across the INL Site, based on cheatgrass distribution analyses. They also suggested that because cheatgrass is so widely distributed, susceptible plant communities are at greater risk of cheatgrass dominance in the future. Rather than a true invasion into an area where the species did not previously exist, this recent increase may instead reflect the expansion of cheatgrass in susceptible plant communities where it already occurred as a minor component of the herbaceous understory.

One hypothesis, which may explain the significant increase in cheatgrass abundance over the previous five years, is that the current abundance of the species may reflect a change in the “carrying capacity” for cheatgrass based on the health of the native plant community. Or, from a different perspective, the resistance of the native plant community to dominance by cheatgrass may have decreased between 2006 and 2011. Decreases in the resistance of native sagebrush steppe plant communities could be attributed to any number or combination of biotic and/or abiotic factors. Some of the more readily assessed factors which may have affected native plant community health at the INL Site, and the entire region, over the past five years have been outlined in Chapter 1. They include increased soil disturbance, changes in the frequency and intensity of land use, short-term losses of shrubs from plant communities due to wildland fire, and large fluctuations in the seasonality of precipitation. Other potential contributing factors, which may be a little more difficult to evaluate, involve biotic interactions, weather-based fluctuations in species’ competitive abilities, and changes in water and/or nutrient balance which negatively impact the ability of native species to function optimally.

The “introduced annual forbs” functional group is largely dominated by desert alyssum, but also includes other species from the mustard family like herb sophia (*Descurainia sophia*) and tall tumbled mustard (*Sisymbrium altissimum*), and chenopods like Russian thistle (*Salsola kali*) and saltlover (*Halogeton glomeratus*). Introduced annual forbs were detectable in density frames of 45, or about half, of the active LTV plots during the 2011 sample period (Figure 2-7a). This is a decrease of seven plots between 2006 and 2011 and most of those plots are located in the northeast portion of the INL Site (Figures 2-7a and 2-7b). Although the overall distribution of plots with detectable densities of introduced annual forbs declined over the past five years, densities generally increased in the remaining plots. Most of the plots located within about 8 km (5 mi) of the intersection of the two LTV macro-transects had introduced annual forb densities of more than 250 individuals/m² in 2011 (Figure 2-7a). When compared with the distribution patterns of cheatgrass (Figures 2-5a and 2-5b), introduced annual forbs are much more predictable from one sample period to the next. With the exception of the seven plots in which species of this functional group became undetectable, densities of introduced annual forbs in each LTV plot generally increased to the next highest density class between 2006 and 2011 (Figure 2-7a and 2-7b).

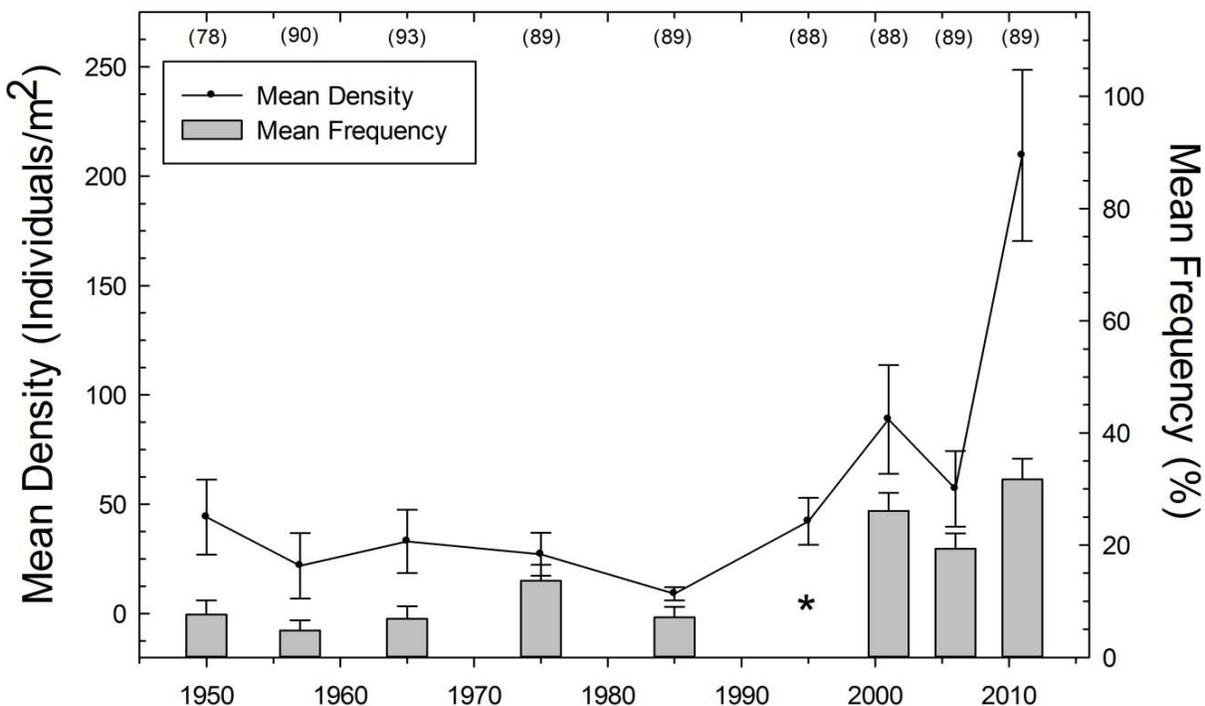


Figure 2-6. Density and frequency trends for *Bromus tectorum* on the Long-Term Vegetation Transect permanent plots at the Idaho National Laboratory Site from 1950 to 2011. Data are means \pm 1 SE. *Frequency data are missing from the 1995 data archives. Numbers in parentheses at the top of the frame indicate the number of plots for which data were available in each sample year.

Statistical trend analysis of introduced annual forb density confirms a quasi-exponential increase in abundance, which began around 1995 and continues through the 2011 sample period (Figure 2-8). Density of introduced annual forbs averaged nearly 340 individuals/m² across the 89 active LTV plots in 2011 and mean density was significantly greater in 2011 than in all prior sample periods (Appendix A, Tables A-6a and A-6b). Mean frequency increased slightly between 2006 and 2011, but the difference was not significant. However, mean frequency of introduced annual forbs was significantly greater in 2011 than it was in 2001, and it was significantly greater from 2001 to 2011 than in all prior sample periods (Appendix A, Tables A-6a and A-6b). Consequently, it appears as though the within plot-level, or small-scale distribution of species represented by this functional group, is increasing. The trend in frequency increase is more linear in nature than the density increase.

The abundance and distribution trends of cheatgrass and introduced annual forbs are not easily interpreted within the context of seasonal precipitation patterns. Forman et al. (2010) reported that native annual forbs function as ephemerals, where spikes in abundance tend to coincide with sample periods of higher than average spring and summer precipitation. Densities of native annual forbs during the 2011 sample period are consistent with this interpretation. Mean density of individuals in this functional group was slightly, but not significantly, lower in 2011 than in 2006 (Figure 2-9). Total spring precipitation was also lower in 2011 than in 2006 (Chapter 1, Figure 1-3). In general, a few sample years with the highest means are significantly different than a few sample years with the lowest means, for both density and frequency (Appendix A, Tables A-7a and A-7b). Nevertheless, the trend is not directional, and the sample periods with the highest densities and frequencies are also some of the wettest on record. In contrast, density and frequency data for cheatgrass and introduced annual forbs, primarily desert alyssum, confirm that abundance trends for these species are increasing directionally, they don't respond to precipitation as predictably, and they are currently more abundant by an order of magnitude when compared to native annuals.

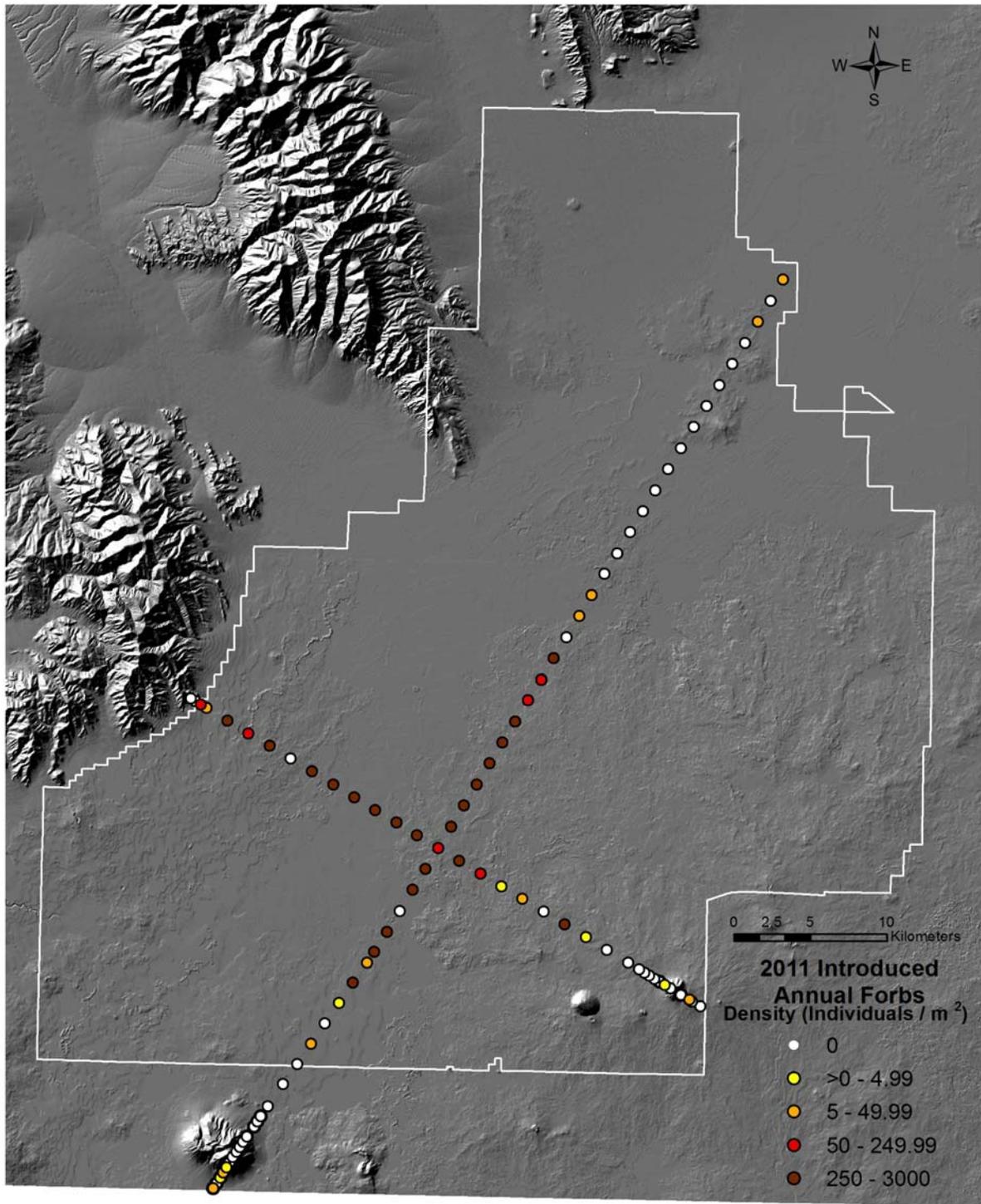


Figure 2-7a. Introduced annual forb distribution by density class on the Long-Term Vegetation Transect permanent plots at the Idaho National Laboratory Site in 2011.

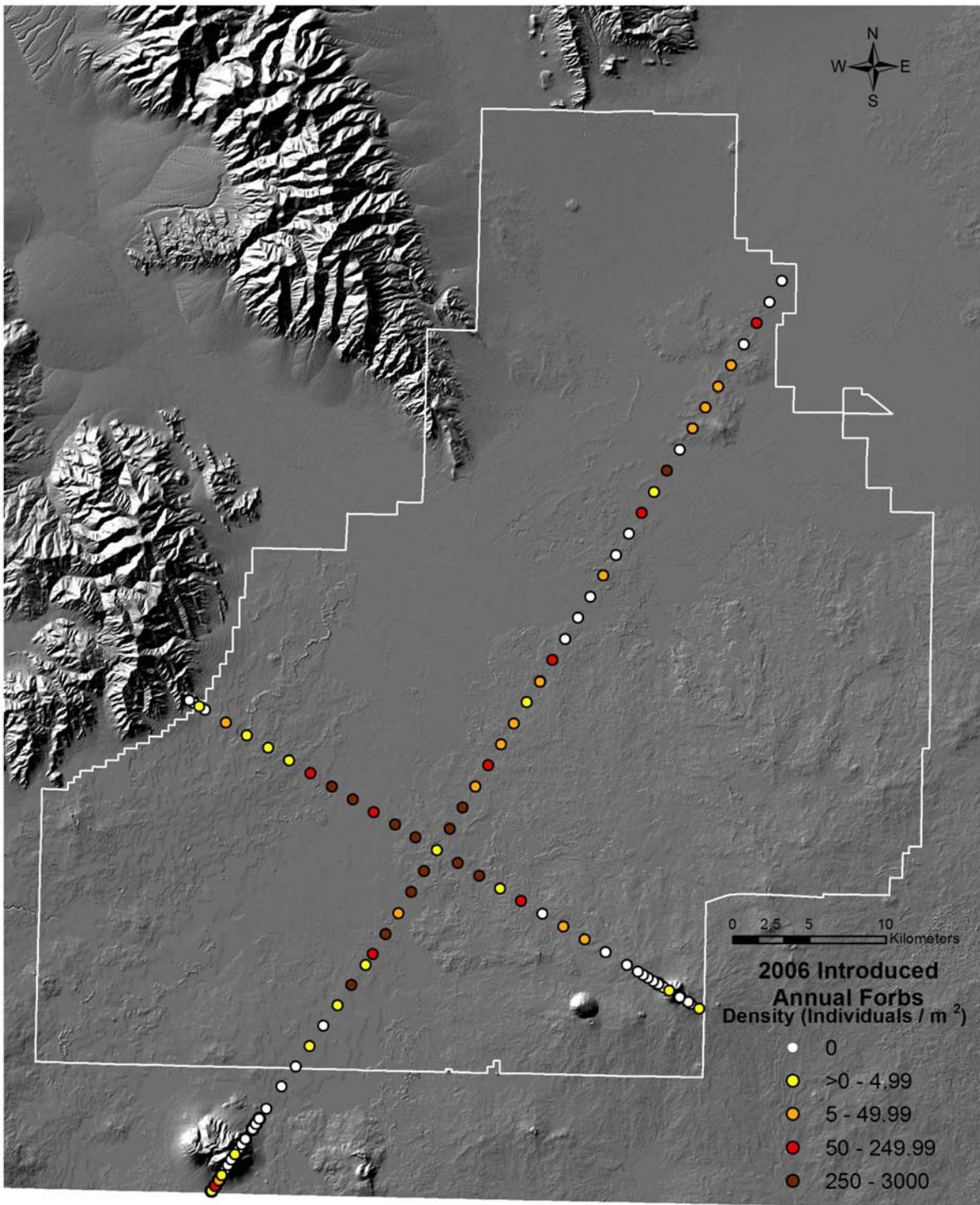


Figure 2-7b. Introduced annual forb distribution by density class on the Long-Term Vegetation Transect permanent plots at the Idaho National Laboratory Site in 2006.

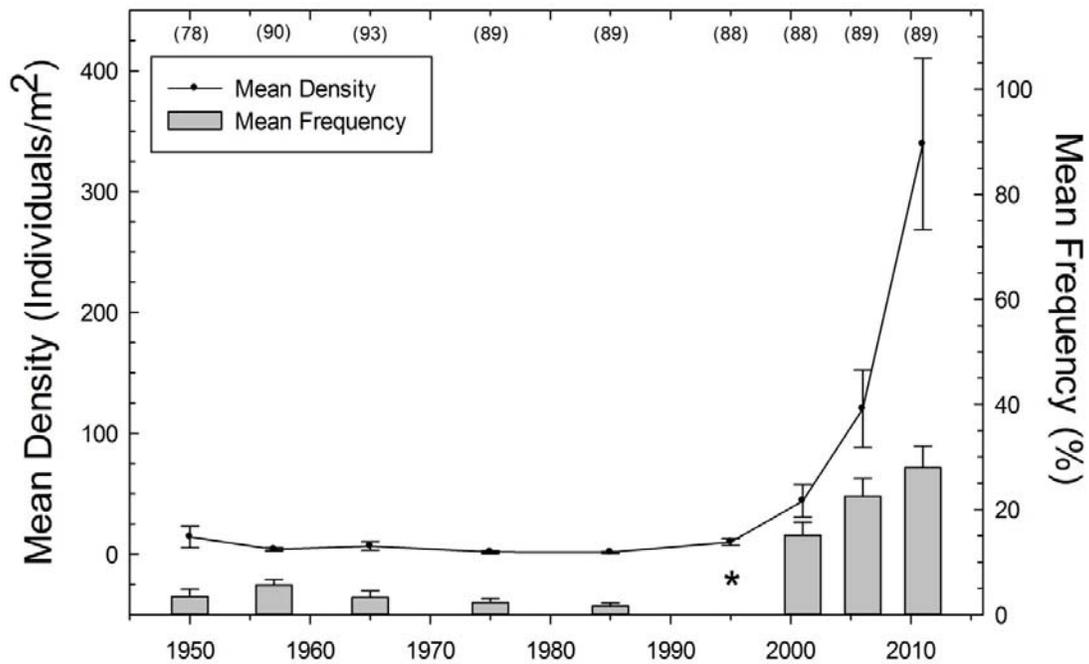


Figure 2-8. Density and frequency trends for introduced annual forbs on the Long-Term Vegetation Transect permanent plots at the Idaho National Laboratory Site from 1950 to 2011. Data are means \pm 1 SE. *Frequency data are missing from the 1995 data archives. Numbers in parentheses at the top of the frame indicate the number of plots for which data were available in each sample year.

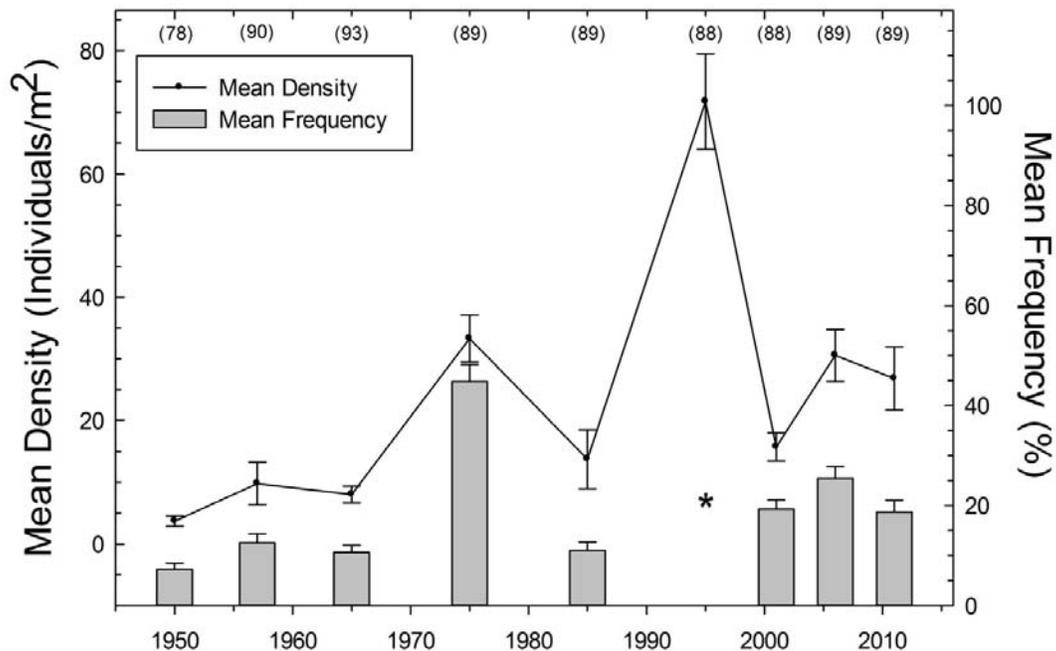


Figure 2-9. Density and frequency trends for native annual forbs on the Long-Term Vegetation Transect permanent plots at the Idaho National Laboratory Site from 1950 to 2011. Data are means \pm 1 SE. *Frequency data are missing from the 1995 data archives. Numbers in parentheses at the top of the frame indicate the number of plots for which data were available in each sample year.

2.4 Conclusions

A common theme emerging from the long-term trend analyses updated with the 2011 data is that vegetation composition on the LTV plots and across the INL Site is beginning to deviate noticeably from historical ranges of variability. Anderson and Inouye (2001) argued that native plant communities on the INL Site have the capacity to undergo compositional changes based on environmental fluctuations, but those changes are reversible and represent the extent of variation possible for healthy, native plant communities. Results from analysis of the 2011 dataset indicate, however, that the current distribution and abundance of introduced species has far exceeded values previously observed on the LTV plots.

Crested wheatgrass, cheatgrass, and desert alyssum all increased precipitously between the 2006 and 2011 sample periods. The significant increase in cheatgrass was somewhat surprising because it is a departure from its historical fluctuating trend. The increases in crested wheatgrass and desert alyssum were much more predictable, as the 2011 abundance values of those two species fell along increasing trend lines which were defined by previous analyses (Forman et al. 2010). The abundance trend for crested wheatgrass is one of linear increase, while the abundance trend for desert alyssum is one of exponential increase. Interestingly, these predictable trends indicate that these species are not responding to fluctuating environmental conditions, especially precipitation, in the same way as natives, which may allow them to exploit native plant communities experiencing environmental stress.

When considered in the context of ecological resistance and resilience theory, native plant communities on the INL Site may have potentially been exposed to greater environmental stress over the past five to ten years than in the previous several decades. Potential stressors include regional-scale changes in weather patterns, changes in land cover, and changes in land use, as discussed above. As the distribution and abundance of introduced species increases, the potential for these species to further alter environmental conditions for native species through indirect biotic interactions also increases. For instance, the soil moisture dynamics of a crested wheatgrass dominated plant community are very different from those of a native plant community (Anderson and Forman 2002). Additionally, desert alyssum is often abundant in interspaces, which may ultimately impact the spatial patterning of nutrient resources in native plant communities. Both of these examples are mechanisms through which native plant communities may become more invisable through time.

Trend analyses through the 2011 sample period indicate that natives have not yet been noticeably impacted by environmental stress and/or increases in the abundance of non-natives. In fact the current abundance metrics for native species and functional groups remain consistent with historical trends. Stabilization of big sagebrush abundance trends over the past decade is also positive. However, the abundance and distribution of non-natives is much greater than it has ever been, and as of the 2011 sample period, mean herbaceous cover from introduced species was equivalent to that from natives, when averaged across the LTV plots. Both of these results signify departures from historical trends of species composition and abundance and may be an indication of changes in vegetation condition moving forward.

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CHAPTER 3. THE T-17 FIRE: A DIRECT COMPARISON OF PLANT COMMUNITIES PRE- AND POST-FIRE

3.1 Introduction

Historically, wildland fire was a natural disturbance in the sagebrush-steppe ecosystem with average fire rotation intervals of 200 to 350 years (Baker 2011). An examination of aerial photographs from the Idaho National Laboratory (INL) Site reveals that fire scars 50, 100, and even approaching 200 years old are still identifiable, often within a matrix of larger and older sagebrush stands (unpublished data, Seanne Patrick-Buckwalter pers. comm.). This suggests that wildland fires were probably an infrequent event. However, increased human activity on roadways and in the remote portions of the INL Site have the potential to change that. Increased human activity brings with it an increase in the risk of fire ignition and the INL Site has experienced a number of large wildland fires since 1994.

Data collected on areas burned on the INL Site showed rapid re-establishment of native herbaceous species within two to three years following fire (Blew and Forman 2010, Buckwalter 2002). Similar post-fire native species cover was described on sagebrush rangelands near Pocatello, Idaho (Ratzlaff and Anderson 1995). Conversion to annual grasses following fire has not been documented on the INL Site except where pre-fire disturbance or firefighting efforts have resulted in the loss of herbaceous perennial cover. A recent study on the INL Site found that cheatgrass (*Bromus tectorum*) abundance declined following the 2010 Jefferson Fire (Rew et al. 2012) and Forman et al. (2010) reported that cheatgrass responded more like native ephemerals on the burned INL Site Long-Term Vegetation (LTV) Transect plots than as a true post-disturbance colonizer.

This is in contrast to other areas in the sagebrush steppe where the perennial understory plants have been replaced by exotic annual grasses like cheatgrass. In areas dominated by annuals, fire is likely to help promote dominance by annuals (West and Hassan 1985). Return of perennials on such sites generally requires intervention by artificial seeding. Non-native annual plants have been found to be dominant or co-dominant species in some INL Plant communities, but to date these areas are generally quite limited in spatial extent (Shive et al. 2011). At this time, post-fire seeding of native perennial remains unnecessary across much of the INL Site.

Generalizations from fire ecology studies from the INL Site and other southeast Idaho locations suggest that the plant community after a fire will be a reflection of the community present before the fire, with the exception of big sagebrush (*Artemisia tridentata*; Ratzlaff and Anderson 1995, Buckwalter 2002, Blew and Forman 2010). Native plant communities in good pre-burn ecological condition will return to diverse, native plant communities within a few growing seasons post-burn and can resist invasion by non-native species. These generalizations have been used to form the basis for management recommendations of burned areas on the INL Site and are the basis for the following guidelines (Blew and Forman 2010).

- 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.
- Managing for vigor of the perennial grasses should be the highest priority on burned areas.
- A healthy pre-fire plant community can increase the ability of a site to recover following fire even under very adverse conditions such as severe drought.

These generalizations have been based on the assumption that the pre-fire condition of the plant communities on those study sites reflected the general conditions reported for plant communities elsewhere on the INL Site or other anecdotal evidence. We have not had the opportunity to conduct a study under controlled conditions where the status and condition of the pre-fire plant community had been clearly documented. Furthermore, the most recent LTV data indicate increases in cheatgrass and other introduced annuals (see Chapter 2), which may affect vegetation recovery and associated management recommendations.

In the summer of 2011, the full suite of LTV data were collected and sampling was completed during the first week of August. On August 25, the T-17 Fire started and subsequently burned 11 LTV plots along T-17 (Figure 3-1). We re-sampled the burned plots using the same process during the same timeframe in 2012. These data will help us characterize good condition sagebrush steppe recovery in yearly increments with the added information regarding pre-fire vegetation condition, current weather patterns, proximity to disturbance, and presence/absence of invasive species.

3.2 Methods

3.2.1 Study Site

The 11 LTV plots that burned during the T-17 Fire are located along the northern arm of the macro-transect which generally runs north/south. Plots are spaced approximately every mile and are adjacent to T-17. Most plots were burned completely during the 2011 fire, but two plots had a few standing shrubs, which were mostly dead, at the periphery. Of the 11 burned plots, nine were previously classified as Big Sagebrush Shrubland Vegetation, and two were located in a complex between Wyoming Big Sagebrush Shrubland and Green Rabbitbrush/Streambank Wheatgrass (Western Wheatgrass) Shrub Herbaceous Vegetation (Shive et al. 2011, Figure 3-1). Additional detail regarding the vegetation and environmental conditions of the INL Site and LTV plots can be found in Forman et al. (2010).

3.2.2 Sample Design

The twelfth sampling of the LTV plots was completed as scheduled in 2011. The 11 plots which burned shortly thereafter were sampled again in 2012 using the same techniques. Three permanent transects within each plot were sampled using density counts, line interception, and point interception. Density and frequency were measured using 20, 0.3 x 1.0 m quadrats located along two transects of each plot. Line intercept data were collected along two transects and point intercept data were collected using 50 point frames placed along all three transects. Density and frequency data and point interception data were collected for all taxa, and line interception data were collected for perennial grasses and shrubs. Line interception measurements follow guidelines suggested by Canfield (1941) and point interception data were collected using a method developed by Floyd and Anderson (1982). See Forman et al. (2010) for detailed sampling methodology. In 2012, we sampled the plots during the same time frame (late-June to mid-July), within about one week of when they were sampled in 2011.

3.2.3 Statistical Analysis

Analyses comparing pre- and post-fire vegetation abundance and composition were conducted primarily with the point interception data because it is the only data type for which all taxa are sampled on the same scale. One-way repeated measures ANOVAs (Zar 1999) were used to compare pre- and post-fire functional group abundance. We used Jaccard's coefficient and percentage similarity to compare pre- and post-fire abundance and species composition for each plot. Both metrics were calculated according to Krebs (1999).

3.3 Results

In general, one year following fire, total vegetation cover was considerably reduced on all 11 plots. Total absolute cover ranged from about 25% to over 60% in 2011 and was reduced to between 4% and 14% in 2012 (Figure 3-2). Decreases were significant for shrubs, native perennial forbs, succulents, and introduced annuals/biennials. Mean cover values were not significantly different pre- and post-fire for native perennial graminoids or for native annual and biennial forbs (Table 3-1, Appendix B). Post-fire cover values indicate little, if any recovery of shrubs on the burned plots at one year. In 2012, four plots had no shrub cover, six had less than 1% shrub cover and one plot had a little over 2% shrub cover (Figure 3-2, Appendix C). Comparatively, pre-fire absolute cover values of native shrubs ranged from a little more than 10% to nearly 30% (Figure 3-2, Appendix C). Pre-fire, big sagebrush was the most abundant shrub on all of the plots (Appendix C). Because big sagebrush doesn't re-sprout after fire, it is not surprising that it was absent on all but one plot (0.67%) one year post-fire.

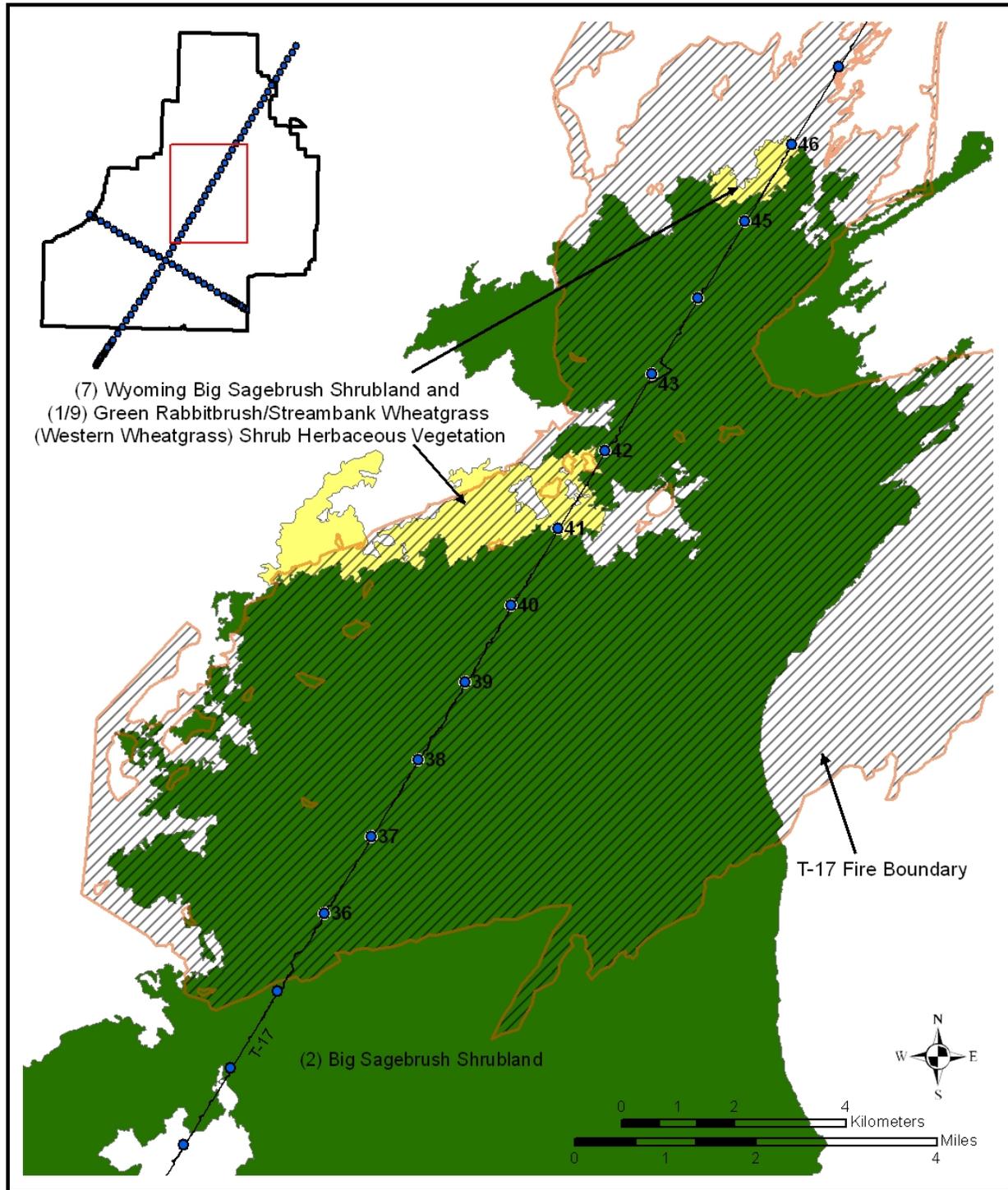


Figure 3-1. Location of 11 Long-Term Vegetation Transect plots which burned during the 2011 T-17 Fire. Vegetation classes are from Shive et al. (2011).

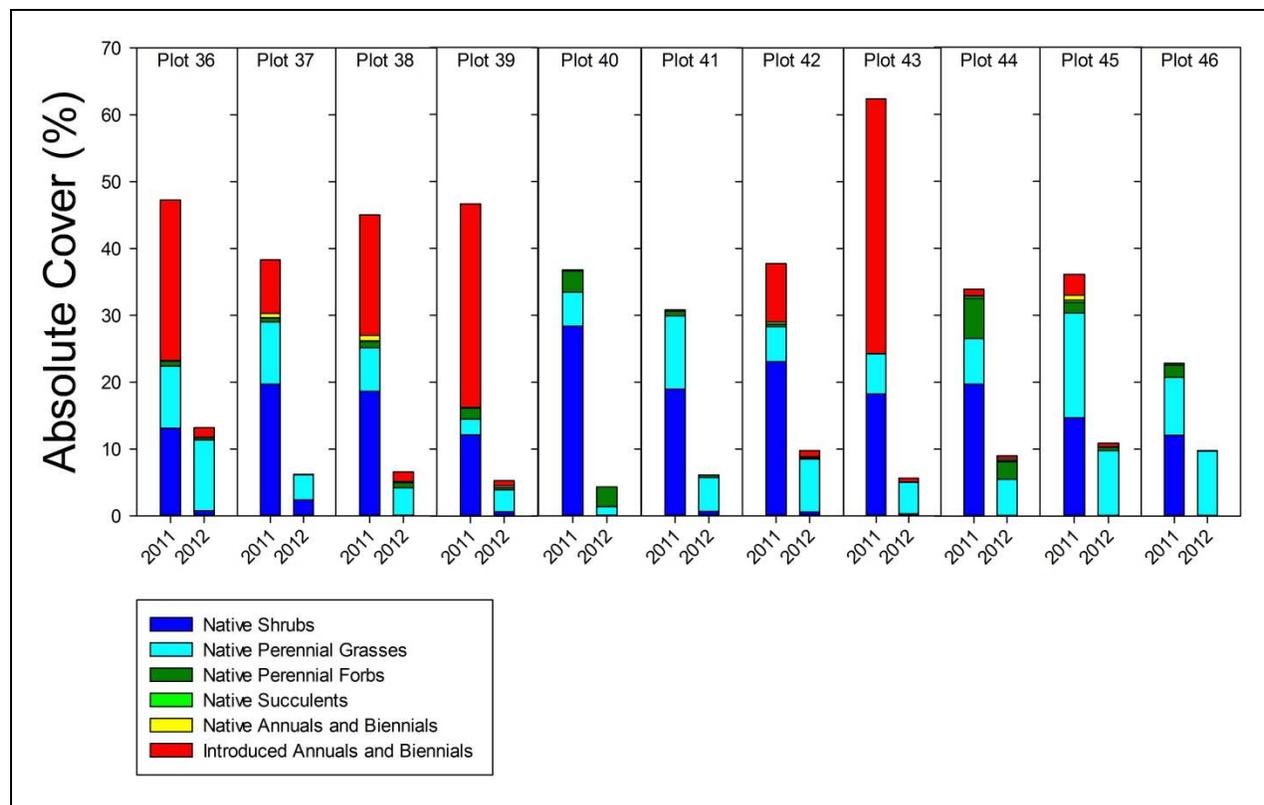


Figure 3-2. Absolute cover by functional group for 11 Long-Term Vegetation Transect plots at the Idaho National Laboratory Site before (2011) and after (2012) the T-17 Fire.

Native perennial grasses recovered well within the first year post-fire. When averaged across the burned plots, the cover of native perennial graminoids decreased by only about 2%, and that decrease was not significant (Table 3-1). Interestingly, rhizomatous grass cover, specifically from streambank wheatgrass (*Elymus lanceolatus*), was higher post-fire on nine of the 11 plots, and remained the same on one. In fact, streambank wheatgrass was the dominant post-fire grass on most plots, even when it had not been the most abundant grass pre-fire (Appendix C). While perennial bunchgrass cover from some species increased on some plots post-fire, absolute cover from a single species seldom exceeded 1% (Appendix C). Indian ricegrass (*Achnatherum hymenoides*), squirreltail (*Elymus elymoides*), needle and thread (*Hesperostipa comata*), Sandberg bluegrass (*Poa secunda*), and sand dropseed (*Sporobolus cryptandrus*) all increased in at least one plot post-fire. Plot 40 was the only plot that did not have a post-fire increase in native grasses, but cover of lemon scurfpea (*Psoraleidium lanceolatum*), a native perennial forb, did increase slightly between 2011 and 2012. Lemon scurfpea was one of the most abundant herbaceous species on this plot pre-fire as well. (Figure 3-2, Appendix C).

One very interesting result of the post-fire survey is the significant decrease of introduced annuals and biennials. On average, the absolute cover of introduced species dropped from 12% to less than 1% and the decrease was statistically significant (Table 3-1, Appendix B). Two introduced species which are becoming increasingly problematic across the LTV plots as a whole (see Chapter 2), cheatgrass and desert alyssum (*Alyssum desertorum*), decreased substantially or became undetectable on plots where they had been documented pre-fire during the 2011 sampling (Appendix C). In fact, non-native annuals were nearly co-dominant in the herbaceous understory of a few plots pre-fire. One introduced biennial species, yellow salsify (*Tragopogon dubius*), appeared on six of the 11 plots post-fire and it was only present on one plot pre-fire. We have not previously noted a relationship between fire and yellow salsify; however, it is an opportunistic species with a hardy and readily dispersed seed.

Table 3-1. Mean absolute cover by functional group and one-way repeated measures ANOVA results comparing pre- and post-fire vegetation on 11 Long-Term Vegetation Transect plots at the Idaho National Laboratory Site.

	2011	2012	Significant
Native Shrubs	18.04	0.48	Yes
Native Perennial Graminoids	7.81	5.98	No
Native Perennial Forbs	1.60	0.74	Yes
Native Succulents	0.16	0.03	Yes
Native Annuals and Biennials	0.23	0.09	No
Introduced Annuals and Biennials	11.96	0.55	Yes

Species richness declined post-fire on seven plots and increased on two plots (Table 3-2). The Jaccard coefficient was used to compare species composition on each plot pre- and post-fire; it ranged from 0.21 to 0.60 and only four plots were above 0.50 (Table 3-2). This result indicates that in terms of species composition, on seven of the 11 plots more than half of the species present in a plot pre-fire were lost or replaced with a different species post-fire. Percentage similarity, which factors in the relative abundance of each species, was below 50% for all plots (Table 3-2). For many plots, the low percentage similarity value reflects a shift from dominance by shrubs pre-fire to dominance by herbaceous species post-fire. The relatively low values for the Jaccard coefficient, on the other hand, reflect a change in the identity of species occupying each plot. The loss of shrub species would decrease the Jaccard coefficient between the pre- and post-fire sample periods, but only a few shrub species were present in each plot, so several herbaceous species likely also changed between 2011 and 2012. The changes in herbaceous composition may have resulted from the fire, or they may reflect shifts in ephemerals due to differences in environmental conditions between one growing season and another.

Table 3-2. Species richness, Jaccard coefficient, and percentage similarity comparing pre-and post-fire composition of 11 Long-Term Vegetation Transect plots on the Idaho National Laboratory Site.

	2011 Species Richness	2012 Species Richness	# of Species Common to Both Sample Periods	Jaccard Coefficient	Percentage Similarity
Plot 36	16	16	12	0.60	35.41
Plot 37	18	5	5	0.28	33.26
Plot 38	18	11	8	0.38	37.52
Plot 39	18	18	13	0.56	26.64
Plot 40	11	4	4	0.36	15.60
Plot 41	13	4	4	0.31	33.69
Plot 42	12	14	8	0.44	25.68
Plot 43	10	9	7	0.58	14.94
Plot 44	14	15	10	0.53	41.45
Plot 45	19	10	8	0.38	47.84
Plot 46	13	4	3	0.21	14.62

While much of the reduction in cover between the two sample periods can be attributed to post-fire shrub losses, precipitation was well below average during the first half of 2012 (January – June). Less than 5 mm of precipitation fell in three of those six months. Although July precipitation was above average, it is typically much too late in the summer for most species to germinate (Figure 3-3). Additionally, much of the sampling had been completed earlier in the season, so any increase in late summer cover would not have been documented. The very dry conditions which persisted through the late winter and spring of 2012 may help explain some of the significant post-fire declines in some herbaceous functional groups, like native perennial forbs and introduced annuals and biennials. The lack of precipitation during the early fire recovery period also makes the post-fire abundance of native perennial grasses, especially rhizomatous grasses, quite remarkable.

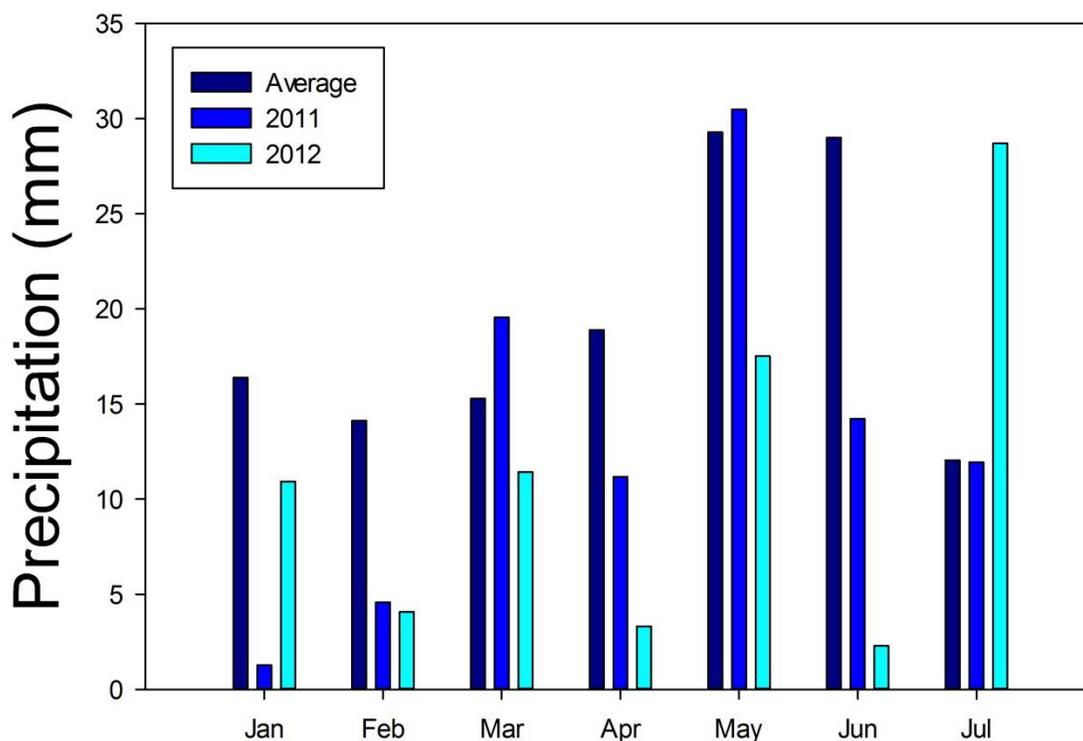


Figure 3-3. Total monthly precipitation from late winter through early summer for 2011 and 2012. Average precipitation is calculated from 1950-2012. Precipitation data are from Central Facilities Area, Idaho National Laboratory Site.

3.4 Discussion

In recent post-fire recovery studies at the INL Site (Rew et al. 2012, Blew and Forman 2010), we have observed that the native species composition of post-fire communities generally reflect that of pre-fire communities, with exception of shrubs. Good condition sagebrush steppe communities generally recover post-fire to native species which were a component of the plant community pre-fire. Some post-fire vegetation recovery studies have even suggested that the post-fire abundance of many species may be more affected by precipitation than by fire (Rew et al. 2012, Colket and Bunting 2003). Within this context, many of the changes identified in vegetation on the LTV plots after the T-17 Fire were expected, but a few results were unexpected and we recommend that additional assessments on these plots should be conducted in the near future to ascertain the process of post-fire recovery.

Despite the extremely dry conditions during the first growing season following the T-17 Fire, recovery of native herbaceous species, particularly perennial grasses, was notable. We reported a similar response from the post-fire plant community on the Tin Cup Fire scar, where the third growing season post-fire was exceptionally dry, but native grass cover was relatively unaffected (Blew and Forman 2010). On the T-17 Fire, streambank wheatgrass showed the most positive recovery response when compared to other species, native or introduced. Rhizomatous grasses, especially streambank wheatgrass, appear to be very resilient in the short-term and may provide important cover and soil stabilization immediately post-fire, even with less than optimal precipitation. Healthy rhizomatous grass populations may be essential to favorable recovery in the years immediately post-fire, especially under conditions of high water stress.

Introduced annual species are of particular concern in post-fire sagebrush steppe communities as they can, under some conditions, dominate entire regions (e.g. Monsen and Kitchen 1994). Results from this limited data set, indicating a striking post-fire decline in introduced annual abundance, suggest a different post-fire response of introduced annuals than may be otherwise expected. Post-fire decreases in the abundance of cheatgrass and desert alyssum on the plots affected by the T-17 Fire were also counter to recent trends for these species at the INL Site. In terms of density, long-term trend analyses from the LTV plots indicate increases in introduced annuals between the 2011 and previous sample periods (see Chapter 2).

Annual species, both introduced and native, are responsive to precipitation throughout the course of a growing season. Many annuals function as ephemerals and the timing of precipitation may be as important as the total amount. The LTV data suggest that the precipitation response is different for natives as opposed to introduced species; natives respond predictably to gross precipitation totals, while the responses of introduced species are less straightforward and are likely linked to other environmental conditions (see Chapter 2). Nevertheless, low cover values for all annual species after the T-17 Fire is at least partially attributable to very little precipitation during the early spring/summer period prior to the first post-fire growing season.

Aside from the effects of precipitation on post-fire abundance of annual species, this is not the first data set from the INL Site to document post-fire decreases in cheatgrass. Similar decreases have occurred on other individual LTV plots subsequent to fire. Forman et al. (2010) discuss a handful of plots which showed steady increases in cheatgrass density over several decades, but experienced steep declines in cheatgrass abundance for one or more sample periods (up to ten years) post-fire. In several of these plots, cheatgrass density decreased to undetectable levels in the sample year immediately after the fire. These plots experienced a variety of post-fire precipitation patterns because they are located in different burn scars from fires which occurred over different time periods. Unfortunately, the LTV data aren't sufficient to adequately characterize this pattern or test mechanism which may be driving it due to the intermittent nature of the data collection. More recent research from the INL Site, which is independent of the LTV plots, also confirmed patterns of post-fire decreases in cheatgrass abundance over substantial spatial extents (Rew et al. 2012).

The minor increase in yellow salsify across the 11 plots which burned in the T-17 fire is the only apparent increase noted for an introduced species after the fire. Ideally, monitoring these 11 plots annually for the next 3 to 5 years may provide insight on the recovery of native species and on the redistribution and spread of introduced species following fire. Short-term annual data collection will also allow us to characterize the relative importance of precipitation on recovery, especially under more moderate conditions than occurred in 2012.

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CHAPTER 4. BIG SAGEBRUSH POPULATION ECOLOGY: DEMOGRAPHY OF A FOUNDATION SPECIES

4.1 Introduction

Historically, nearly three quarters of the Idaho National Laboratory (INL) Site was dominated by big sagebrush (*Artemisia tridentata*; McBride et al. 1978). Currently, only about one half of the INL Site is mapped as communities with a big sagebrush canopy (Shive et al. 2011). In terms of spatial extent, much of the loss of big sagebrush over the past few decades is from wildland fires. Although big sagebrush losses from fire are quite dramatic because of their “acute” nature, “chronic” declines in big sagebrush abundance in areas where it once dominated are equally as concerning. Between 1975 and 2006 average big sagebrush cover on the unburned, core LTV plots declined from more than 20% to less than 10% (Forman et al. 2010). The cause of this trend is unknown, but Anderson (1986) suggested that it began as part of a widespread big sagebrush die-off in the mid- 1970’s and has persisted, at least in part, because of a general lack of vigor.

Loss of big sagebrush vigor has often been described as decadence, where a decadent individual has experienced the death of over 50% of its aboveground biomass (Kindschy 1994) and a decadent stand is thought to be the result of an older stand age distribution (Anderson 1986, Bunting et al. 1987, Cook et al. 1994). Though very little empirical research has been conducted in an effort to demonstrate a clear relationship between stand age structure and stand condition, many management recommendations for decadent stands are based on the assumption that they are old. Before cheatgrass became a critical rangeland management issue for example, prescribed fire was a commonly recommended management strategy because both intentional fire suppression (Cook et al. 1994) and lack of fire due to natural stochasticity (Bunting et al. 1987) were thought to be primary causes for stand decadence. The logical inference was that individuals in a decadent stand are too old to maintain recruitment rates necessary for the stand to perpetuate itself. The management implication based on that inference was that an improvement in stand condition may be achieved via disturbance to remove old individuals to make space for younger, more fecund shrubs. However, more recent field research suggests that this logical inference is not supported.

A field investigation reported by Wambolt et al. (2001) indicated that unburned sites tend to support more juvenile individuals than burned sites and other investigators have concluded that sagebrush stands can maintain themselves indefinitely in the absence of fire (Lommasson 1948, Passey and Hughie 1962, Wambolt and Hoffman 2001). Furthermore, the average age at which big sagebrush individuals are reported to die is less than half the potential maximum lifespan for this species (Booth 1947, Ferguson 1964, Wambolt and Hoffman 2001). Results from these studies suggest that decadence may not be a direct result of declines in reproduction because of older stand age distributions and that there is a disconnection between our current understanding of stand-age structure and its relationship to stand condition for big sagebrush. This knowledge gap is manifested in management strategies which fail to address the real cause or causes of declines in stand condition. It is clear that more research on big sagebrush population dynamics is necessary before effective management strategies can be developed to address stand-level declines.

In order to better understand losses in big sagebrush abundance and declines in stand condition at the INL Site, we conducted a study on the population structure of big sagebrush stands. The sagebrush demography study was designed with three primary objectives in mind:

1. To determine the typical stand age structure or range of stand age structures for mature big sagebrush stands. Demographic data may be used to determine mean stand age, characterize the age distribution of stands, define the turnover rates of populations, understand variability in age distributions from one stand to another, and measure the effects of population status on the abundance of other species in the stand. Big sagebrush has been described and managed as a long-lived woody species and Miller (1976) postulated that long-lived species cannot maintain stable age distributions indefinitely,

largely due to local environmental fluctuations experienced by populations over many years. Though the abiotic conditions which define the semi-arid sagebrush steppe ecosystem are marked by abrupt spatial and temporal discontinuities in resource availability, very little work has been done to understand how big sagebrush populations change over the landscape and through time. The first objective of this study addresses basic stand level age distributions and characterizes the range of variability which may be associated with big sagebrush population metrics across several stands at the INL Site.

2. To investigate how stand age structure relates to shrub die-off and stand condition for big sagebrush. As discussed above, the age or population status of big sagebrush stands may not be easily inferred from stand condition, vigor, or the appearance of “decadence.” In fact stands may appear to have an abundance of standing dead individuals or low sagebrush cover for any number of reasons. Stand die-off may result from small mammal herbivory (Frischknecht and Baker 1972), insect defoliators (Gates 1964, Pringle 1960), fungal diseases (Sturges and Nelson 1984), or climactic stressors like drought or abnormally wet conditions which cause anoxic soil conditions (Nelson et al. 1989). Objective 2 will explore the relationship between the apparent condition of a stand and its age structure, and determine the extent to which factors other than age may influence the condition, or health of a stand.
3. To examine the dynamics of sagebrush stand replacement in the absence of wildland fire. Objective 3 will focus on understanding how mature big sagebrush stands replace themselves. The INL Site supports extensive stands of big sagebrush, many of which have not burned within at least the past few hundred years. With a better grasp of the patterns and processes controlling recruitment in mature stands, we can develop more effective monitoring strategies and appropriate management actions to increase recruitment and improve the habitat value of a stand for sagebrush obligates. Specific hypotheses about the frequency of recruitment events and the consistency of annual recruitment patterns across multiple stands will be addressed with this objective.

4.2 Methods

4.2.1 Study Site

This study was conducted on the INL Site, which is located on the Snake River Plain in Southeast Idaho. The INL Site occupies about 2,300 km² and has a mean elevation of about 1,500 m. The surficial geology of the area reflects local volcanic activity, primarily basalt flows which range in age from about 3,000 years on the southern boundary to nearly one million years in the northwestern extent of the INL Site. Fluvial and alluvial processes associated with the Big Lost River, lakebed sediments from ancient Lake Terreton, alluvial deposits adjacent to surrounding buttes and mountain ranges, and accumulations of loess have resulted in soils which range from a few centimeters to several meters deep over the basalt bedrock. Windblown soils generally tend to accumulate in depressions and on the leeward sides of basalt ridges (Anderson et al. 1996).

The INL Site climate is continental in nature and is characterized by large diurnal and seasonal temperature fluctuations. Average annual temperature is 5.6 °C and average annual precipitation is approximately 208 mm. A portion of the annual precipitation occurs as snow and snow cover can persist from a few weeks to several months each winter. April, May, and June tend to be the wettest months each year and much of the deeper soil moisture recharge annually is from a combination of snowmelt and spring rain. Minimal cloud cover, especially through the summer months, results in solar heating of the surface during the day and radiant cooling to the atmosphere at night. Low humidity and frequent, moderate winds also add to the high level of environmental exposure to biota in this high desert environment.

The vegetation of the INL Site is typical of the greater sagebrush steppe ecosystem in North America. Big sagebrush dominated communities generally host a diverse component of native forbs and perennial grasses, including both rhizomatous and bunch grasses. Across the INL Site, Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) is abundant and widespread, while basin big sagebrush (*Artemisia tridentata* ssp. *tridentata*) occurs in a more localized distribution, where soils are deeper and have greater available soil moisture (Shumar and Anderson 1986). Plant communities which are not dominated by big sagebrush, either because they have been burned in recent wildland fires or because they do not have the appropriate abiotic resources to support big sagebrush stands, are often dominated by native grasses (e.g. *Hesperostipa comata*, *Achnatherum hymenoides*, *Elymus lanceolatus*, *Elymus elymoides*), green rabbitbrush (*Chrysothamnus viscidiflorus*), salt desert shrubs (*Atriplex confertifolia*, *Grayia spinosa*, and *Atriplex falcata*), dwarf sagebrush species (*Artemisia nova* and *Artemisia arbuscula*), or Utah Juniper (*Juniperus osteosperma*). Non-native species are abundant in some INL Site plant communities, the most common of which are crested wheatgrass (*Agropyron cristatum* or *Agropyron desertorum*), cheatgrass (*Bromus tectorum*), and desert alyssum (*Alyssum desertorum*).

4.2.2 Study Design

Vegetation sampling occurred in June and July of 2006. Fourteen plots were located in the central portion of the INL Site. We chose plot locations which were relatively undisturbed. There were no facilities or upgraded roads in close proximity to the study area, livestock grazing had been discontinued about 55 year prior to data collection on the study area, and there were no known fires on the study area within the century before sampling took place. A vegetation map, which was completed subsequent to sampling for this project (Shive et al. 2011), indicated that twelve plots were located in mixed and/or hybridized big sagebrush communities and two plots were located in Wyoming big sagebrush communities (Figure 4-1). Mixed and/or hybridized big sagebrush communities are generally dominated by Wyoming big sagebrush, but contain small, localized stands of basin big sagebrush or individuals with intermediate morphological characteristics between those two subspecies. Sample plots were 1 ha in size and were positioned so as to sample visually homogenous stands of Wyoming big sagebrush. We selected plots which represented a wide range of conditions with respect to big sagebrush size, density, cover, and apparent decadence. All but two plots were spaced at least 1km apart to avoid spatial autocorrelation and in an attempt to sample plots with different local environmental conditions. Two plots, which are about 500 m apart, were located in closer proximity to one another in order to help populate the middle of the “decadence” continuum we were attempting to sample. All plots were located within 15 km or less of one another. Plot identification numbers are discontinuous because more than 14 potential plot locations were initially identified, but only 14 plots were ultimately sampled.

Plots were sampled for shrub cover by species, sagebrush density, sagebrush condition rank, and stem cross sections were collected for live and dead individuals. All variables were sampled using a stratified random design. For each 1 ha plot, we established a 100 m baseline with ten transects, each 100 m in length and perpendicular to the baseline (e.g. Maier et al. 2001). Transects were randomly located along the baseline. Shrub cover was sampled along 25 m segments of eight transects using line interception to measure shrub canopy (Canfield 1941); we selected random locations for the start point of each 25 m segment. We estimated big sagebrush density using shrub counts in belt transects which were 1 m x 10 m and located randomly along each of the ten sample transects. One hundred big sagebrush individuals, ten individuals along each of the ten transects, were selected randomly and ranked according to condition. We used a five-point scale (Table 4-1) to describe condition.

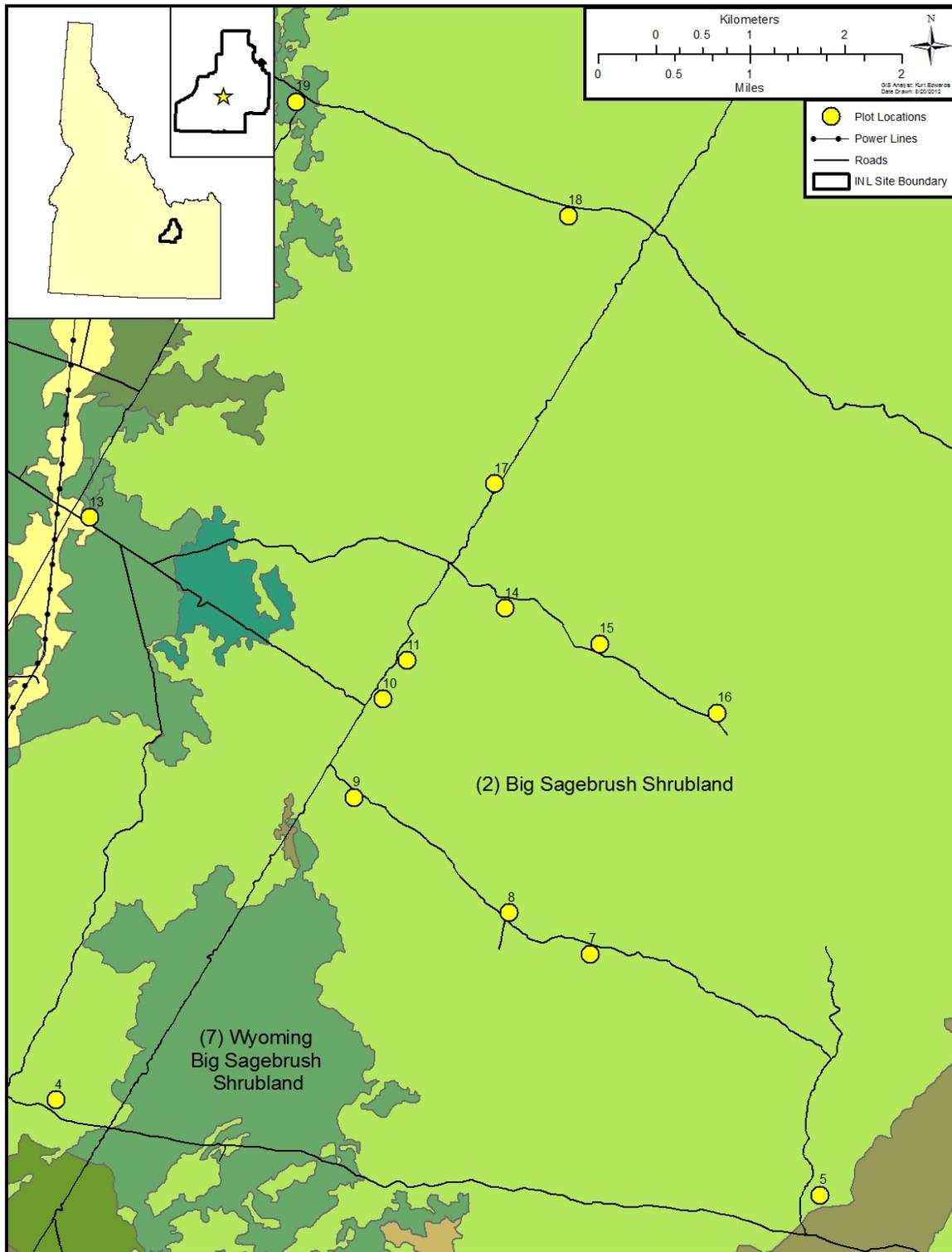


Figure 4-1. Locations of the 14 plots sampled for the sagebrush demography study. Vegetation classes are from Shive et al. 2011.

Table 4-1. Condition rank classes used for describing the condition of individual big sagebrush shrubs for the sagebrush demography study.

Condition Rank	Condition Description
1	Shrub is completely dead but standing and still rooted.
2	Approximately ¼ of the shrub's canopy is alive.
3	Approximately ½ of the shrub's canopy is alive.
4	Approximately ¾ of the shrub's canopy is alive.
5	All or nearly all the shrub's entire canopy is alive.

Stem cross sections from approximately 70 individuals were collected from each plot for determining shrub ages via ring counts. We randomly selected 5 individuals with at least a partial living canopy from each of the ten sample transects, for a total of 50 live shrubs. Once a random point was selected along the transect line, the individual with the closest stem was removed at the soil surface. If there were no living individuals within 10 m of the random point, we didn't sample an individual from that location; therefore, fewer than 50 live individuals were sampled for some plots. We also selected five standing dead individuals at random from each of four sample transects, for a total of 20 dead big sagebrush shrubs. See Figure 4-2 for a representative diagram of the plot sampling layout.

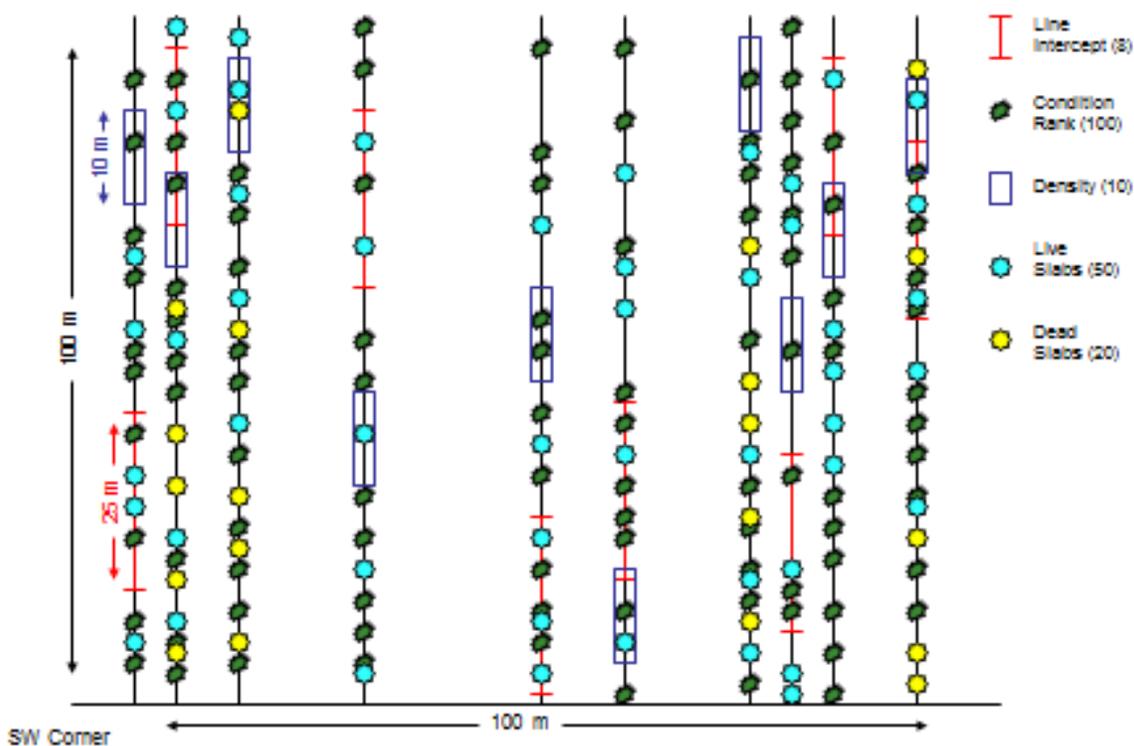


Figure 4-2. Representative diagram showing the plot layout for a sagebrush demography study at the INL Site (not to scale).

Once the big sagebrush individual with the stem closest to the sample point was located, the stem of the plant was wrapped with wire a few centimeters above the soil surface. The stem was then removed at the soil surface. We used pruning shears for removing small individuals and a reciprocating saw for removing larger individuals. For multi-stemmed shrubs, we ensured that we collected the stem containing the central structure (i.e. the innermost ring and center pith, if it was still intact). We also collected any stems adjacent to the main stem if they appeared to be part of the same sequence but were detached due to delamination between two annual rings. All but about 10 cm of the base of each stem was removed and discarded as stem cross sections were collected in the field.

Big sagebrush stem sections were prepared and counted in the laboratory according to guidelines established by Ferguson (1964). The base of each section was sanded on a belt sander using three, progressively finer grits. We then applied vegetable oil to the samples to enhance contrast between rings. We counted rings under a dissecting stereoscope at about 20x magnification. A few samples were missing their center pith and some of their innermost rings. When this occurred, we estimated the number of missing rings based on the diameter of the missing section compared with the rest of the cross-section. Each sample was counted independently by two different observers. When the number of rings counted by both observers was not the same for a cross-section, a third observer made a final determination of the age of the sample.

4.2.3 Data Analysis

Because this study is descriptive in nature, bar charts and summary statistics are used extensively to characterize stand age distribution and compare age structure among stands. We also used multiple correlation analysis to assess the strength of the relationships between variables sampled in each plot. SigmaStat was used for both summary statistics and multiple correlation analyses and SigmaPlot was used for graphical analyses.

4.3 Results

4.3.1 General Age Distributions and Precipitation Patterns

A total of 636 big sagebrush stems were collected from live shrubs and ring counts from those stem cross sections indicate mean age of all live individuals pooled across all sample plots is 14.9 years. The youngest individuals sampled germinated in the year sampling occurred and the oldest individual germinated in 1921, 85 years prior to sampling for this project. Because many seedlings do not persist past the first few growing seasons due to self-thinning and environmental stress (e.g. Owens and Norton 1989), we were concerned that stand age distributions may be skewed by an abundance of individuals which would not mature and contribute to the continued reproduction of the stand. However, only 43 individuals, or less than 7% of the total sample, were two years old or less. Removing individuals less than three years old from the pooled mean only increases it by about one year to 15.9 years. Individuals more than two years old are generally reproductively mature and begin producing seed under all but the most extreme environmental conditions (Smith et al. 1997). The pooled mean age of live shrubs from this study was generally much younger than we had anticipated. Only 14 shrubs, or about 2% of the live individuals sampled, were more than 40 years old.

In a summary of annual age classes pooled across all sample plots (Figure 4-3), there was at least one individual represented in every age class between 1970 and 2006. These results indicate at least some recruitment occurs every year, even very dry years (Figure 4-4). Going back beyond 1970, it is not possible to interpret the lack of individuals in an annual age class as lack of recruitment that year or as losses of those individuals from mortality. The pooled age class data also show variability in the number of individuals represented in each age class each year, which may stem from irregular recruitment patterns related to precipitation trends. Within the decade prior to sampling for this study, 1998 is the annual age class with the greatest number of individuals (Figure 4-3) and it was also the year with the highest total annual precipitation (Figure 4-4). Likewise, three consecutive years within the decade prior to sampling for this study were among the driest on record, each with less than half of the mean annual precipitation (Figure 4-4). The sagebrush age classes corresponding to these dry years also have relatively few individuals when compared to other age classes within that decade. Though it is possible that smaller age classes

reflect mortality rather than recruitment, it is unlikely that individuals were established during very dry years and were lost in subsequent years with more favorable weather conditions, especially because these individuals were only two to four years old when the data were collected.

Further evidence supporting the relationship between general precipitation patterns and the abundance of recruited individuals is found by comparing decadal precipitation patterns and corresponding age class counts. The decade prior to data collection (1997-2006) was the driest in the 56-year precipitation record (Figure 4-4), with seven of the ten years receiving less than average precipitation. A total of 246 individuals sampled among the 14 plots were established during the same decade. Precipitation during the decade beginning 10 years prior to sampling (1987-1996) was not as dry as the more recent decade (Figure 4-4) and more big sagebrush individuals, a total of 275, were established in that decade. Because the mean age of all individuals sampled in this study was relatively young (mid-teens), the effects of mortality on age class distribution make it difficult to extrapolate decadal patterns further back in time. However, recruitment was likely greater in the decade beginning 20 years prior to data collection for this study as annual precipitation was greater during that time period than either of the two more recent decades.

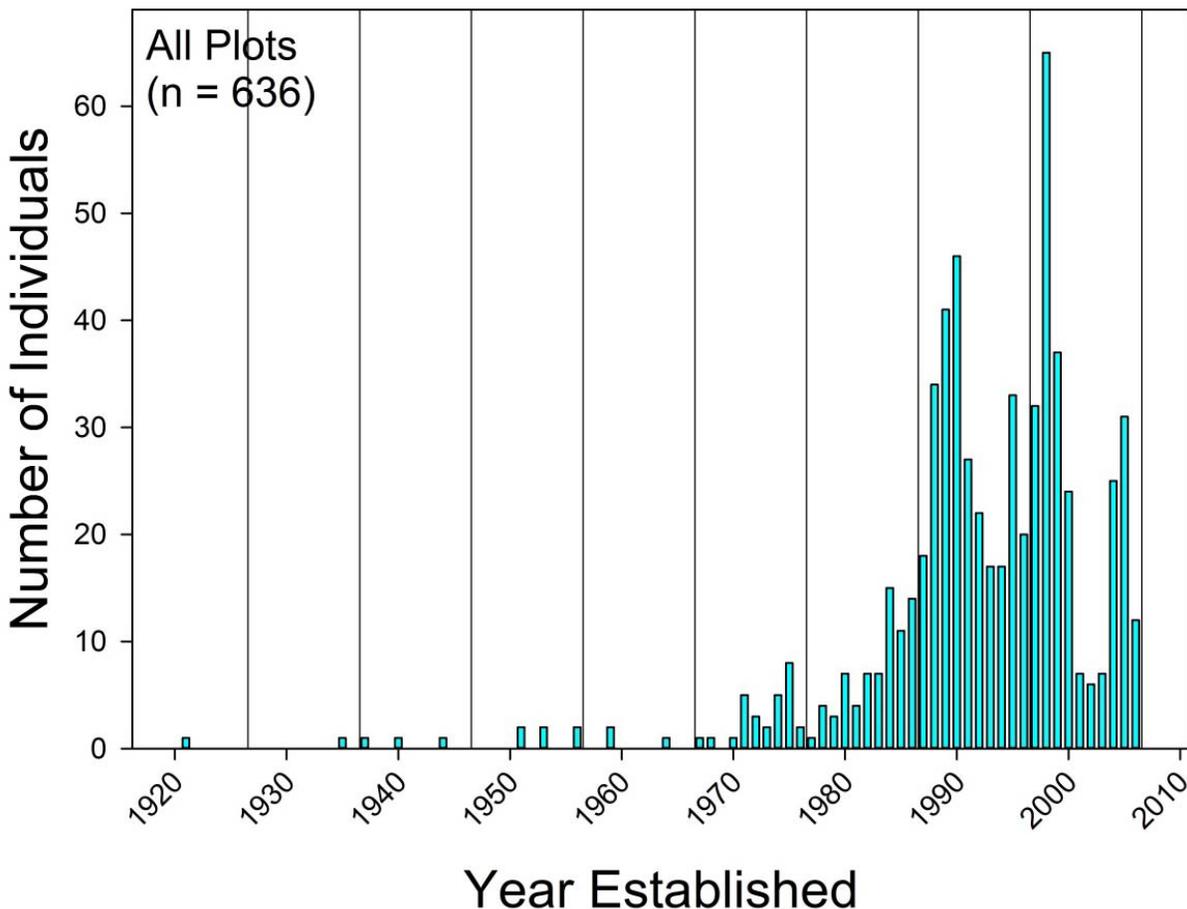


Figure 4-3. Annual age class distributions for 636 big sagebrush sampled on the Idaho National Laboratory Site. Data are all individuals pooled from 14 sample plots.

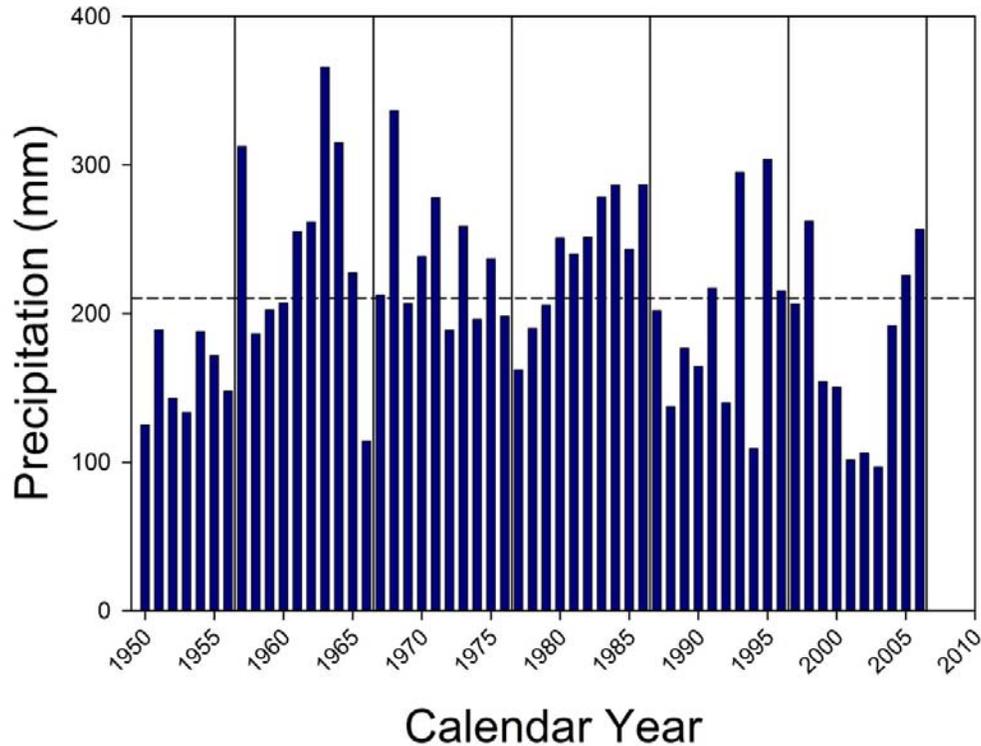


Figure 4-4. Total annual precipitation from the Central Facilities Area at the Idaho National Laboratory Site. The dashed line represents mean annual precipitation.

While there does appear to be some correspondence between gross precipitation patterns and the number of individuals recruited into the big sagebrush population, not all recruitment patterns are directly attributable to annual precipitation trends. For example, the second largest age class in the decade prior to sampling was from 1999 (Figure 4-3), but annual precipitation from 1999 was well below average (Figure 4-4). Alternately, in 2006 annual precipitation totals ranked second highest compared to those of the previous decade (Figure 4-4), but fewer seedlings established that year than in 1999 (Figure 4-3). Seasonal precipitation timing provides a weak explanation at best. Spring precipitation was greater in March, April, and June of 2006 than in those same months of 1999 (Figure 4-5). Furthermore, monthly precipitation totals were well below average in late summer and fall of 1999, a pattern which would be expected to increase juvenile mortality. Seedlings germinating in the spring of 2006 were sampled in June and July, so lack of late season moisture and related juvenile mortality would not have affected this age class. It is apparent from these results that factors other than precipitation affect recruitment. Other environmental variables like temperature or wind probably influence the size of annual age classes, as do more complex and localized concepts like topographical context and the influence of surrounding plant communities.

4.3.2 Stand-level Age Class Distributions

Variability among age distributions by plot, or stand, supports the suggestion that local conditions may contribute to population structure at least as much as precipitation patterns. Stand age distributions for each plot, along with summary statistics and photographs are shown in Appendix D. Age classes are summarized in one-, five-, and ten-year increments. Given the relatively young mean age of the individuals sampled for this study, the ten-year age class increments generally don't provide much information about stand age structure, but are included for comparison because many population studies of woody species are presented in this format. The one year age class increments offer enough detail for understanding annual recruitment patterns and the five year age class increments illustrate meaningful trends in population structure.

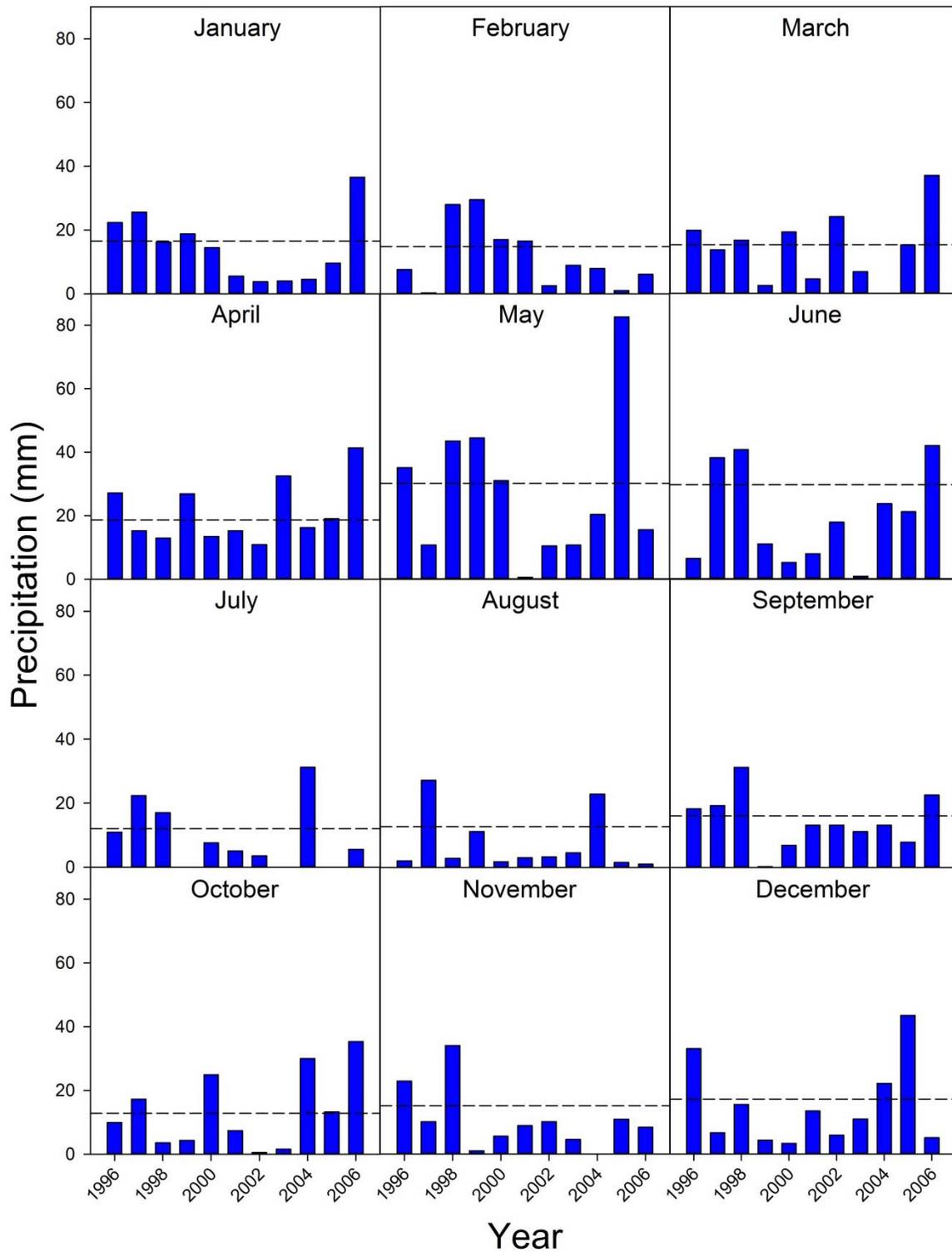


Figure 4-5. Total monthly precipitation from the Central Facilities Area at the Idaho National Laboratory Site from 1996-2006. The dashed line represents mean monthly precipitation.

Mean plot age ranges from 7.3 years for Plot 9 to 22.6 years for Plot 5 (Table 4-2). A visual evaluation of a stand is often used to describe stand health or determine whether a stand is decadent. Variables like big sagebrush cover, density, and the abundance of standing dead shrubs are used as visual cues and guide qualitative interpretations. Although they may be useful for describing the current ecological condition or habitat value of a stand, visual impressions don't appear to be an accurate indicator of population age or structure. Compare the photographs from Plot 10 and Plot 11 (Appendix D), for example. In terms of condition, or "decadence," they appear to be somewhat similar. If either stand was visually assessed as being in better condition than the other, it would likely be Plot 10. However, there were no individuals established in the six years prior to data collection among the shrubs sampled from Plot 10, while individuals recruited in four of the six years prior to this study were represented in the annual age classes in Plot 11 (Appendix D). In addition, the mean age of the shrubs sampled from Plot 10 was more than twice the mean age of shrubs sampled from Plot 11 (Table 4-2).

Table 4-2. Age summary statistics for living and dead big sagebrush sampled from 14 plots on the Idaho National Laboratory Site. Ages were determined using annual ring counts on stem cross-sections. Samples sizes ranged from 28 to 50 live individuals and approximately 20 dead individuals from each plot.

	Mean Age – Live Samples	Minimum Age – Live Samples	Maximum Age – Live Samples	Mean Age – Dead Samples	Minimum Age – Dead Samples	Maximum Age – Dead Samples
Plot 9	7.3	1	26	27.7	7	58
Plot 11	10.4	1	33	30.3	7	64
Plot 13	10.6	2	22	26.4	9	66
Plot 8	11.2	1	43	26.7	9	53
Plot 19	14.0	2	36	25.9	7	46
Plot 18	14.1	2	27	31.1	7	52
Plot 14	14.5	1	35	33.3	10	59
Plot 15	14.7	1	48	34.9	6	78
Plot 7	15.8	5	40	33.2	13	62
Plot 17	16.0	3	54	35.9	9	75
Plot 16	16.9	1	48	38.9	6	83
Plot 4	21.1	7	86	49.4	17	83
Plot 10	21.5	7	67	33.0	12	67
Plot 5	22.6	5	72	37.3	10	65

Coincidentally, Plots 10 and 11 are located within about 500 m of one another, and are in much closer proximity than any other pairwise set of plots sampled (Figure 4-1). Because the two closest plots have very different age distributions, these results provide further evidence that the factors affecting stand age structure are very site-specific. Both of these plots are located on similar substrate, and soils at each location are similar in texture and depth. Precipitation, in amount and timing, is probably also the same for both of these plots as they are located close enough to one another to be exposed to the same weather conditions and storm events, making microclimate and local environmental context important considerations.

All 14 stands exhibited an uneven age structure and we found no evidence of any stand sampled for this study being completely dominated by one age cohort in data summarized by either annual or five year age classes (Appendix D). A range of patterns in age class distribution were represented by the stands sampled in this study. Some stands, like those sampled in Plots 10 and 18, are characterized by relatively flat histograms, with no more than three individuals in any annual age class (Appendix D). Age class distributions of other plots, such as Plots 7 and 11, are typified by histograms with one or more distinct modes (Appendix D). Just as there are appreciable differences in histogram shape among plots, there are also meaningful differences in the number of age classes represented from shrubs sampled in each plot. For example, only 16 annual age classes were represented by the shrubs sampled from Plot 9, while ages from shrubs sampled from Plots 5 and 16 were distributed among 24 annual age classes. Between 47 and 50 live individuals were sampled in Plots 5, 9, and 16 and increased sampling effort would certainly increase the number of age classes represented in each stand, however, stands would still differ in terms of age distribution, and the differences may even be magnified by additional sampling.

Although the overall mean age of shrubs sampled for this study is much younger than we had expected, there is evidence in the stand age structures for populations which are comparatively younger or older and which are either increasing or declining. These trends are most apparent in the age distributions which are summarized by 5 year increments. The plots with the lowest mean age, like Plot 9 and Plot 11, tend to have the largest cohorts in the 0-5 and 6-10 year-old age classes, while plots with the highest mean age, like Plots 5 and 10, tend to have the largest cohorts in the 16-20 and 21-25 year-old age classes (Appendix D). An abundance of individuals in the youngest age classes indicates a population is healthy and likely growing and an abundance of individuals in age classes which are older than the mean age of individuals sampled for this study suggests the populations in those stands are declining. Comparative age, cohort size, and population trend for a stand are often, but not always related. An exception is illustrated in Plot 7, where the largest cohort is in the 6-10 year-old age class, but nearly two thirds of the shrubs sampled from this stand are older than those in the largest age class, signaling that the population status of the stand may be declining more than is suggested by the size of the 6-10 year-old cohort.

Some stands appear to share similar age class distribution patterns. Both Plot 8 and Plot 9 are characterized by population structures with a distinct mode in the youngest age classes and class size decreases as age increases. The five youngest annual age classes of these two plots would even share the same ranks if each were independently ranked by magnitude of class size. Nevertheless, population structures which are common among multiple plots tend to be unusual in this data set (Appendix D). In fact, among the annual age class distributions from the 14 plots, the year with the largest age class is different for all but a couple of plots. Likewise, the “empty” age classes, or age classes which were not represented by any individuals sampled from a given plot, were inconsistent among the 14 plots.

To summarize the results of our analysis of the variability associated with stand age distribution of big sagebrush at the INL Site, population structures of different stands tend to be very site specific. The population status of the 14 plots we sampled ranged from increasing, with an abundance of individuals in young age classes to declining, marked by very few individuals in young age classes. There is certainly some indication that environmental conditions affect population dynamics on a localized scale. Although general precipitation patterns are reflected in some stand age distributions, it appears as though optimal recruitment conditions for one stand are different from another, as reflected in variability in the size, or even presence of a particular class in comparisons of specific annual age classes among the 14 plots. Differences in age class histogram shape among the 14 plots also suggest that recruitment is more episodic in some stands and more gradual in others. While mortality patterns are not easily addressed with this data set, it is likely that mortality patterns and rates differ among stands as well.

4.3.3 Age Class Distributions and Other Stand-level Variables

In addition to age of live shrubs, we sampled age of dead big sagebrush individuals, shrub cover by species, cover by dead shrubs, condition rank of big sagebrush, and big sagebrush density to assess any potential relationships between these measures of stand condition and population status. In particular, we were interested in identifying any

variables which could be used as an indicator of stand age structure or trend in mean age. The stem cross-sections used for ring counts on dead individuals were not cross-dated, so any inferences made about these data are restricted to mean shrub age at death rather than the year or any associated environmental conditions when individuals died. We should also note that the data for dead individuals may not fully represent very young or very old individuals. Young individuals tend to decompose more quickly than older shrubs, which have greater woody biomass, and are probably lost from the stand more quickly. The oldest individuals are more likely to be missing a large section of the inner rings and pith or the lateral branch containing the center of the stem, making them unsuitable for exact ring counts.

The mean age of dead big sagebrush shrubs ranged from 25.9 years on Plot 19 to 49.4 years on Plot 4 (Table 4-2). Just as with the mean age of live shrubs, we were surprised by the relatively young average age at which individuals died. Mean age of dead shrubs from this study was four to five times younger than the maximum reported lifespan of the species (Ferguson 1964). The youngest dead individuals sampled for this study were six years old and the oldest were 83 years old (Table 4-2). Of the 279 dead individuals sampled, the two oldest were still less than half the age of the oldest known individuals (Ferguson 1964) and more than two thirds of the dead shrubs sampled for this study were 40 years old or younger at the time of death. Thus, mortality of big sagebrush at the INL Site cannot be simply attributed to advanced age.

Results of a multiple correlation analysis indicate a significant, positive relationship between mean age of live individuals and mean age of dead individuals in a plot (Table 4-3, Appendix E). In other words, mortality occurs at a younger age in plots which generally have younger age structures. Based on these results, it is reasonable to infer that the rate at which populations turn over can vary substantially from one stand to another, or that population dynamics are site-specific. It is also possible that "old age" is more of a function of stand age structure and rates of change in specific populations than of some physiologically-defined maximum.

Mean age of dead individuals is the only variable with a statistically significant relationship to mean age of the live stand population (Table 4-3). Therefore, we cannot assume that shrub cover, density, or even individual condition is a reliable indicator of the age of a stand. Other significantly positive relationships include those between rabbitbrush cover and mean age of dead individuals, and between rabbitbrush cover and cover by dead shrubs. These correlation results should be interpreted cautiously, however, as they are driven by one plot with anomalously high green rabbitbrush cover (Appendix E). This plot also had a mean age of dead individuals nearly 10 years greater than the plot with the next highest mean and the highest dead shrub cover of all plots sampled. There is also not a significant relationship between mean age of dead individuals and cover by dead shrubs (Table 4-3).

There are significant positive correlations between all pairwise combinations of three additional variables, mean big sagebrush cover, mean big sagebrush density, and mean condition rank of big sagebrush (Table 4-3). The relationship between cover and density is not surprising, because greater numbers of individuals, especially mature shrubs, often result in higher mean cover. The positive relationship between condition rank and the two measures of big sagebrush abundance is also somewhat intuitive, as the average amount of living biomass on shrubs in a stand should be reflected in abundance values.

Table 4-3. Results of a multiple correlation analysis comparing several indicators of population and ecological status on plots established in 14 big sagebrush stands at the Idaho National Laboratory Site.

	Mean Age Dead	Big Sagebrush Cover	Green Rabbitbrush Cover	Dead Shrub Cover	Mean Condition Rank	Mean Density
Mean Age Live	r = 0.72 p < 0.01 n = 14	r = -0.35 p = 0.22 n = 14	r = 0.23 p = 0.43 n = 14	r = 0.12 p = 0.69 n = 14	r = -0.17 p = 0.57 n = 14	r = -0.33 p = 0.25 n = 14
Mean Age Dead		r = -0.35 p = 0.22 n = 14	r = 0.65 p = 0.01 n = 14	r = 0.31 p = 0.28 n = 14	r = -0.19 p = 0.51 n = 14	r = -0.22 p = 0.45 n = 14
Big Sagebrush Cover			r = -0.24 p = 0.41 n = 14	r = -0.23 p = 0.43 n = 14	r = 0.87 p < 0.01 n = 14	r = 0.66 p = 0.01 n = 14
Green Rabbitbrush Cover				r = 0.54 p = 0.05 n = 14	r = -0.33 p = 0.24 n = 14	r = -0.37 p = 0.19 n = 14
Dead Shrub Cover					r = -0.39 p = 0.17 n = 14	r = -0.09 p = 0.76 n = 14
Mean Condition Rank						r = 0.73 p < 0.01 n = 14

We originally recorded big sagebrush condition rank because we thought it may provide an acceptable indicator of the mean age or population structure of a stand, but results of the multiple correlation analysis failed to support a relationship between mean plot age and mean condition rank. Nevertheless, condition rank does provide some insight into stand condition and population dynamics, particularly mortality. Only four of the 14 plots sampled had a mean condition rank of “3” or greater. Recall that about one half of the canopy of a big sagebrush shrub must be alive to be assigned a rank of “3” and ranks lower than “3” indicate that less than half of a shrub’s canopy is alive. Therefore, on average more than one half of the canopy of the individuals sampled was dead in all but four stands. Mean condition rank ranged from 2.2 on Plot 14 to 3.5 on Plot 7. When averaged across all plots, 47 of 100 randomly sampled individuals were standing dead shrubs (Figure 4-6) and only about 30 individuals were fully alive. The extent of standing dead individuals was not always apparent in visual assessments of the plots (Appendix D), and in many stands it was probably veiled by cover of live biomass from all species. There are a few possible explanations for the relatively high ratio of dead to live individuals. Standing dead shrubs may persist for many years in a stand due to slow rates of decay, or mortality rates may be higher than would be expected for a long-lived, woody species. These possibilities are not mutually exclusive and both likely factor into the abundance of standing dead shrubs.

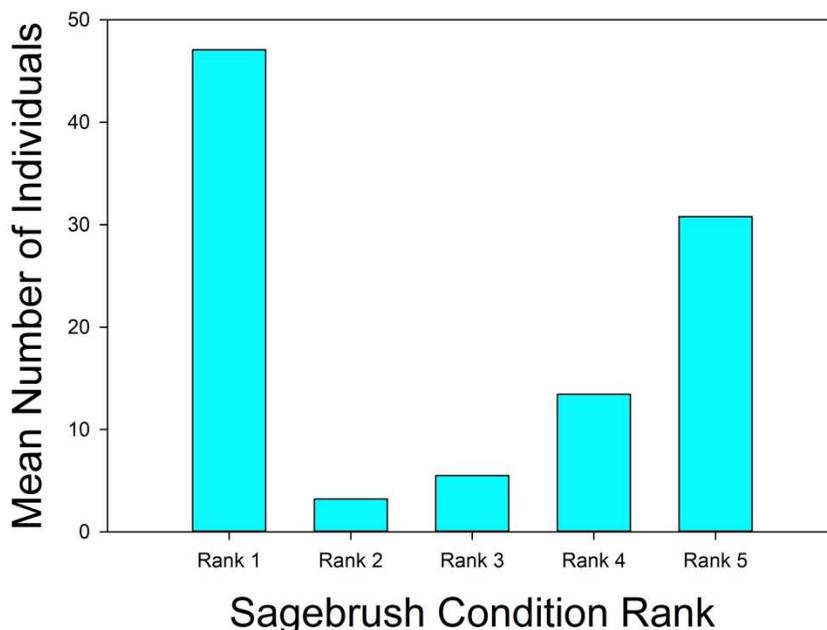


Figure 4-6. Condition ranks for big sagebrush averaged across 14 plots from the INL Site.

4.4 Discussion

4.4.1 By Objective

4.4.1.1 *Objective 1: To determine the typical stand age structure or range of stand age structures for mature big sagebrush stands.*

Big sagebrush is generally considered a long-lived species, an assumption which is probably based on early demographic literature (e.g. Blaisdell 1953, Ferguson 1964). More recent research suggests that mean stand age is somewhat younger than previously thought (Cawker 1980, Perryman et al. 2001), but mean big sagebrush age from stands sampled for this study are very young even within the context of these more recent studies. Perryman et al. (2001) sampled nine Wyoming big sagebrush stands in Wyoming and the stand with the youngest mean age was approximately equivalent to the three oldest stands of the 14 sampled from the INL Site (about 21 years). Age estimates from this study are consistent with other work conducted locally. Colket (2003) estimated big sagebrush age on seven undisturbed stands at the INL Site with results very similar to those of this effort. Results of demography work over the past decade and particularly from this study suggest that big sagebrush may in fact be very short-lived when compared to other woody species.

All 14 stands sampled for this study exhibited uneven-aged populations; they were represented by cohorts from multiple age classes. Although a few investigators have reported even-aged populations in big sagebrush (Bartolome and Heady 1978), most have found that undisturbed stands are typically uneven-aged (Passey and Hughie 1962, Ferguson 1964, Roughton 1972, Cawker 1980). Even-aged population structures are typical of species characterized by very infrequent recruitment episodes, often related to a catastrophic disturbance or another major event. The population age structures of stands sampled for this study are indicative of frequent recruitment, the magnitude of which is annually variable.

In addition to temporal variability in recruitment, spatial variability in recruitment is evident among stands and likely results in population age structures which differ at fine spatial scales across the INL Site. Within a limited spatial range of about 15 km, stand age structure ranges from populations with an abundance of individuals in young age

classes to those with populations dominated by individuals in comparatively older age classes. Mechanisms beyond just variability in recruitment have been used to explain variability in population dynamics from one stand to another including differences in microclimate conditions (Ferguson 1964), differences in seed production (Evans and Black 1993) and differential survival of young and mature individuals under conditions of climatic stress (Campbell and Harris 1977). The close proximity of the stands sampled in this study suggest that population parameters affecting stand age distribution, like natality and mortality are probably highly influenced by microclimate and other site-specific conditions because all plots were located closely enough to experience similar weather conditions.

Stable age distributions are those in which population size is either stationary or changing at a constant rate over several generations. Stochasticity generally prevents long-term stability in natural populations (Miller 1976). Therefore, the population age patterns and dynamics resulting from this study are probably not static. In fact, some authors describe a cyclic pattern in big sagebrush stand condition, where a stand may transition from “thick and thrifty” to “decadent,” and eventually begin regenerating itself (Lomasson 1948, West 1984). Though we were unable to demonstrate a simple, linear relationship between mean stand age and stand condition, as measured by the amount of standing dead biomass, it is entirely possible that both stand condition and age structure are cyclic, but decoupled due to the persistence of standing dead individuals in the canopy for long periods of time.

Management actions which aim to improve decadent big sagebrush stands, such as chemical applications, mechanical treatments, and prescribed fire are often unnecessary, unsuccessful, or cause negative impacts to the native sagebrush steppe plant community (Rhodes et al. 2010, Ripplinger 2010). These manipulative strategies may not be successful because they are based on management techniques used in forested ecosystems (Welch 2005). With mean stand ages of 20 years or less, big sagebrush shrublands are not particularly analogous to tree-dominated forests. Furthermore turnover rates of mature, undisturbed stands are likely much higher than previously thought. Developing more effective big sagebrush management strategies will require moving away from techniques based on the population dynamics of other woody dominants and assessing big sagebrush as a shorter-lived perennial species with unique population characteristics.

4.4.1.2 Objective 2: To investigate how stand age structure relates to shrub die-off and stand condition for big sagebrush.

Because stand turnover occurs much more quickly than we had initially hypothesized, current big sagebrush abundance values are probably not a direct result of the die-off reported from the 1970s. The die-off may still be affecting population sizes through indirect effects like an overall reduction in seed production due to fewer mature individuals in a stand. Most of the general decline in abundance over the past few decades, however, is more likely a matter of reduced recruitment and increased mortality or mortality at younger ages. Trends in age class size reflect trends in annual precipitation in this study, a pattern which others have described as well (Maier et al. 2001). Over the past two decades annual precipitation at the INL Site has been above average in only about five years. Some of the driest years on record were also reported during this same time period. If hotter, drier weather patterns persist in the future it is not unreasonable to predict continued decreases in big sagebrush population sizes and related abundance values.

In the younger age classes the relationship between precipitation and age class size is easily attributed to decreased recruitment under unfavorable conditions (Meyer 1994), but weather patterns are also likely to affect mortality rates. Results from this study suggest that big sagebrush shrubs are not dying from “old age” as the mean age at death on the INL Site is much younger than the oldest reported individuals across the region (Ferguson 1964). Characterizing natality and mortality rates for big sagebrush is beyond the scope of this study, but we can address recruitment to some extent because aging live individuals allows us to determine the year in which they established. Mortality, on the other hand, is especially difficult to assess because aging dead shrubs only allows us to determine the age at death, not necessarily the year they died. In general, mortality rates and patterns are difficult to interpret from static age structures (Johnson et al. 1994). Though it does seem rational to conclude that because individuals are dying at a fraction of the maximum reported age for the species, mortality is probably affected by precipitation patterns as well

as by interactions between precipitation and other stressors. Herbivory by small mammals (Frischnecht and Baker 1972), defoliating moths (Gates 1964), and beetles, as well as fungal diseases like snowmold and black stem rust (Sturges and Nelson 1984) have all been shown to lower the vigor of individual shrubs leading to increased mortality throughout big sagebrush stands (Parmenter et al. 1987).

One of the specific goals of this study was to identify a measure of stand condition which could be easily sampled and is representative of mean stand age so that stand age could potentially be estimated without the effort of destructive sampling and counting rings. Unfortunately none of the variables measured in this study were a reliable surrogate for mean stand age. Often our general visual interpretation of the condition of a stand was not even indicative of actual stand age, an observation which is supported by a lack of correlation between mean condition rank and mean stand age. Just as mean stand condition rank was not directly related to mean stand age, neither were big sagebrush abundance (density or cover), nor were the cover of other shrubs in a stand, particularly green rabbitbrush. Even though we were unable to identify a measure of stand condition which is predictive of age, we are able to conclude that management actions proposed for big sagebrush stands should never be based solely on visual evaluation of health, condition, or amount of decadence because some stands which appear decadent actually have an abundance of individuals in younger age classes. From a sagebrush steppe management perspective, if mortality is generally not from old age and the appearance of decadence is not a result of an overabundance of individuals which are no longer reproducing, removal of mature individuals based on that premise would be an ineffective strategy for manipulating stand condition. Furthermore, based on the results of this study the concepts of “stand condition” and “decadence” as they relate to population age, are not particularly meaningful and an abundance of standing dead individuals is typical for all mature stands, regardless of population status.

4.4.1.3 Objective 3: To examine the dynamics of sagebrush stand replacement in the absence of wildland fire.

Wambolt et al. (2001) found that at least some big sagebrush recruitment occurs every year in post-fire communities. Our results indicate that the same is true of undisturbed, mature big sagebrush stands at the INL Site, where individuals established even during the driest year on record. Although at least some big sagebrush shrubs were recruited across the study site in all years, they were not recruited in every stand each year. Based on our results, mechanisms controlling stand replacement which include mortality, but are particularly relevant to recruitment, appear to be related to a combination of general precipitation patterns, and fine-scale microsite conditions. Generalized across the study site, annual recruitment patterns are cyclic and patterns in annual age class size reflect patterns in annual precipitation, but recruitment in some stands appears to be more affected by annual precipitation than in others. Site-specific environmental factors, like soil moisture holding capacity, slope and aspect, and total vegetation cover may moderate or alter the effects of precipitation on recruitment among various stands.

The relative importance of different factors affecting stand replacement (i.e. regional precipitation patterns as well as local environmental conditions) probably alter population growth parameters, including recruitment rates, mean population longevity, and mortality patterns and result in turnover dynamics which are very localized. Colket (2003) also found that big sagebrush population characteristics on the INL Site were site specific. The localized differences in turnover and the interactions of complex site-specific mechanisms controlling stand replacement at the INL Site makes identifying mature stands which may benefit from manipulative management very difficult. For the same reason, it would be challenging to predict the outcome of mechanical, chemical or prescribed fire treatments on a “decadent” stand. This issue is exacerbated because turnover rates for big sagebrush stands sampled for this study were much higher than expected, which may make population structure more responsive to annual weather patterns and ultimately more vulnerable to climate change.

Big sagebrush stands are often manipulated using disturbances which are supposed to reduce the abundance of older, “decadent” individuals and create space for younger, “healthier” shrubs (e.g. Bunting et al. 1987). Our results suggest that disturbance is not required for stand replacement. In fact, all stands sampled for this project had a mean age of living individuals of less than 25 years and a mean age of individuals at death of less than 50 years.

Stands chosen for this study were selected because they had not experienced any known catastrophic disturbance within at least the past century, so their big sagebrush populations have turned over several generations within that time period. A number of other investigators have also concluded that sagebrush stand can maintain themselves indefinitely in the absence of fire or other disturbance (Lommasson 1948, Passey and Hughie 1962, Wambolt and Hoffman 2001), further questioning the necessity for creating disturbance in the name of stand health.

4.4.2 Synthesis

Prior to data collection for this study, we anticipated using population structures reported from old-growth forests as an analogous framework for understanding population dynamics of mature big sagebrush stands. We knew that mean stand age would be younger and rates of change in population structure would occur more quickly for big sagebrush than for a typical old-growth species like Douglas fir (*Pseudotsuga menziesii*), but we theorized patterns of stand replacement may be similar. The results from this study show that mean big sagebrush age at the INL Site is much younger than expected and recruitment patterns are so variable in both space and time that interpretations of big sagebrush stand age distributions within the context of old-growth models are not meaningful. In fact the results of this study have prompted us to reconsider many of the conventional assumptions made about the biology of big sagebrush.

Big sagebrush has long been described and managed as a slow-growing, long-lived species (e.g. Tisdale and Hironaka 1981), but many of the life history characteristics, or strategies, of the species are not necessarily consistent with those of a typical K-adapted species. Grime (1977) classified arid plant species which are typified as being evergreen sclerophylls, with slow-growth rates and limited nutrient turnover as stress tolerant species, which occupy the K end of the r-K continuum. This view may be overly simplistic though, because big sagebrush also tends to become sexually mature within a few years (Young et al. 1989), produces an abundance of small seeds (Young et al. 1989), and isn't necessarily highly competitive (Anderson and Inouye 2001, Welch 2005), all of which are traits commonly attributed to r-strategists (Pianka 1972). Results from this study also suggest that adult mortality is related to herbivory, disease, and weather conditions more than to cover, density, or stand condition; which point to density independence – another trait of r-adapted species (Pianka 1972).

In their original discussion of r- and K-selection, MacArthur and Wilson (1967) describe an r-selected species as one that maximizes productivity rather than efficiency. Although it is capable of withstanding extended drought periods, the resource-acquisition strategy of big sagebrush favors maximizing assimilation of carbon and mineral nutrients under favorable conditions with little regard to efficiency (DeLucia and Schlesinger 1991). In terms of its resource-acquisition strategy, it is particularly adept at achieving impressive photosynthetic rates under optimal conditions, which occur only sporadically in their semi-arid habitats (Smith et al. 1997). We found that recruitment patterns, particularly relationships between age class size and favorable precipitation and/or microsite conditions, mirror resource utilization patterns in big sagebrush. In an effort to better understand concepts of r- and K- selection within the context of plant ecology, Gadgil and Solbrig (1972) determined that an r-selected species need not be an annual, a colonizing species, or even herbaceous, but a common characteristic among r-selected species is that they often result from environments with discontinuously available resources, like those where big sagebrush dominate the plant community.

If big sagebrush functions more like an r-adapted species than previously considered, as is implied by stand turnover rates documented by this study, then an obvious question becomes: why do sagebrush stands take up to 100 years to recover post-burn at the INL Site? Prevey et al. (2010) have recently described big sagebrush as a foundation species, which is a dominant plant species exhibiting disproportionate control over ecosystem processes. In addition to being described as a foundation species, big sagebrush can also be considered an ecosystem engineer. Specifically, an autogenic engineer has been defined as an organism which changes the local environment through its physical structures (Jones et al. 1994). As an ecosystem engineer, sagebrush is capable of modulating microsite conditions such as humidity, temperature, wind speed, soil stability, and light levels within a stand. It also regulates the spatial distribution of resources including nutrients (Bechtold and Inouye 2007) and soil moisture availability

(Richards and Caldwell 1987). Therefore, big sagebrush germination and establishment are probably facilitated by the amelioration of microsite conditions by mature big sagebrush. Other studies from the INL Site have demonstrated that during dry years very few microsites favorable for big sagebrush recruitment are available on burn scars (Blew and Forman 2010) and elsewhere, researchers have found greater recruitment in unburned sites compared when burned sites (Wambolt et al. 2001). Basically, prolonged recovery times of big sagebrush stands post disturbance are more likely a function of big sagebrush not being particularly well adapted to fire, or to the amount of environmental exposure in post-fire communities, than to it being a K-adapted species.

Results of our investigation into big sagebrush population dynamics, especially when interpreted within the context of other aspects of big sagebrush biology provide some insights into the characteristics of the species which differ from conventional wisdom. Much of the discussion provided here reaches beyond the conclusions which can be drawn from data collected for this study, but many of these hypotheses could be tested with empirical data. Reconsidering the biology of this species from a different perspective may change the approach to conservation and management of the sagebrush steppe ecosystem.

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APPENDIX A

SUMMARY STATISTICS TABLES, ANOVA RESULTS TABLES (CHAPTER 2)

Table A-1a. Estimates of mean cover for native perennial functional groups from the line-interception data on the “core” Long-Term Vegetation Transect plots at the Idaho National Laboratory from 1950 to 2011. Minimum significant difference indicates the value at which means of pairwise comparisons between two sample years within each functional group becomes significant.

Total		Shrub		Grass	
Year	Mean	Year	Mean	Year	Mean
1950	16.91	1950	16.41	1950	0.50
1957	17.89	1957	16.88	1957	1.01
1965	25.39	1965	23.88	1965	1.51
1975	30.56	1975	23.89	1975	6.68
1978	26.39	1978	23.10	1978	3.30
1983	20.50	1983	17.45	1983	3.05
1985	23.22	1985	21.66	1985	1.57
1990	22.60	1990	18.61	1990	3.99
1995	24.86	1995	21.91	1995	2.95
2001	19.86	2001	17.39	2001	2.47
2006	22.16	2006	17.67	2006	4.49
2011	22.98	2011	19.33	2011	3.64
Minimum Significant Difference	6.48		6.48		1.90

Table A-1b. Results tables for one-way repeated measures ANOVAs comparing mean cover for native perennial functional groups from the line-interception data on the “core” Long-Term Vegetation Transect plots at the Idaho National Laboratory from 1950 to 2011.

Total, One-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	42	28018.16	667.10		
Year	11	6639.94	603.63	8.18	<0.001

Shrub, One-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	42	33209.61	790.71		
Year	11	3901.22	354.66	4.56	<0.001

Grass, One-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	42	1392.25	33.15		
Year	11	1378.70	125.34	19.13	<0.001

Table A-2a. Estimates of mean cover for *Artemisia tridentata* (ARTR) and *Chrysothamnus viscidiflorus* (CHVI) from the line-interception data on the “core” Long-Term Vegetation Transect plots at the Idaho National Laboratory from 1950 to 2011. Minimum significant difference indicates the value at which means of pairwise comparisons between two sample years within each species becomes significant. The significance column indicates whether the mean cover difference between each species was significant for a given sample period.

Year	ARTR	CHVI	Significant
1950	16.57	0.99	Yes
1957	15.90	1.15	Yes
1965	22.90	2.37	Yes
1975	21.68	3.48	Yes
1978	18.40	3.70	Yes
1983	13.48	2.98	Yes
1985	16.24	5.76	Yes
1990	11.79	4.69	Yes
1995	13.63	7.61	Yes
2001	8.88	6.38	No
2006	8.33	7.43	No
2011	10.12	7.05	No
Minimum Significant Difference	5.11	5.39	

Table A-2b. Results table for two-way repeated measures ANOVA comparing mean cover for *Artemisia tridentata* (ARTR) and *Chrysothamnus viscidiflorus* (CHVI) from the line-interception data on the “core” Long-Term Vegetation Transect plots at the Idaho National Laboratory from 1950 to 2011.

Two-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	32	12599.31	393.73	0.85	0.68
Species	1	21262.73	21262.73	43.75	<0.001
Species x Plot	32	15550.64	485.96		
Year	11	2414.58	219.51	7.03	<0.001
Year x Plot	352	10996.31	31.24		
Species x Year	11	7660.25	696.39	13.49	<0.001

Table A-3a. Estimates of mean cover for native perennial bunch grasses and native perennial rhizomatous grasses from the line-interception data on the “core” Long-Term Vegetation Transect plots at the Idaho National Laboratory from 1950 to 2011. Minimum significant difference indicates the value at which means of pairwise comparisons between two sample years within each functional group becomes significant. The significance column indicates whether the mean cover difference between each functional group was significant for a given sample period.

Year	Bunch	Rhizomatous	Significant
1950	0.27	0.07	No
1957	0.60	0.20	No
1965	0.78	0.37	No
1975	4.11	1.46	Yes
1978	1.92	1.41	No
1983	1.54	1.51	No
1985	0.90	0.47	No
1990	2.64	1.44	Yes
1995	2.47	0.47	Yes
2001	2.07	0.58	Yes
2006	3.95	0.70	Yes
2011	2.90	0.94	Yes
Minimum Significant Difference	1.31	1.39	

Table A-3b. Results table for two-way repeated measures ANOVA comparing mean cover for native perennial bunch grasses and native perennial rhizomatous grasses from the line-interception data on the “core” Long-Term Vegetation Transect plots at the Idaho National Laboratory from 1950 to 2011.

Two-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	32	524.26	16.38	1.077	0.42
Growth Form	1	290.22	290.22	19.19	<0.001
Growth Form x Plot	32	483.92	15.12		
Year	11	479.15	43.56	15.53	<0.001
Year x Plot	352	987.43	2.81		
Growth Form x Year	11	203.26	18.48	6.82	<0.001

Table A-4a. Estimates of mean cover for native perennial grasses and introduced perennial grasses from the line-interception data on the “core” Long-Term Vegetation Transect plots at the Idaho National Laboratory from 1950 to 2011. Minimum significant difference indicates the value at which means of pairwise comparisons between two sample years within each functional group becomes significant. The significance column indicates whether the mean cover difference between each functional group was significant for a given sample period.

Year	Native	Introduced	Significant
1957	1.01	0.00	No
1965	1.51	0.02	Yes
1975	6.68	0.00	Yes
1985	1.57	0.03	Yes
1995	2.95	0.31	Yes
2001	2.47	0.55	Yes
2006	4.49	0.77	Yes
2011	3.64	1.00	Yes
Minimum Significant Difference	1.38	N/A	

Table A-4b. Results table for two-way repeated measures ANOVA comparing mean cover for native perennial grasses and introduced perennial grasses from the line-interception data on the “core” Long-Term Vegetation Transect plots at the Idaho National Laboratory from 1950 to 2011.

Two-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	42	622.72	14.83		
Nativity	1	1259.56	1259.56	55.27	<0.001
Nativity x Plot	42	957.09	22.79		
Year	7	610.3	87.20	23.03	<0.001
Year x Plot	294	1113.11	3.79		
Nativity x Year	7	497.44	71.06	13.62	<0.001

Table A-5a. Estimates of mean density and frequency for *Bromus tectorum* from the density/frequency data on the all of the Long-Term Vegetation Transect plots for which data were available in each sample year at the Idaho National Laboratory from 1950 to 2006. Minimum significant difference indicates the value at which means of pairwise comparisons between two sample years within either density or frequency becomes significant. *Frequency data were not available for 1995.

Density		Frequency	
Year	Mean	Year	Mean
1950	44.08	1950	7.63
1957	21.85	1957	4.78
1965	33.01	1965	6.88
1975	27.07	1975	13.65
1985	9.04	1985	7.14
1995	42.14	1995	*
2001	88.73	2001	26.08
2006	56.99	2006	19.33
2011	209.49	2011	31.69
Minimum Significant Difference	120.76		11.70

Table A-5b. Results tables for one-way ANOVAs comparing mean density among sample periods and mean frequency among sample periods for *Bromus tectorum*. Data are from all of the Long-Term Vegetation Transect plots for which data were available in each sample year at the Idaho National Laboratory from 1950 to 2011.

Density, One-way ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Year	8	2636099.22	329512.40	9.89	<0.001

Frequency, One-way ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Year	7	62613.95	8944.85	14.00	<0.001

Table A-6a. Estimates of mean density and frequency for introduced annual forbs from the density/frequency data on the all of the Long-Term Vegetation Transect plots for which data were available in each sample year at the Idaho National Laboratory from 1950 to 2011. Minimum significant difference indicates the value at which means of pairwise comparisons between two sample years within either density or frequency becomes significant. *Frequency data were not available for 1995.

Density		Frequency	
Year	Mean	Year	Mean
1950	14.15	1950	3.40
1957	3.90	1957	5.61
1965	6.68	1965	3.23
1975	1.68	1975	2.25
1985	1.44	1985	1.63
1995	9.91	1995	*
2001	43.92	2001	15.11
2006	120.22	2006	22.53
2011	339.29	2011	27.98
Minimum Significant Difference	118.55		9.50

Table A-6b. Results tables for one-way ANOVAs comparing mean density among sample periods and mean frequency among sample periods for introduced annual forbs. Data are from all of the Long-Term Vegetation Transect plots for which data were available in each sample year at the Idaho National Laboratory from 1950 to 2011.

Density, One-way ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Year	8	8827887.77	1103485.97	17.47	<0.001

Frequency, One-way ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Year	7	65970.14	9424.31	21.36	<0.001

Table A-7a. Estimates of mean density and frequency for native annual forbs from the density/frequency data on the all of the Long-Term Vegetation Transect plots for which data were available in each sample year at the Idaho National Laboratory from 1950 to 2011. Minimum significant difference indicates the value at which means of pairwise comparisons between two sample years within either density or frequency becomes significant. *Frequency data were not available for 1995.

Density		Frequency	
Year	Mean	Year	Mean
1950	3.70	1950	7.24
1957	9.79	1957	12.56
1965	8.02	1965	10.59
1975	33.31	1975	44.83
1985	13.72	1985	11.01
1995	71.72	1995	*
2001	15.72	2001	19.261
2006	30.58	2006	25.449
2011	26.82	2011	18.71
Minimum Significant Difference			11.46

Table A-7b. Results tables for one-way ANOVAs comparing mean density among sample periods and mean frequency among sample periods for native annual forbs. Data are from all of the Long-Term Vegetation Transect plots for which data were available in each sample year at the Idaho National Laboratory from 1950 to 2011.

Density, One-way ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Year	8	302208.39	37776.05	23.71	<0.001

Frequency, One-way ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Year	7	89855.63	12836.52	32.28	<0.001

APPENDIX B

ANOVA RESULTS TABLES (CHAPTER 3)

Results tables for one-way repeated measures ANOVAs comparing mean cover for vegetation functional groups before and after the T-17 Fire. Data are from 11 Long-Term Vegetation Transect plots at the Idaho National Laboratory Site.

Native Shrubs, One-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	10	124.39	12.44		
Year	1	1696.12	1696.12	136.13	<0.001

Native Perennial Graminoids, One-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	10	170.21	17.02		
Year	1	18.42	18.42	3.76	0.081

Native Perennial Forbs, One-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	10	34.54	3.45		
Year	1	4.00	4.00	7.88	0.019

Native Succulents, One-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	10	0.17	0.02		
Year	1	0.09	0.09	11.85	0.006

Native Annuals and Biennials, One-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	10	0.59	0.06		
Year	1	0.12	0.12	1.82	0.207

Introduced Annuals and Biennials, One-way Repeated Measures ANOVA Results					
Source of Variation	DF	SS	MS	F	P
Plot	10	973.15	97.31		
Year	1	715.35	715.35	8.04	0.018

APPENDIX C

PLOT COVER SUMMARIES (CHAPTER 3)

Absolute cover by species summaries from point-interception data on 11 Long-Term Vegetation Transect plots at the Idaho National Laboratory Site. Data were collected a few weeks prior to the T-17 Fire and again one year later.

Plot 36

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	7.50	0.00
<i>Chrysothamnus viscidiflorus</i>	5.11	0.44
<i>Ericameria nana</i>	0.33	0.22
<i>Gutierrezia sarothrae</i>	0.00	0.11
<i>Linanthus pungens</i>	0.17	0.00
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	1.72	0.56
<i>Elymus elymoides</i>	1.67	1.00
<i>Elymus lanceolatus</i>	5.56	8.22
<i>Hesperostipa comata</i>	0.33	0.61
<i>Poa secunda</i>	0.00	0.17
Native Perennial Forbs		
<i>Astragalus calycosus</i>	0.06	0.06
<i>Lomatium dissectum</i>	0.33	0.17
<i>Machaeranthera canescens</i>	0.06	0.06
<i>Phlox hoodii</i>	0.17	0.00
<i>Schoenocrambe linifolia</i>	0.11	0.06
Native Succulents		
<i>Opuntia polyacantha</i>	0.11	0.00
Native Annuals and Biennials		
<i>Chenopodium leptophyllum</i>	0.00	0.06
Total Native Cover	23.23	11.74
Introduced Annuals and Biennials		
<i>Alyssum desertorum</i>	17.33	1.00
<i>Bromus tectorum</i>	6.67	0.39
<i>Tragopogon dubius</i>	0.00	0.06
Total Introduced Cover	24.00	1.45
Total Vegetative Cover	47.23	13.19

Plot 37

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	11.28	0.00
<i>Chrysothamnus viscidiflorus</i>	6.06	1.78
<i>Tetradymia canescens</i>	2.33	0.56
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	1.61	0.00
<i>Elymus elymoides</i>	0.33	0.00
<i>Elymus lanceolatus</i>	2.28	3.50
<i>Hesperostipa comata</i>	5.11	0.28
Native Perennial Forbs		
<i>Astragalus filipes</i>	0.33	0.06
<i>Erigeron pumilus</i>	0.11	0.00
<i>Machaeranthera canescens</i>	0.06	0.00
<i>Phacelia hastata</i>	0.06	0.00
Native Succulents		
<i>Opuntia polyacantha</i>	0.06	0.00
Native Annuals and Biennials		
<i>Cryptantha scoparia</i>	0.22	0.00
<i>Descurainia pinnata</i>	0.17	0.00
<i>Gayophytum diffusum</i>	0.17	0.00
<i>Mentzelia albicaulis</i>	0.11	0.00
Total Native Cover	30.29	6.18
Introduced Annuals and Biennials		
<i>Alyssum desertorum</i>	0.78	0.00
<i>Bromus tectorum</i>	7.22	0.00
Total Introduced Cover	8.00	0.00
Total Vegetative Cover	38.29	6.18

Plot 38

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	11.06	0.00
<i>Ericameria nana</i>	0.50	0.00
<i>Ericameria nauseosa</i>	2.50	0.00
<i>Grayia spinosa</i>	1.28	0.00
<i>Linanthus pungens</i>	3.28	0.06
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	0.11	0.00
<i>Elymus elymoides</i>	0.83	0.56
<i>Elymus lanceolatus</i>	1.00	1.78
<i>Hesperostipa comata</i>	4.50	0.89
<i>Poa secunda</i>	0.06	0.89
Native Perennial Forbs		
<i>Allium textile</i>	0.17	0.56
<i>Castilleja angustifolia</i>	0.78	0.00
<i>Lomatium dissectum</i>	0.00	0.11
<i>Phlox hoodii</i>	0.06	0.00
<i>Schoenocrambe linifolia</i>	0.00	0.11
Native Succulents		
<i>Opuntia polyacantha</i>	0.06	0.00
Native Annuals and Biennials		
<i>Cordylanthus ramosus</i>	0.78	0.00
<i>Mentzelia albicaulis</i>	0.00	0.17
Total Native Cover	26.97	5.13
Introduced Annuals and Biennials		
<i>Alyssum desertorum</i>	2.83	0.17
<i>Bromus tectorum</i>	15.17	1.28
<i>Descurainia sophia</i>	0.06	0.00
Total Introduced Cover	18.06	1.45
Total Vegetative Cover	45.03	6.58

Plot 39

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	6.61	0.00
<i>Chrysothamnus viscidiflorus</i>	4.56	0.50
<i>Gutierrezia sarothrae</i>	0.72	0.11
<i>Linanthus pungens</i>	0.22	0.00
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	0.11	0.28
<i>Elymus elymoides</i>	0.11	0.06
<i>Elymus lanceolatus</i>	0.61	2.33
<i>Hesperostipa comata</i>	0.83	0.56
<i>Poa secunda</i>	0.72	0.06
Native Perennial Forbs		
<i>Astragalus calycosus</i>	0.00	0.06
<i>Astragalus filipes</i>	0.22	0.06
<i>Astragalus lentiginosus</i>	0.44	0.00
<i>Astragalus purshii</i>	0.06	0.00
<i>Eriogonum ovalifolium</i>	0.72	0.11
<i>Machaeranthera canescens</i>	0.11	0.00
<i>Schoenocrambe linifolia</i>	0.06	0.06
Native Succulents		
<i>Opuntia polyacantha</i>	0.11	0.06
Native Annuals and Biennials		
<i>Aliciella leptomeria</i>	0.00	0.06
<i>Descurainia pinnata</i>	0.00	0.06
<i>Mentzelia albicaulis</i>	0.00	0.17
Total Native Cover	16.21	4.54
Introduced Annuals and Biennials		
<i>Alyssum desertorum</i>	15.06	0.28
<i>Bromus tectorum</i>	15.39	0.17
<i>Tragopogon dubius</i>	0.00	0.28
Total Introduced Cover	30.45	0.73
Total Vegetative Cover	46.66	5.27

Plot 40

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	28.33	0.00
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	1.67	1.17
<i>Elymus elymoides</i>	0.06	0.00
<i>Elymus lanceolatus</i>	0.17	0.00
<i>Hesperostipa comata</i>	3.22	0.17
Native Perennial Forbs		
<i>Machaeranthera canescens</i>	0.50	0.00
<i>Phacelia hastata</i>	0.11	0.06
<i>Psoraleidium lanceolatum</i>	2.50	2.89
<i>Schoenocrambe linifolia</i>	0.06	0.00
Native Annuals and Biennials		
<i>Descurainia pinnata</i>	0.11	0.00
Total Native Cover	36.73	4.29
Introduced Annuals and Biennials		
<i>Alyssum desertorum</i>	0.06	0.00
Total Introduced Cover	0.06	0.00
Total Vegetative Cover	36.79	4.29

Plot 41

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	17.00	0.67
<i>Chrysothamnus viscidiflorus</i>	1.39	0.00
<i>Ericameria nauseosa</i>	0.56	0.00
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	0.22	0.33
<i>Elymus lanceolatus</i>	6.17	4.72
<i>Hesperostipa comata</i>	4.33	0.00
<i>Pascopyrum smithii</i>	0.17	0.00
<i>Poa secunda</i>	0.06	0.00
Native Perennial Forbs		
<i>Astragalus lentiginosus</i>	0.11	0.00
<i>Psoraleidum lanceolatum</i>	0.61	0.39
Native Succulents		
<i>Opuntia polyacantha</i>	0.06	0.00
Native Annuals and Biennials		
<i>Descurainia pinnata</i>	0.06	0.00
Total Native Cover	30.74	6.11
Introduced Annuals and Biennials		
<i>Alyssum desertorum</i>	0.06	0.00
Total Introduced Cover	0.06	0.00
Total Vegetative Cover	30.80	6.11

Plot 42

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	20.67	0.00
<i>Chrysothamnus viscidiflorus</i>	2.28	0.56
<i>Grayia spinosa</i>	0.11	0.00
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	0.00	0.22
<i>Elymus elymoides</i>	0.22	0.56
<i>Elymus lanceolatus</i>	0.67	2.78
<i>Hesperostipa comata</i>	4.33	4.22
<i>Poa secunda</i>	0.00	0.17
Native Perennial Forbs		
<i>Astragalus filipes</i>	0.17	0.00
<i>Crepis acuminata</i>	0.00	0.11
<i>Orobanche fasciculata</i>	0.06	0.06
<i>Schoenocrambe linifolia</i>	0.11	0.00
Native Succulents		
<i>Opuntia polyacantha</i>	0.39	0.05
Native Annuals and Biennials		
<i>Aliciella leptomeria</i>	0.00	0.06
<i>Eriastrum wilcoxii</i>	0.00	0.06
Total Native Cover	29.01	8.85
Introduced Annuals and Biennials		
<i>Alyssum desertorum</i>	0.11	0.06
<i>Bromus tectorum</i>	8.61	0.50
<i>Tragopogon dubius</i>	0.00	0.33
Total Introduced Cover	8.72	0.89
Total Vegetative Cover	37.73	9.74

Plot 43

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	10.50	0.00
<i>Chrysothamnus viscidiflorus</i>	0.89	0.28
<i>Grayia spinosa</i>	6.83	0.00
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	1.89	1.06
<i>Elymus elymoides</i>	0.11	0.22
<i>Elymus lanceolatus</i>	0.44	0.89
<i>Hesperostipa comata</i>	3.33	2.00
<i>Poa secunda</i>	0.22	0.56
Native Perennial Forbs		
<i>Phlox hoodii</i>	0.06	0.00
<i>Schoenocrambe linifolia</i>	0.00	0.06
Total Native Cover	24.27	5.07
Introduced Annuals and Biennials		
<i>Bromus tectorum</i>	38.11	0.22
<i>Tragopogon dubius</i>	0.00	0.33
Total Introduced Cover	38.11	0.55
Total Vegetative Cover	62.38	5.62

Plot 44

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	14.78	0.00
<i>Chrysothamnus viscidiflorus</i>	4.89	0.00
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	0.00	0.67
<i>Elymus lanceolatus</i>	0.44	2.38
<i>Hesperostipa comata</i>	6.39	2.11
<i>Sporobolus cryptandrus</i>	0.00	0.33
Native Perennial Forbs		
<i>Astragalus ceramicus</i>	0.06	0.00
<i>Comandra umbellata</i>	5.17	1.83
<i>Eriogonum ovalifolium</i>	0.17	0.17
<i>Machaeranthera canescens</i>	0.00	0.06
<i>Phacelia hastata</i>	0.11	0.06
<i>Pleiocanthus spinosus</i>	0.44	0.39
<i>Schoenocrambe linifolia</i>	0.06	0.11
Native Succulents		
<i>Opuntia polyacantha</i>	0.33	0.17
Native Annuals and Biennials		
<i>Camissonia minor</i>	0.00	0.22
<i>Chenopodium leptophyllum</i>	0.00	0.06
<i>Descurainia pinnata</i>	0.11	0.00
Total Native Cover	32.95	8.56
Introduced Annuals and Biennials		
<i>Bromus tectorum</i>	0.83	0.22
<i>Tragopogon dubius</i>	0.11	0.22
Total Introduced Cover	0.94	0.44
Total Vegetative Cover	33.89	9.00

Plot 45

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	14.44	0.00
<i>Linanthus pungens</i>	0.22	0.00
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	0.94	0.22
<i>Elymus elymoides</i>	0.06	0.00
<i>Elymus lanceolatus</i>	1.44	2.33
<i>Hesperostipa comata</i>	13.17	7.11
<i>Poa secunda</i>	0.06	0.11
Native Perennial Forbs		
<i>Arenaria franklinii</i>	0.11	0.00
<i>Astragalus calycosus</i>	0.06	0.00
<i>Comandra umbellata</i>	0.22	0.17
<i>Lygodesmia grandiflora</i>	0.06	0.00
<i>Orobanche fasciculata</i>	0.06	0.00
<i>Pleiacanthus spinosus</i>	1.06	0.22
<i>Stephanomeria minor</i>	0.00	0.11
Native Succulents		
<i>Opuntia polyacantha</i>	0.33	0.00
Native Annuals and Biennials		
<i>Chenopodium leptophyllum</i>	0.06	0.00
<i>Descurainia pinnata</i>	0.33	0.06
<i>Eriastrum wilcoxii</i>	0.17	0.00
<i>Eriogonum cernuum</i>	0.22	0.00
Total Native Cover	33.01	10.33
Introduced Annuals and Biennials		
<i>Bromus tectorum</i>	3.11	0.22
<i>Tragopogon dubius</i>	0.00	0.33
Total Introduced Cover	3.11	0.55
Total Vegetative Cover	36.12	10.88

Plot 46

Species	2011	2012
Native Shrubs		
<i>Artemisia tridentata</i>	10.89	0.00
<i>Chrysothamnus viscidiflorus</i>	1.17	0.00
Native Perennial Graminoids		
<i>Achnatherum hymenoides</i>	0.33	0.11
<i>Elymus elymoides</i>	0.33	0.00
<i>Elymus lanceolatus</i>	2.94	9.44
<i>Hesperostipa comata</i>	0.00	0.17
<i>Pascopyrum smithii</i>	5.06	0.00
Native Perennial Forbs		
<i>Astragalus calycosus</i>	0.33	0.00
<i>Erigeron pumilus</i>	0.11	0.00
<i>Eriogonum ovalifolium</i>	0.39	0.00
<i>Grindelia squarrosa</i>	0.83	0.00
<i>Phlox hoodii</i>	0.11	0.00
Native Succulents		
<i>Opuntia polyacantha</i>	0.28	0.06
Native Annuals and Biennials		
<i>Cordylanthus ramosus</i>	0.06	0.00
Total Native Cover	22.83	9.78
Introduced Annuals and Biennials		
Total Introduced Cover	0.00	0.00
Total Vegetative Cover	22.83	9.78

APPENDIX D

PLOT SUMMARIES (CHAPTER 4)

Plot photos, summary statistics, and age class distributions for 14 Sagebrush Demography plots at the INL Site.
Plots are organized by mean age in ascending order.

PLOT 9

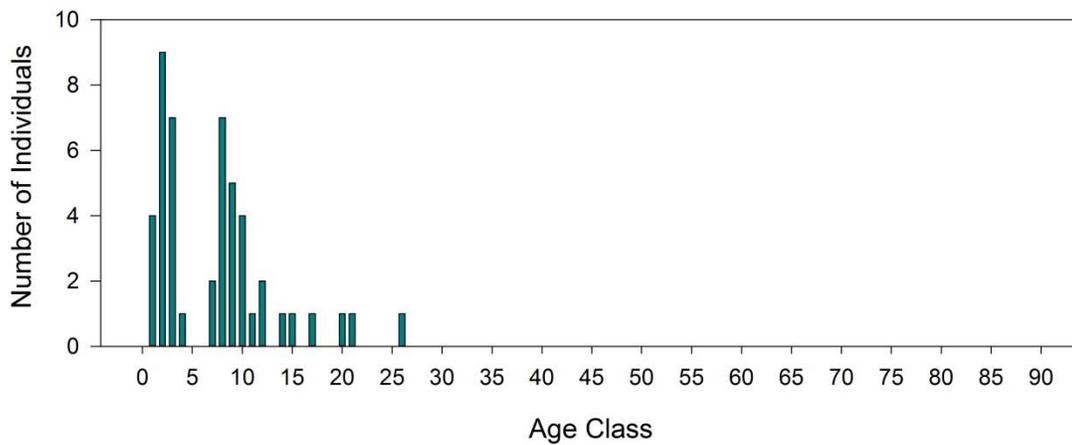
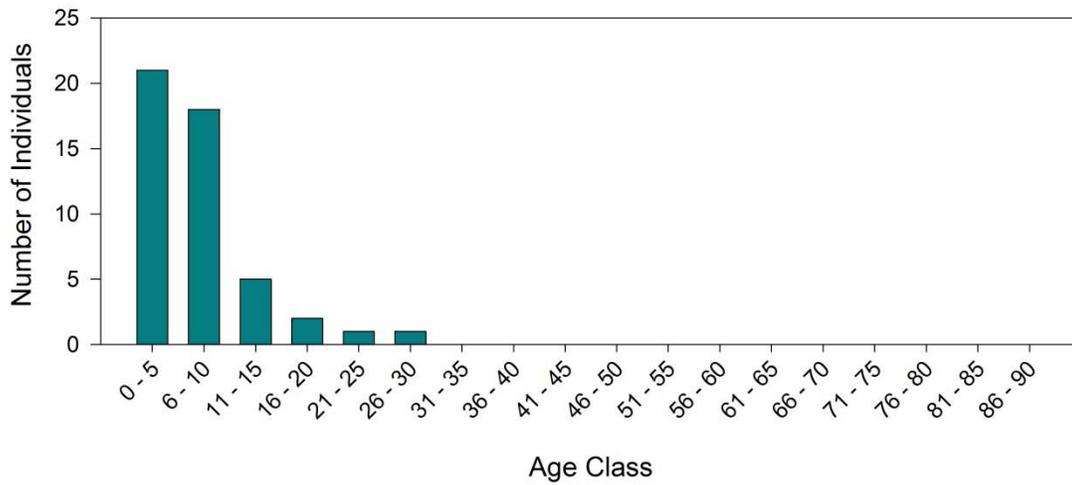
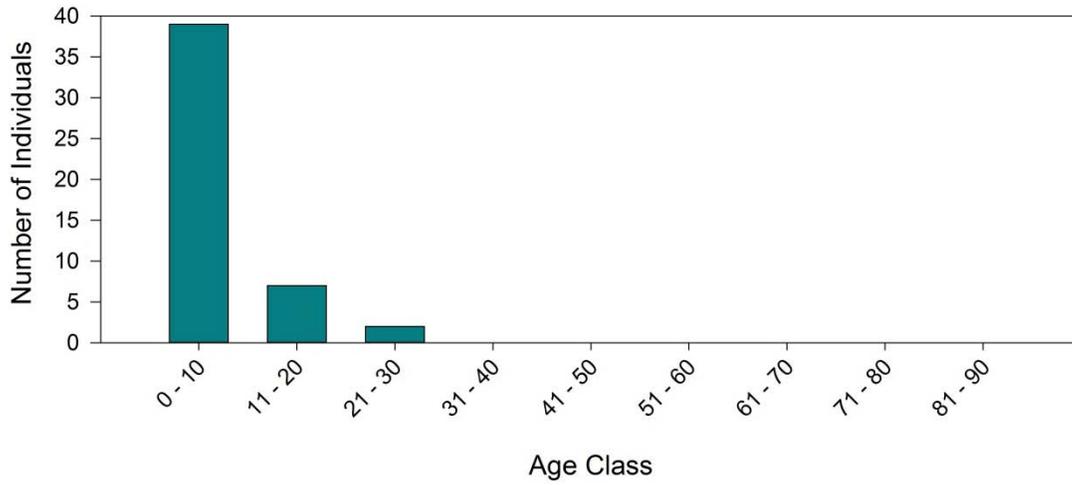


Summary Statistics

Live Shrubs – Mean Age	7.3 years
Live Shrubs – Minimum Age	1 year
Live Shrubs – Maximum Age	26 years
Dead Shrubs – Mean Age	27.7 years
Dead Shrubs – Minimum Age	7 years
Dead Shrubs – Maximum Age	58 years
Mean Big Sagebrush Cover	15.4%
Mean Big Sagebrush Density	12.8 shrubs/m ²
Mean Dead Shrub Cover	11.3%
Mean Condition Rank	2.9

Age Class Distributions

Plot 9
(n = 48)



PLOT 11

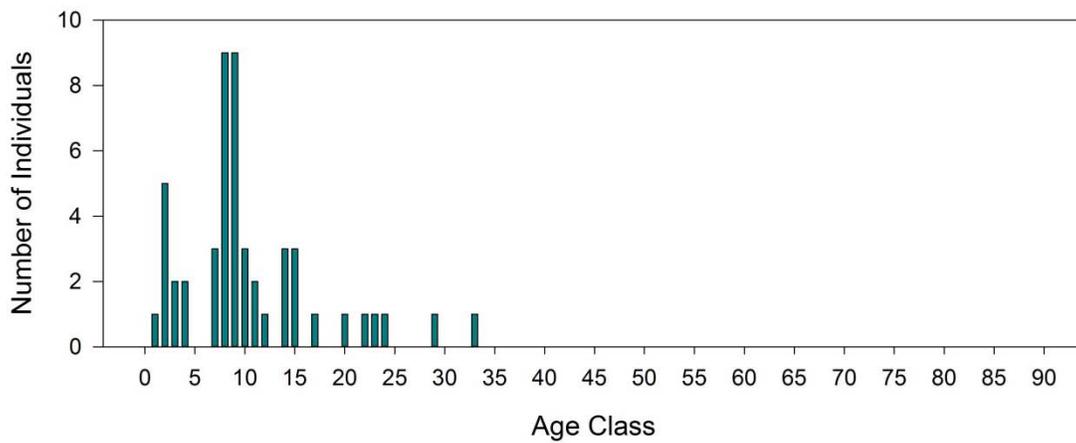
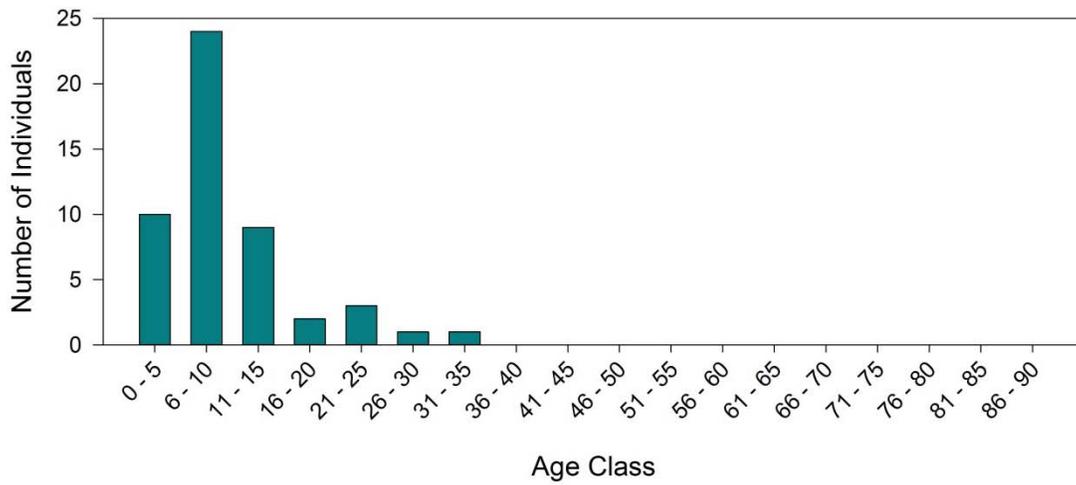
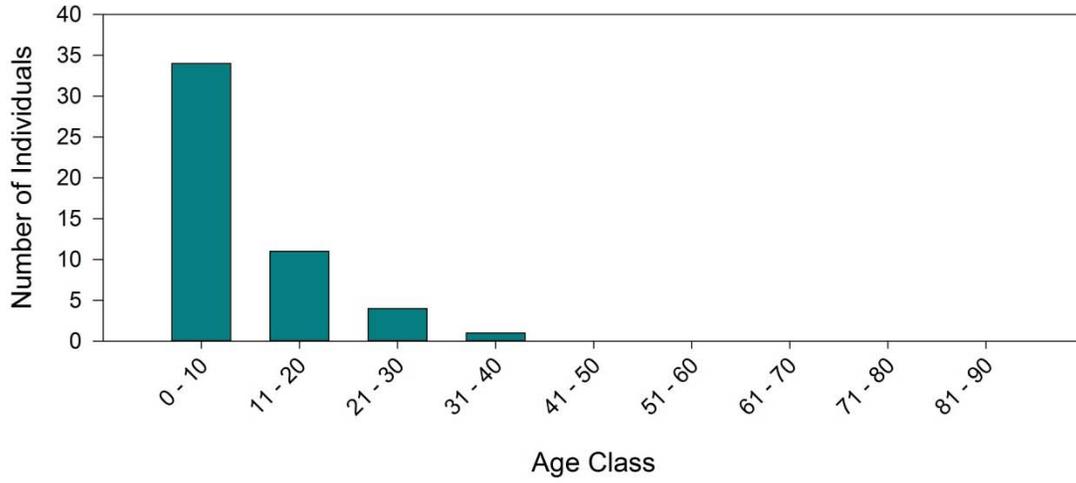


Summary Statistics

Live Shrubs – Mean Age	10.4 years
Live Shrubs – Minimum Age	1 year
Live Shrubs – Maximum Age	33 years
Dead Shrubs – Mean Age	30.3 years
Dead Shrubs – Minimum Age	7 years
Dead Shrubs – Maximum Age	64 years
Mean Big Sagebrush Cover	15.1%
Mean Big Sagebrush Density	3.5 shrubs/m ²
Mean Dead Shrub Cover	9.3%
Mean Condition Rank	2.6

Age Class Distributions

Plot 11
(n = 50)



PLOT 13

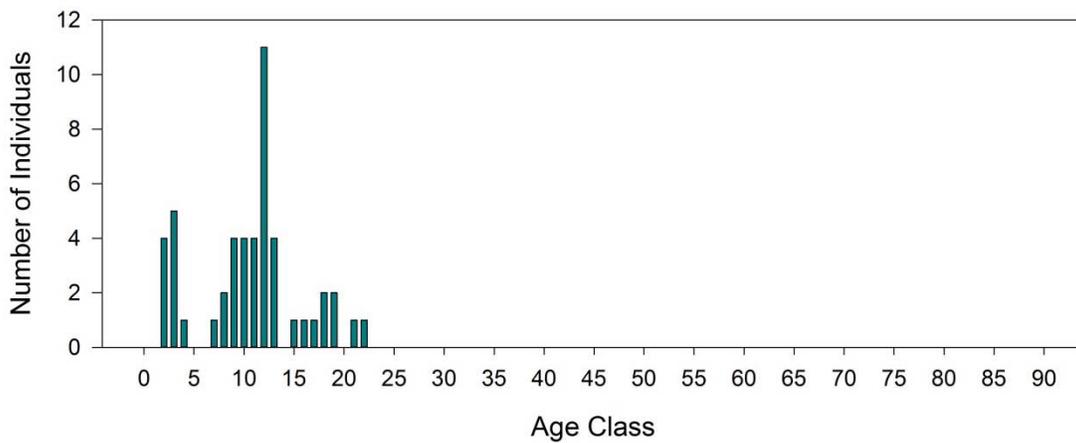
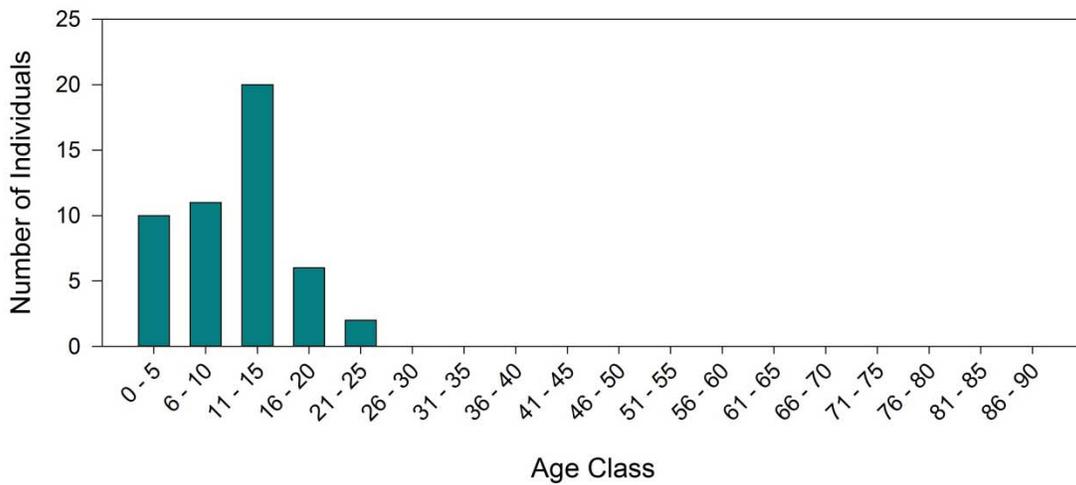
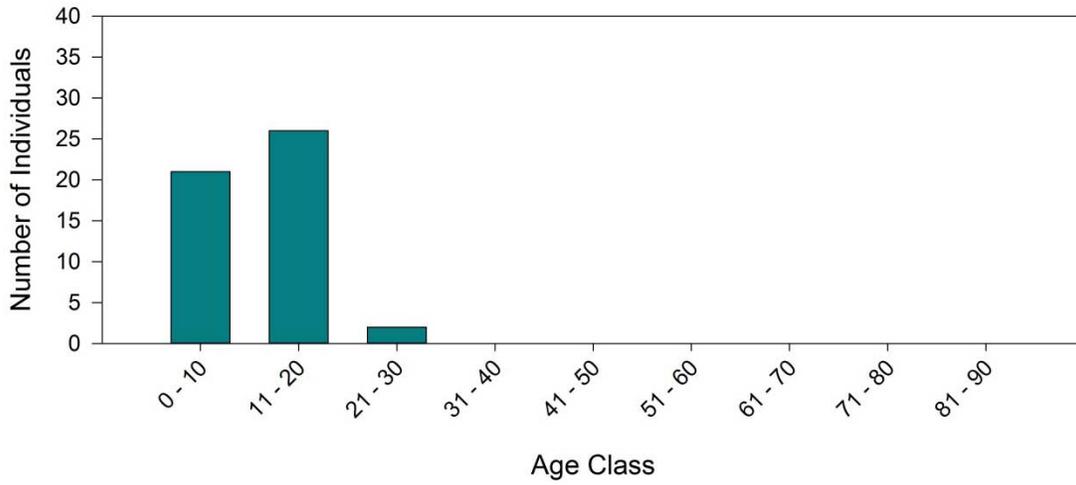


Summary Statistics

Live Shrubs – Mean Age	10.6 years
Live Shrubs – Minimum Age	2 years
Live Shrubs – Maximum Age	22 years
Dead Shrubs – Mean Age	26.4 years
Dead Shrubs – Minimum Age	9 years
Dead Shrubs – Maximum Age	66 years
Mean Big Sagebrush Cover	7.9%
Mean Big Sagebrush Density	1.8 shrubs/m ²
Mean Dead Shrub Cover	11.1%
Mean Condition Rank	2.5

Age Class Distributions

Plot 13
(n = 49)



PLOT 8

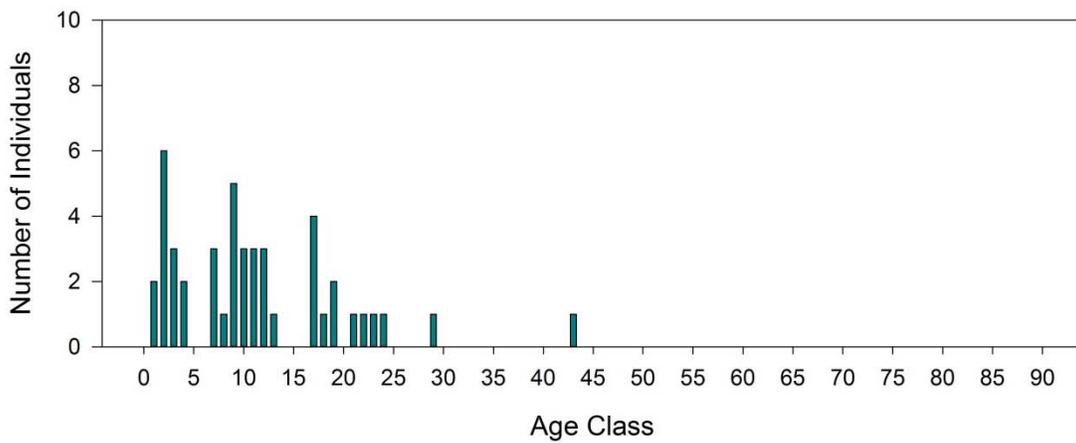
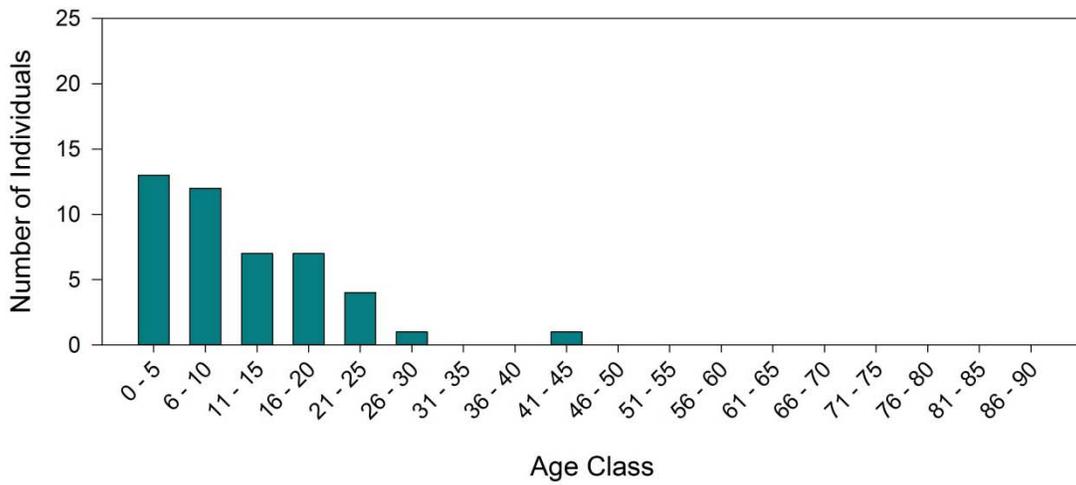
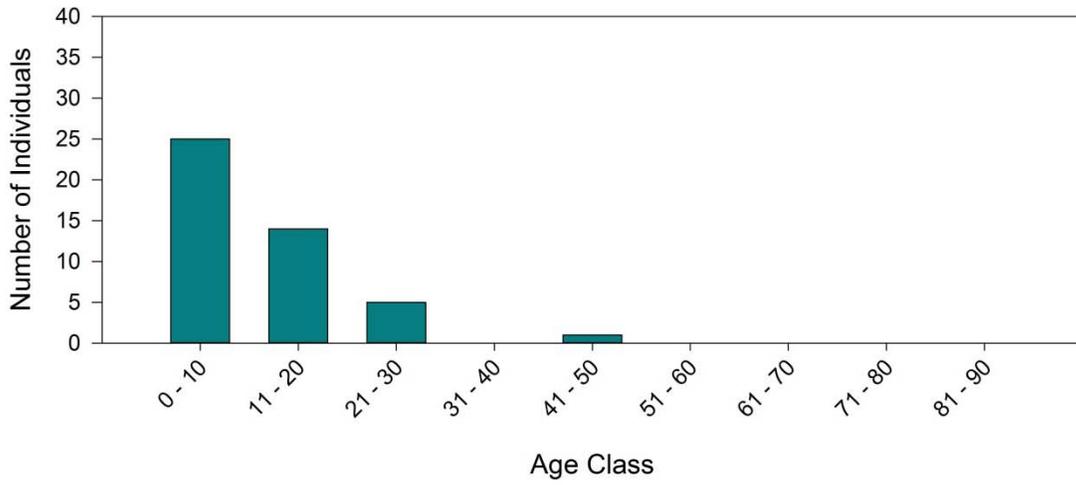


Summary Statistics

Live Shrubs – Mean Age	11.2 years
Live Shrubs – Minimum Age	1 year
Live Shrubs – Maximum Age	43 years
Dead Shrubs – Mean Age	26.7 years
Dead Shrubs – Minimum Age	9 years
Dead Shrubs – Maximum Age	53 years
Mean Big Sagebrush Cover	21.0%
Mean Big Sagebrush Density	4.9 shrubs/m ²
Mean Dead Shrub Cover	7.6%
Mean Condition Rank	3.3

Age Class Distributions

Plot 8
(n = 45)



PLOT 19

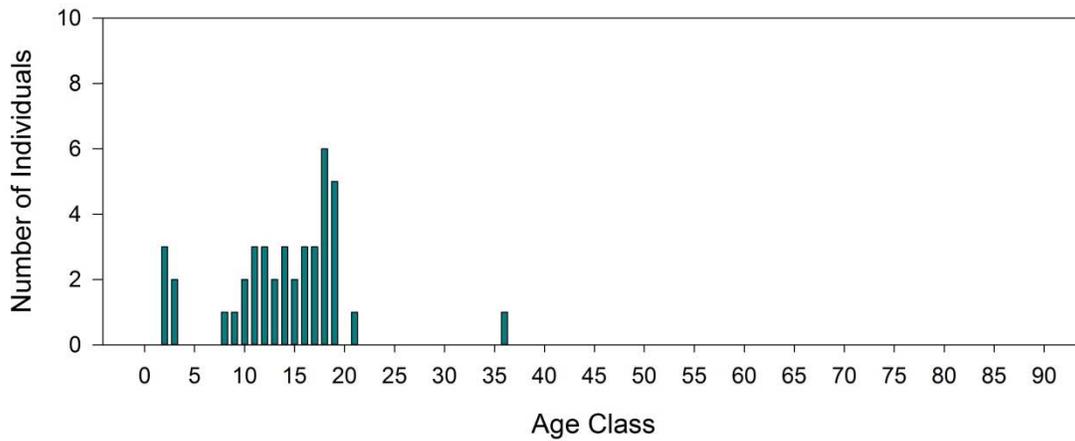
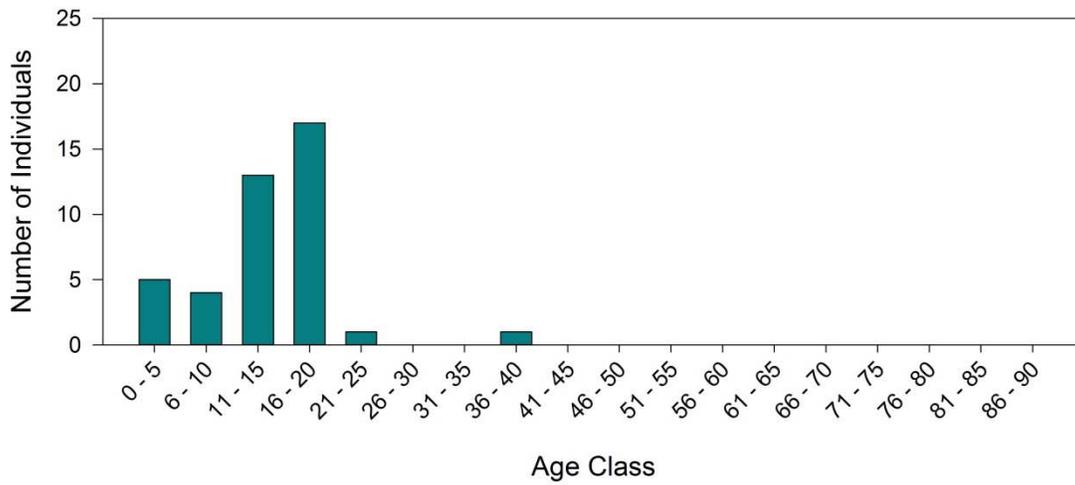
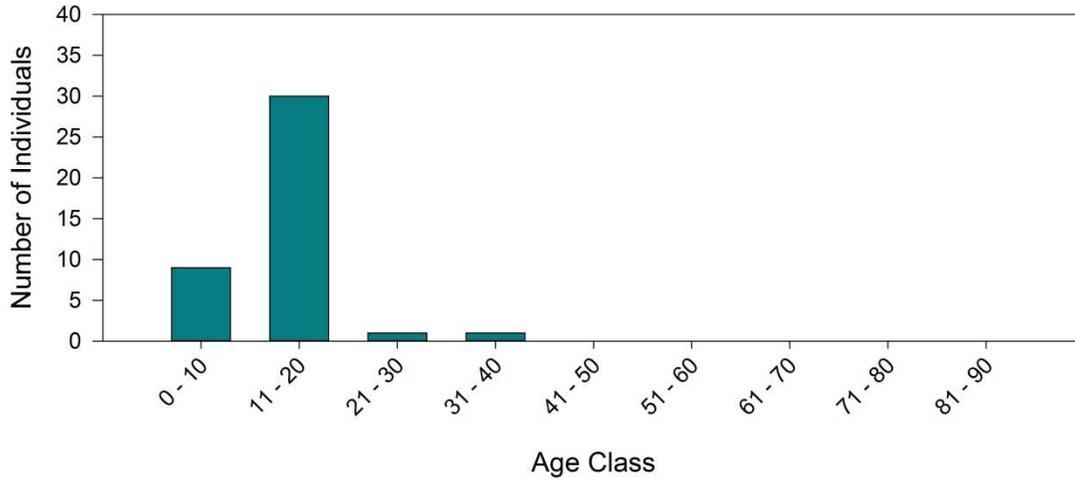


Summary Statistics

Live Shrubs – Mean Age	14.0 years
Live Shrubs – Minimum Age	2 years
Live Shrubs – Maximum Age	36 years
Dead Shrubs – Mean Age	25.9 years
Dead Shrubs – Minimum Age	7 years
Dead Shrubs – Maximum Age	46 years
Mean Big Sagebrush Cover	14.6%
Mean Big Sagebrush Density	5.7 shrubs/m ²
Mean Dead Shrub Cover	10.4%
Mean Condition Rank	2.8

Age Class Distributions

Plot 19
(n = 41)



PLOT 18

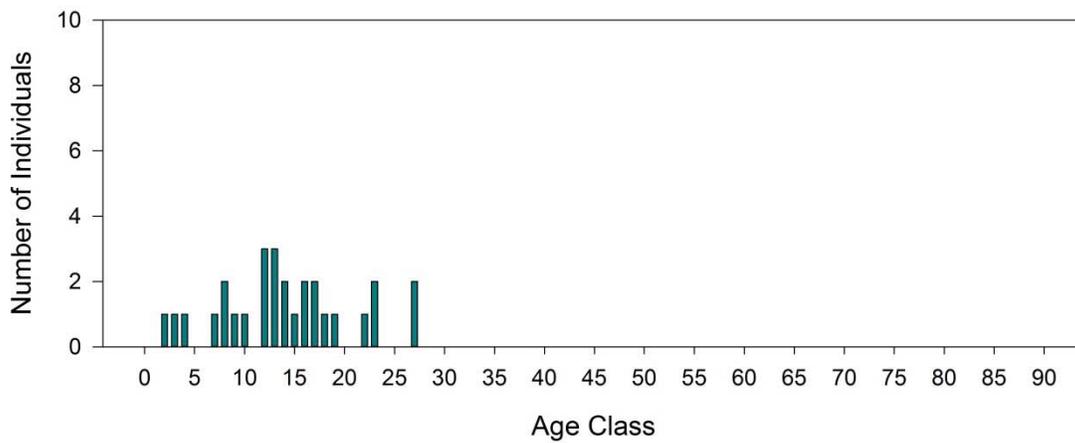
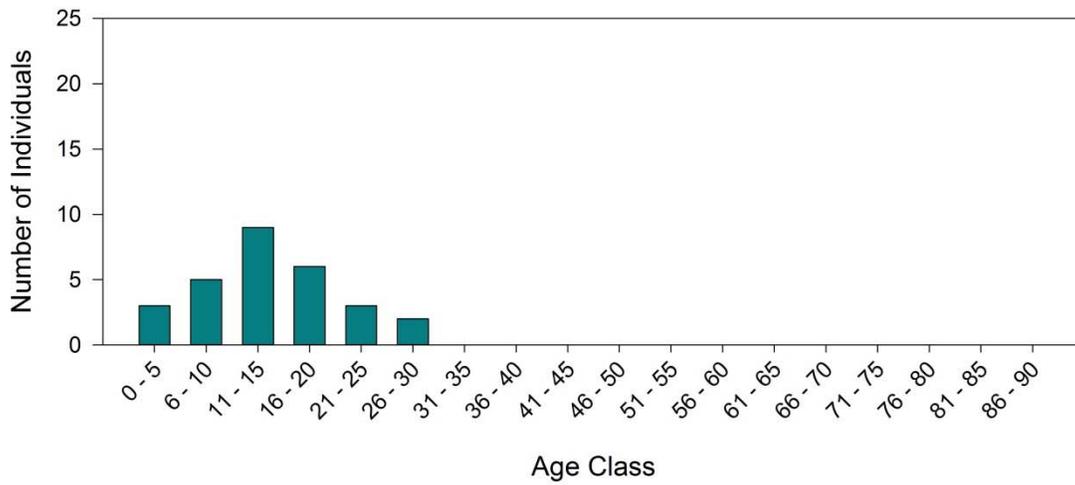
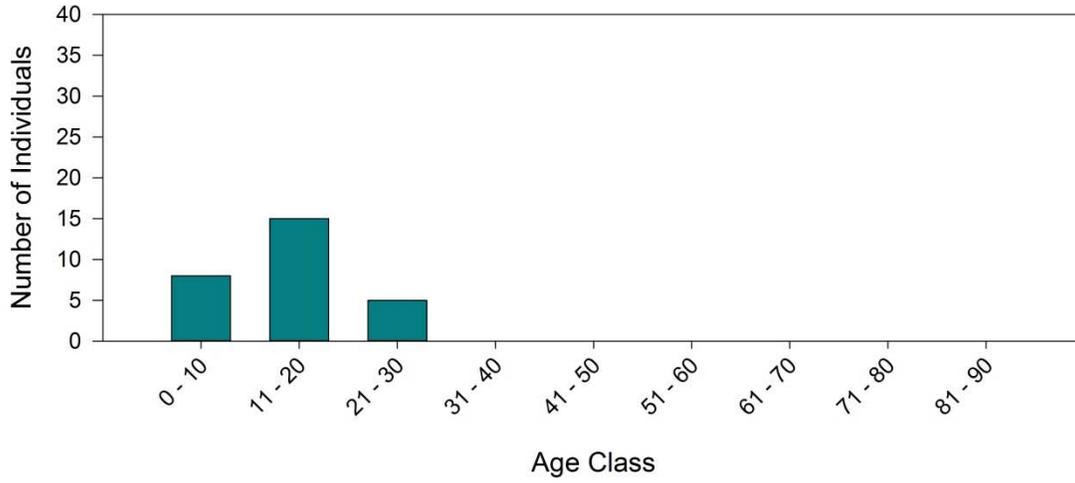


Summary Statistics

Live Shrubs – Mean Age	14.1 years
Live Shrubs – Minimum Age	2 years
Live Shrubs – Maximum Age	27 years
Dead Shrubs – Mean Age	31.1 years
Dead Shrubs – Minimum Age	7 years
Dead Shrubs – Maximum Age	52 years
Mean Big Sagebrush Cover	7.0%
Mean Big Sagebrush Density	1.9 shrubs/m ²
Mean Dead Shrub Cover	9.0%
Mean Condition Rank	2.4

Age Class Distributions

Plot 18
(n = 28)



PLOT 14

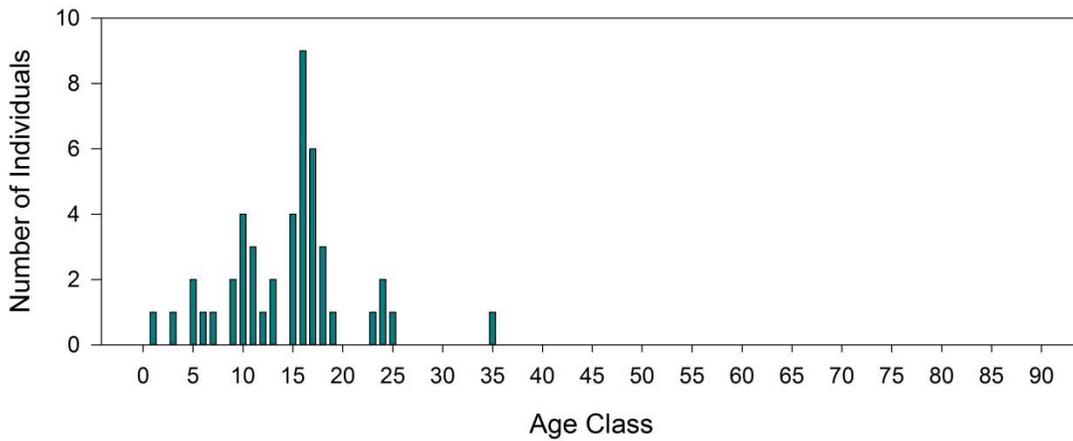
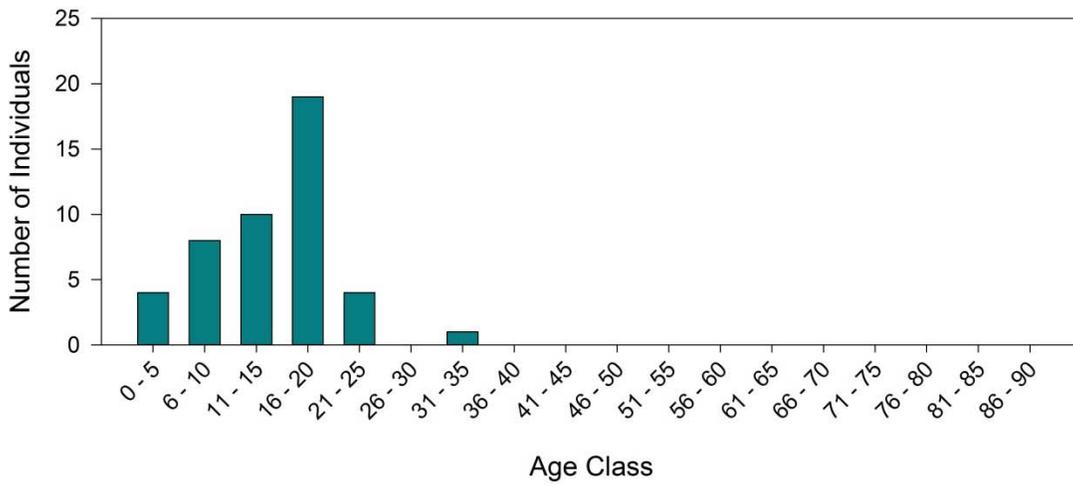
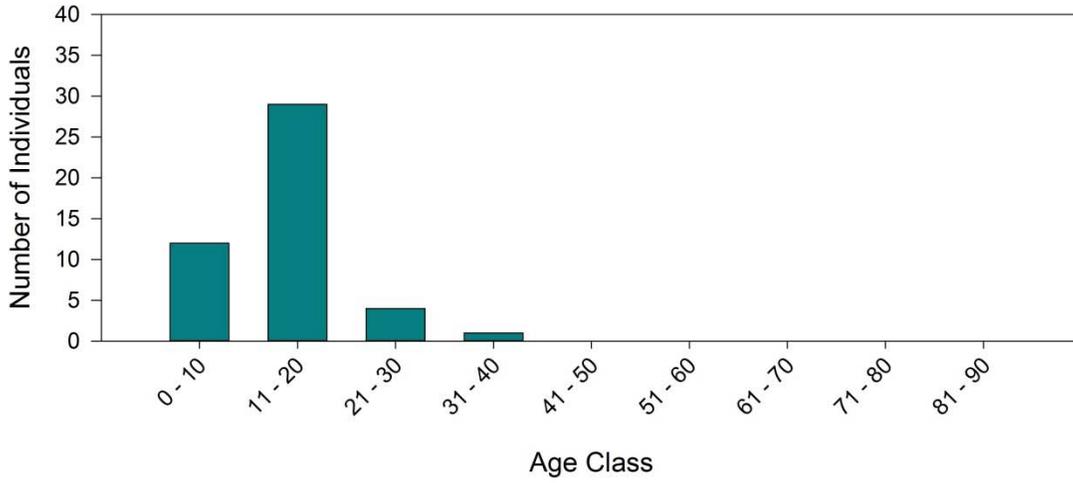


Summary Statistics

Live Shrubs – Mean Age	14.5 years
Live Shrubs – Minimum Age	1 year
Live Shrubs – Maximum Age	35 years
Dead Shrubs – Mean Age	33.3 years
Dead Shrubs – Minimum Age	10 years
Dead Shrubs – Maximum Age	59 years
Mean Big Sagebrush Cover	2.3%
Mean Big Sagebrush Density	0.2 shrubs/m ²
Mean Dead Shrub Cover	8.6%
Mean Condition Rank	2.2

Age Class Distributions

Plot 14
(n = 46)



PLOT 15

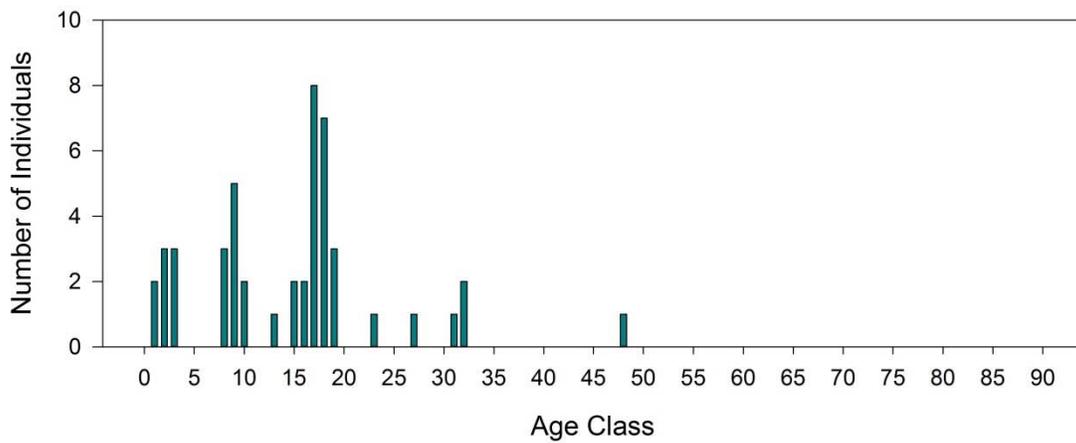
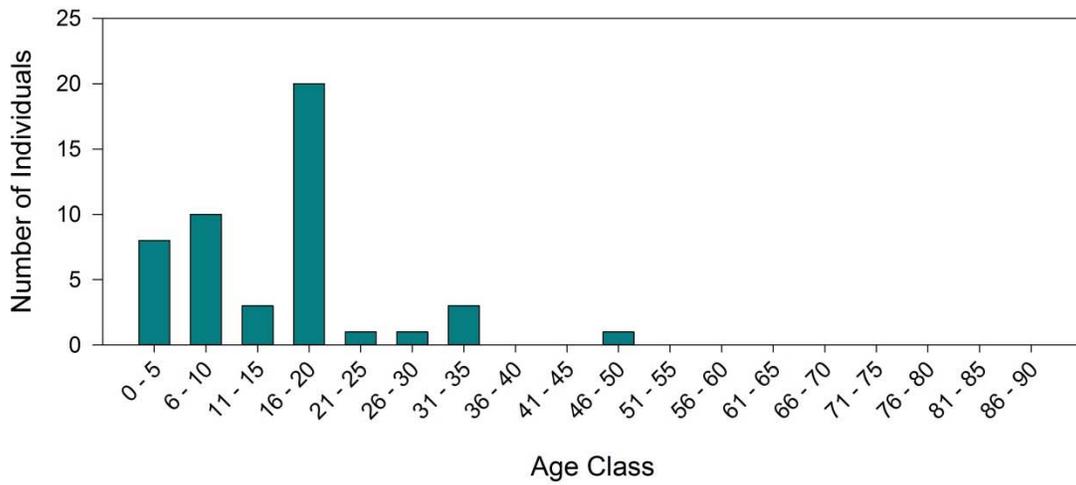
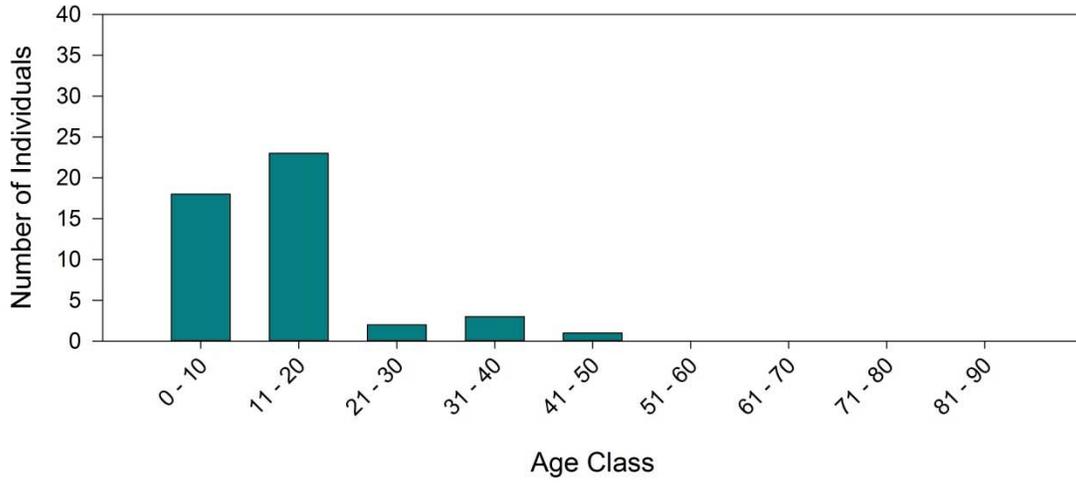


Summary Statistics

Live Shrubs – Mean Age	14.7 years
Live Shrubs – Minimum Age	1 year
Live Shrubs – Maximum Age	48 years
Dead Shrubs – Mean Age	34.9 years
Dead Shrubs – Minimum Age	6 years
Dead Shrubs – Maximum Age	78 years
Mean Big Sagebrush Cover	18.3%
Mean Big Sagebrush Density	10.8 shrubs/m ²
Mean Dead Shrub Cover	9.8%
Mean Condition Rank	3.4

Age Class Distributions

Plot 15
(n = 47)



PLOT 7

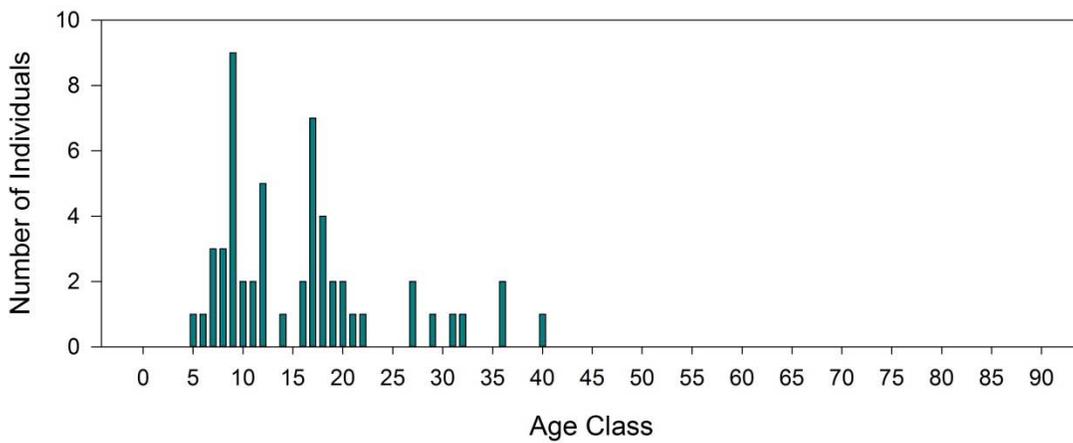
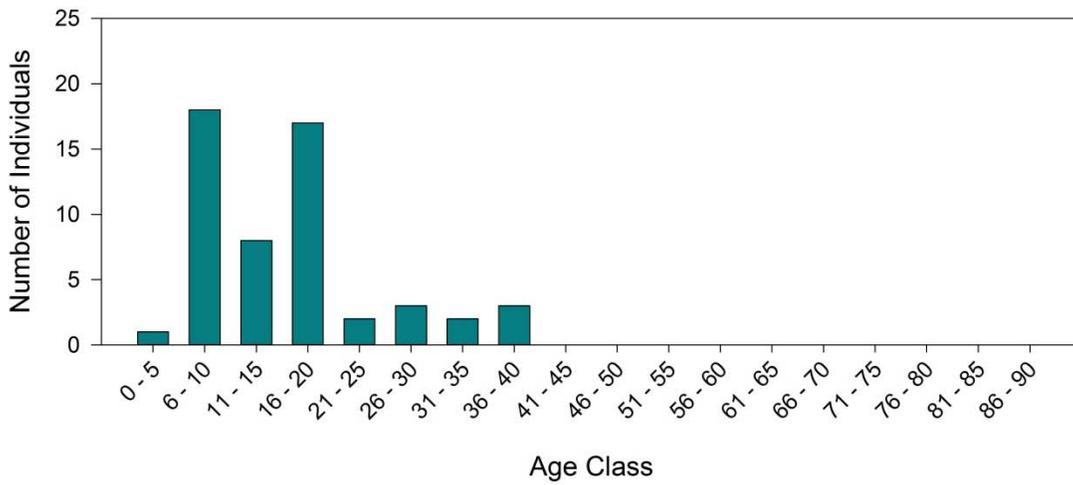
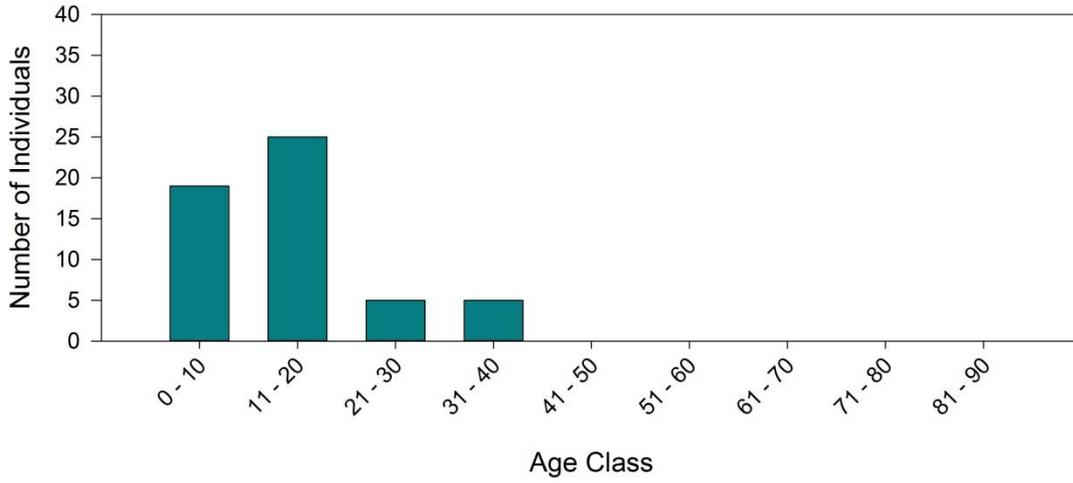


Summary Statistics

Live Shrubs – Mean Age	15.8 years
Live Shrubs – Minimum Age	5 years
Live Shrubs – Maximum Age	40 years
Dead Shrubs – Mean Age	33.2 years
Dead Shrubs – Minimum Age	13 years
Dead Shrubs – Maximum Age	62 years
Mean Big Sagebrush Cover	17.8%
Mean Big Sagebrush Density	7.3 shrubs/m ²
Mean Dead Shrub Cover	5.7%
Mean Condition Rank	3.5

Age Class Distributions

Plot 7
(n = 54)



PLOT 17

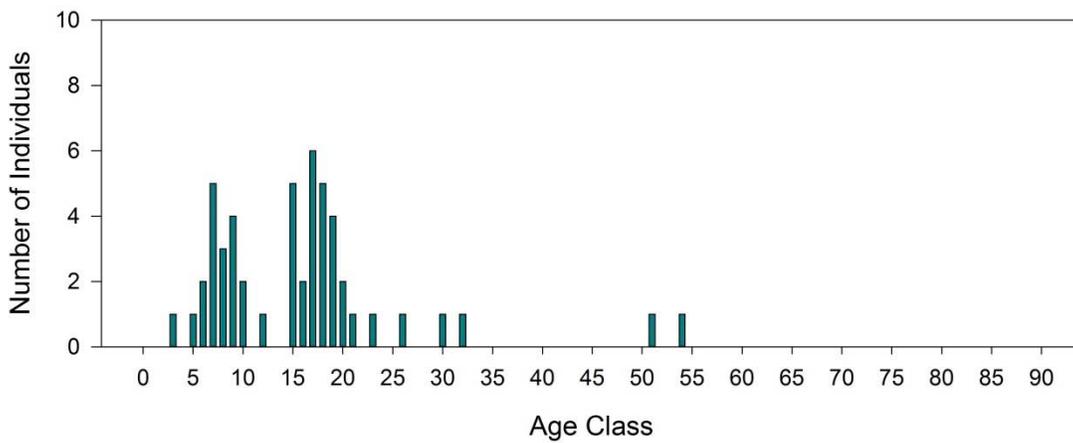
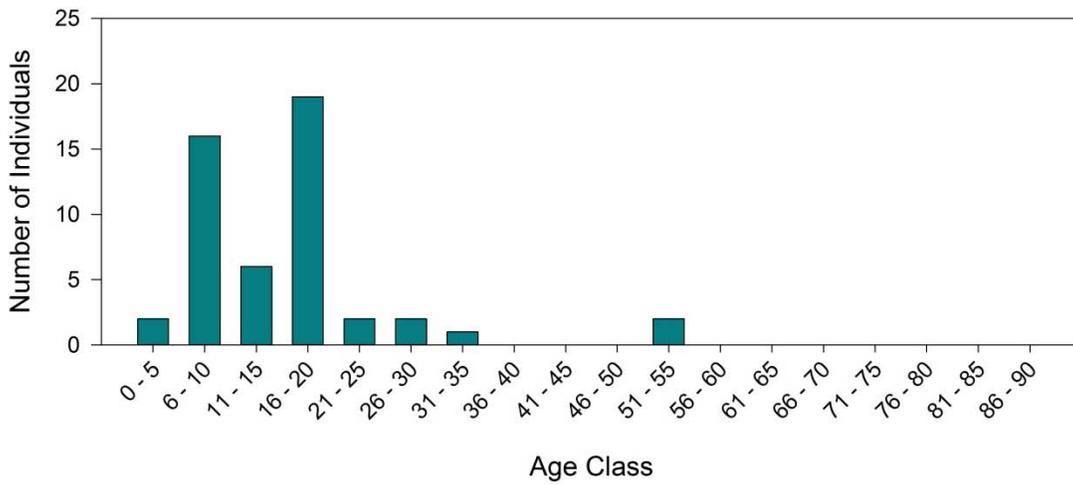
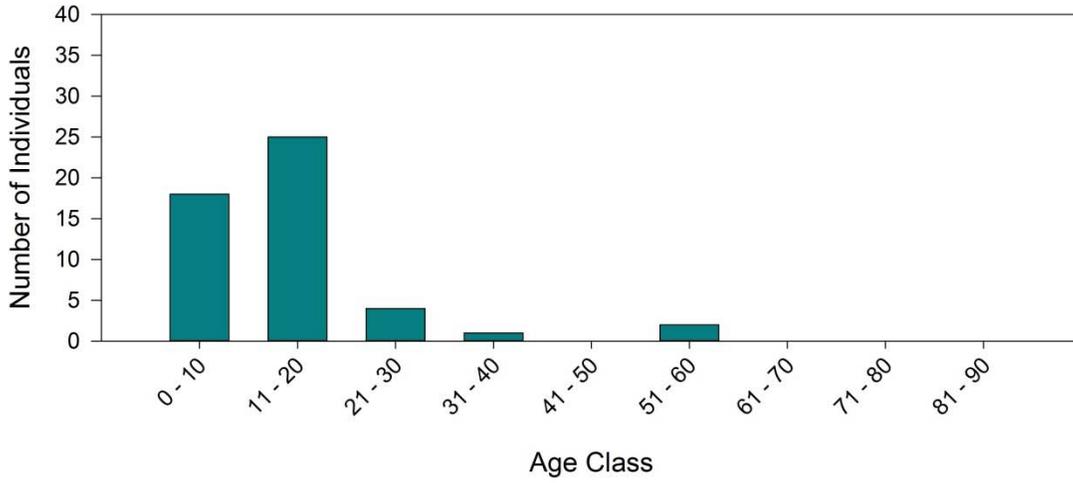


Summary Statistics

Live Shrubs – Mean Age	16.0 years
Live Shrubs – Minimum Age	3 years
Live Shrubs – Maximum Age	54 years
Dead Shrubs – Mean Age	35.9 years
Dead Shrubs – Minimum Age	9 years
Dead Shrubs – Maximum Age	75 years
Mean Big Sagebrush Cover	12.3%
Mean Big Sagebrush Density	6.1 shrubs/m ²
Mean Dead Shrub Cover	8.8%
Mean Condition Rank	3.0

Age Class Distributions

Plot 17
(n = 50)



PLOT 16

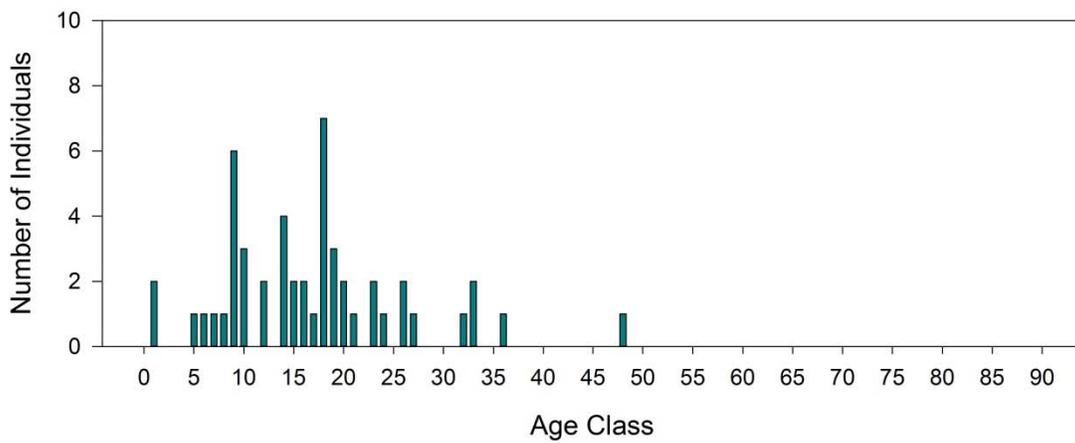
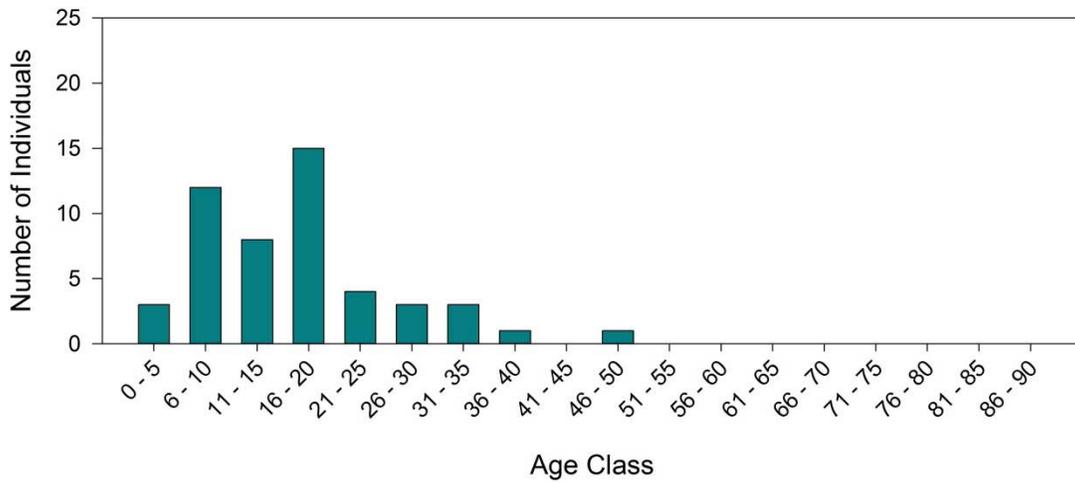
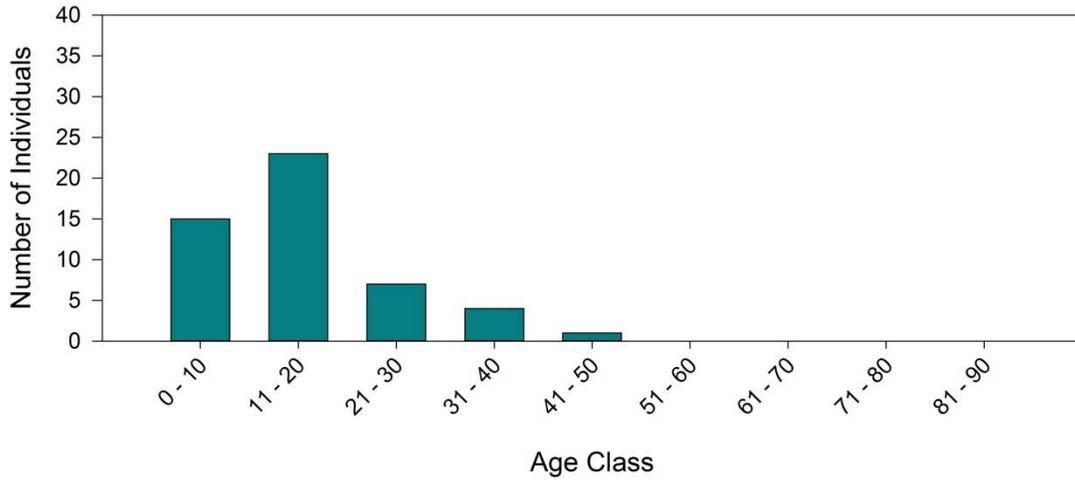


Summary Statistics

Live Shrubs – Mean Age	16.9 years
Live Shrubs – Minimum Age	1 year
Live Shrubs – Maximum Age	48 years
Dead Shrubs – Mean Age	38.9 years
Dead Shrubs – Minimum Age	6 years
Dead Shrubs – Maximum Age	83 years
Mean Big Sagebrush Cover	8.3%
Mean Big Sagebrush Density	6.5 shrubs/m ²
Mean Dead Shrub Cover	7.6%
Mean Condition Rank	2.8

Age Class Distributions

Plot 16
(n = 50)



PLOT 4

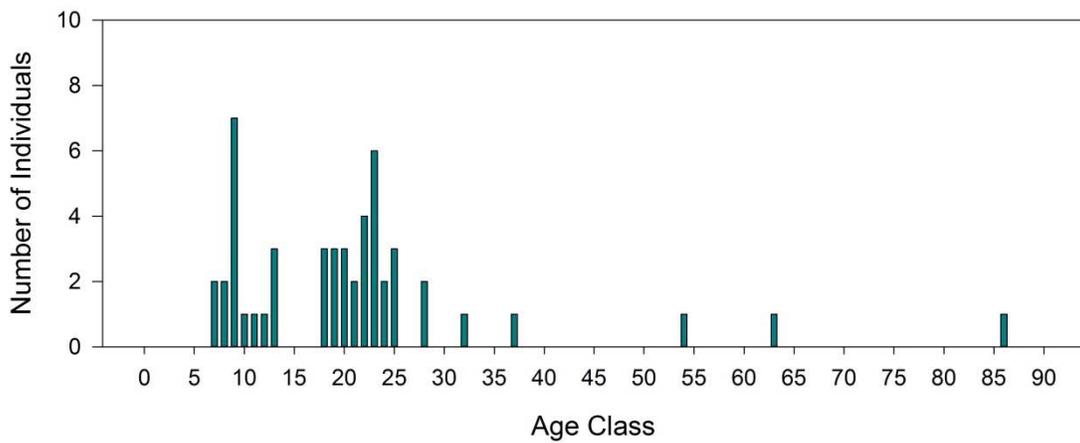
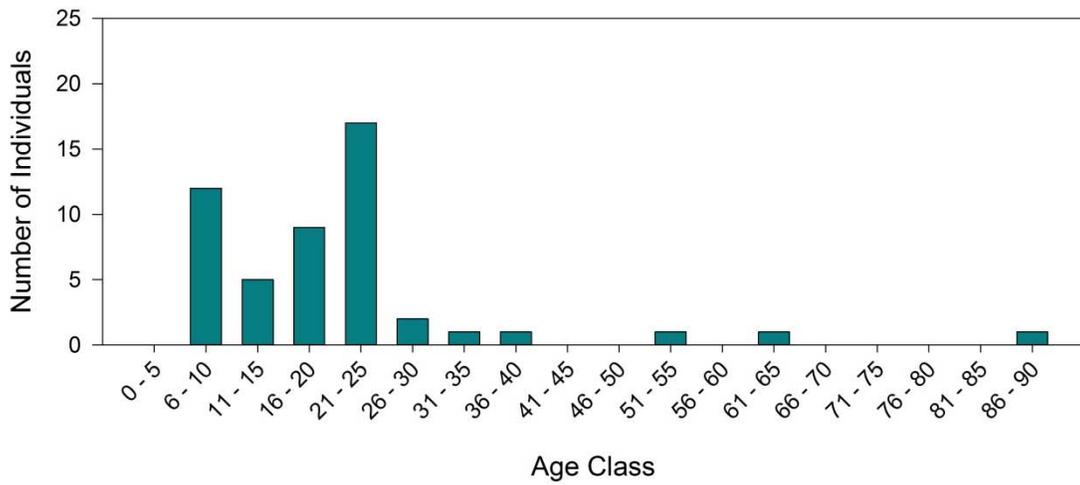
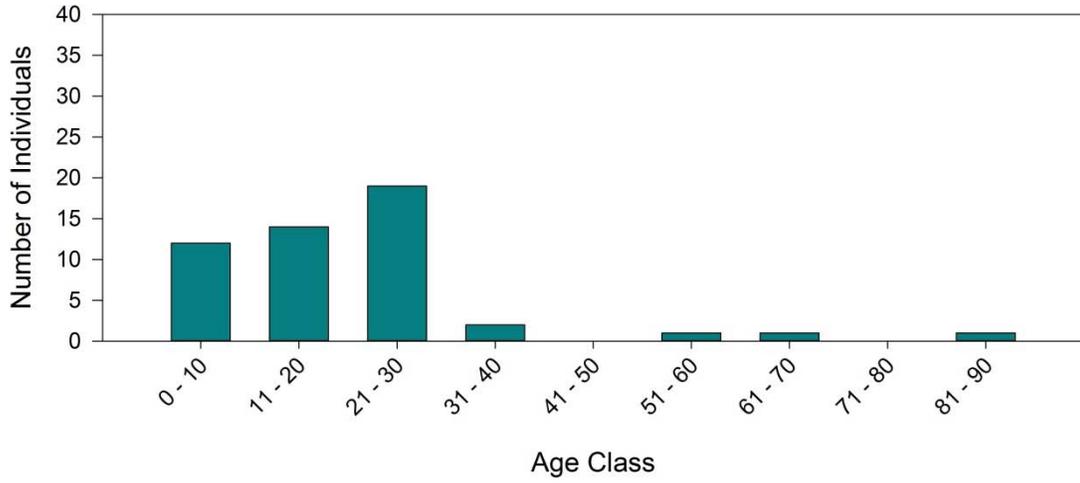


Summary Statistics

Live Shrubs – Mean Age	21.1 years
Live Shrubs – Minimum Age	7 years
Live Shrubs – Maximum Age	86 years
Dead Shrubs – Mean Age	49.4 years
Dead Shrubs – Minimum Age	17 years
Dead Shrubs – Maximum Age	83 years
Mean Big Sagebrush Cover	8.1%
Mean Big Sagebrush Density	1.2 shrubs/m ²
Mean Dead Shrub Cover	16.3%
Mean Condition Rank	2.4

Age Class Distributions

Plot 4
 (n = 50)



PLOT 10

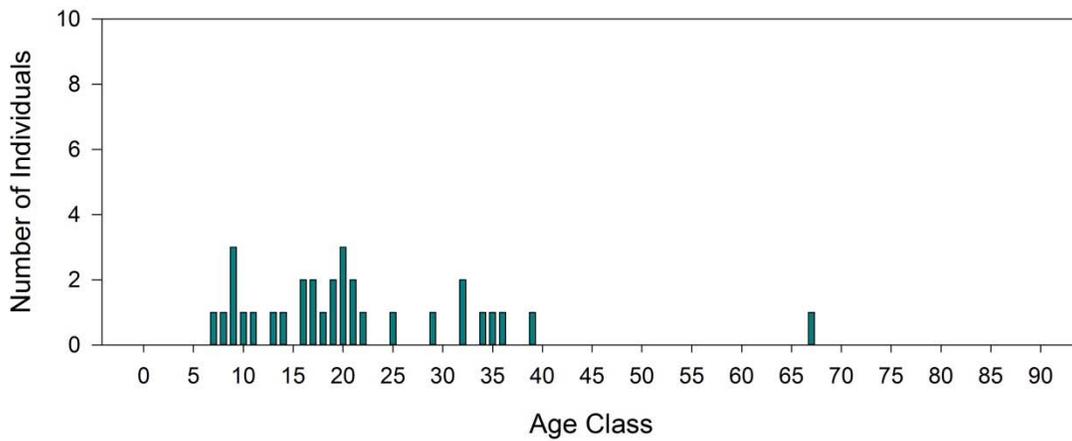
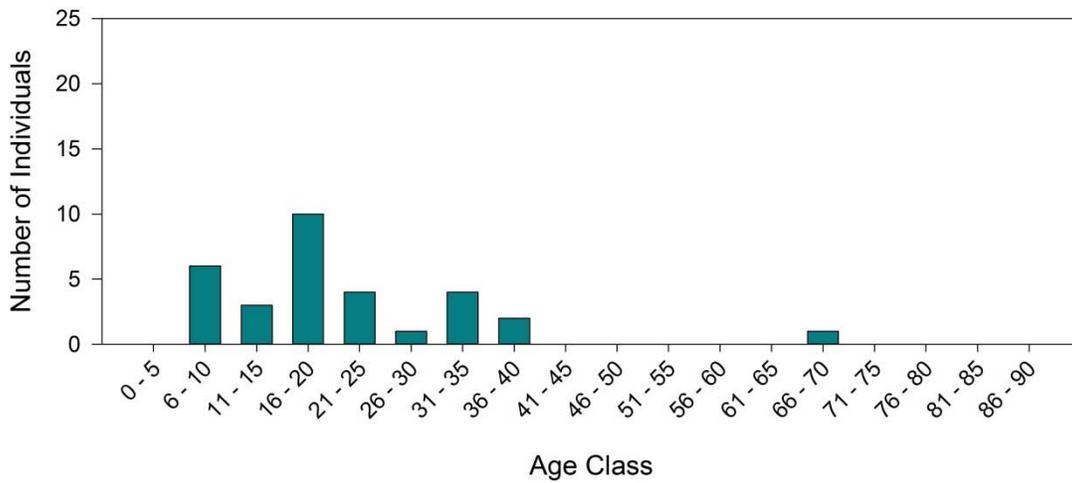
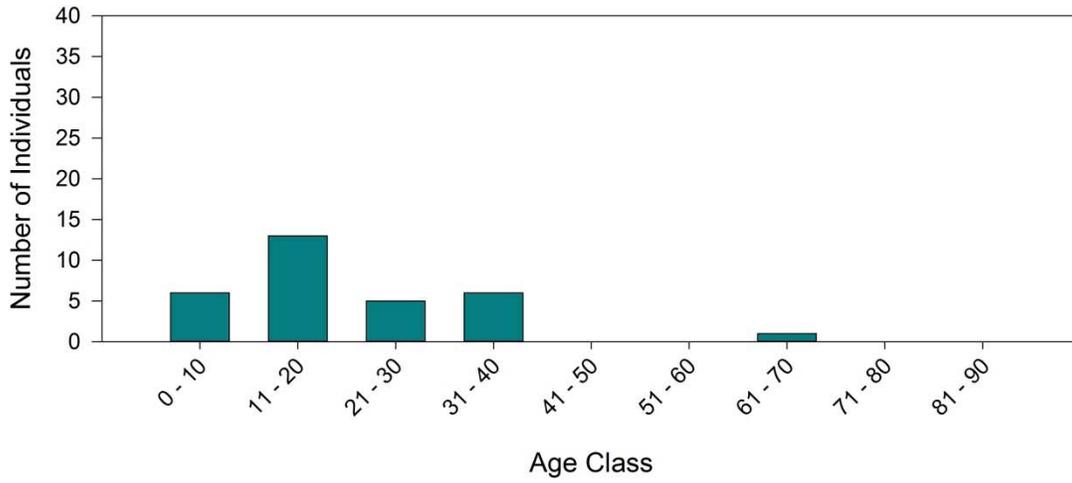


Summary Statistics

Live Shrubs – Mean Age	21.5 years
Live Shrubs – Minimum Age	7 years
Live Shrubs – Maximum Age	67 years
Dead Shrubs – Mean Age	33.0 years
Dead Shrubs – Minimum Age	12 years
Dead Shrubs – Maximum Age	67 years
Mean Big Sagebrush Cover	10.3%
Mean Big Sagebrush Density	5.3 shrubs/m ²
Mean Dead Shrub Cover	13.1%
Mean Condition Rank	2.7

Age Class Distributions

Plot 10
(n = 31)



PLOT 5

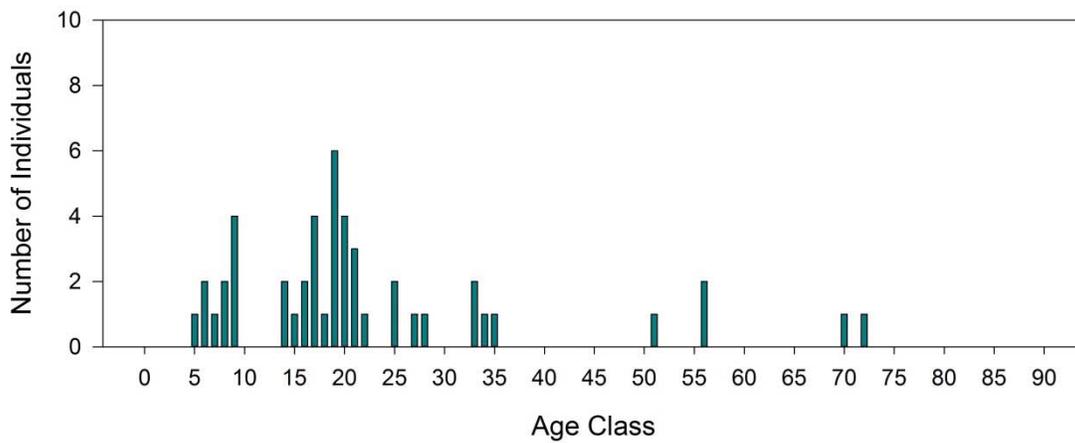
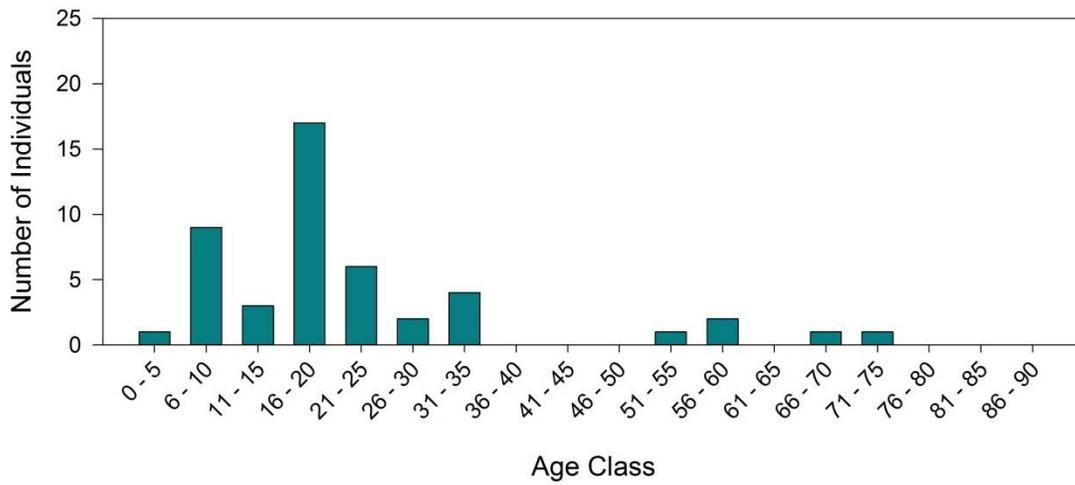
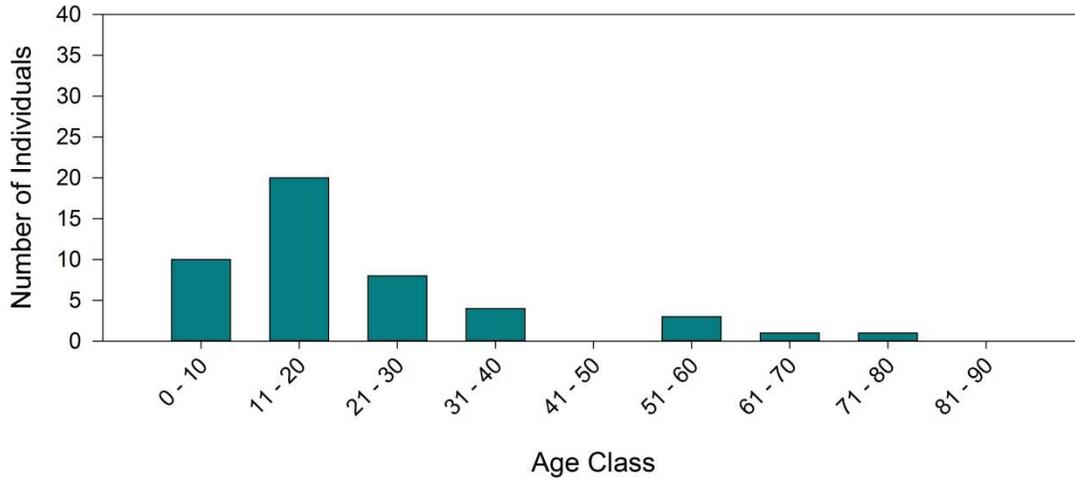


Summary Statistics

Live Shrubs – Mean Age	22.6 years
Live Shrubs – Minimum Age	5 years
Live Shrubs – Maximum Age	72 years
Dead Shrubs – Mean Age	37.3 years
Dead Shrubs – Minimum Age	10 years
Dead Shrubs – Maximum Age	65 years
Mean Big Sagebrush Cover	9.8%
Mean Big Sagebrush Density	2.7 shrubs/m ²
Mean Dead Shrub Cover	6.1%
Mean Condition Rank	2.6

Age Class Distributions

Plot 5
(n = 47)



APPENDIX E

CORRELATION RESULTS (CHAPTER 4)

Scatterplots for significant pairwise comparisons from a multiple correlation analysis comparing mean stand age and several variables thought to be indicative of stand condition.

