LONG-TERM VEGETATION DYNAMICS AND POST-FIRE ESTABLISHMENT PATTERNS OF SAGEBRUSH STEPPE

A Thesis

Presented in Partial Fulfillment of the Requirements for the

Degree of Master of Science

with a

Major in Rangeland Ecology and Management

in the

College of Graduate Studies

University of Idaho

by

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May 2003

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AUTHORIZATION TO SUBMIT

THESIS

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ABSTRACT

Many sagebrush steppe communities have become highly fragmented, largely due to anthropogenic disturbances, agricultural conversion, invasive species, and the increasing frequency of large wildfires. Greater understanding of sagebrush steppe communities is needed to protect and manage these lands from further degradation. The objectives of this study were to monitor long-term vegetation plots and evaluate vegetation dynamics after several recent, large wildfires; study post-fire spatial patterns of big sagebrush (*Artemisia tridentata* Nutt.) establishment; and study temporal patterns of big sagebrush establishment in relation to environmental influences. The study area was located at the Idaho National Engineering and Environmental Laboratory (INEEL) in southeastern Idaho, where sagebrush steppe is the predominant vegetation. Point-interception and density-frequency data since 1985 were used to evaluate vegetation trends at 89 long-term vegetation plots established in 1950, 14 which have burned since 1994. Line-interception, shrub belt transects, and big sagebrush cross-section sampling methods were used to measure spatial and temporal patterns of big sagebrush establishment in four older fires. Precipitation is having a greater influence than the recent fires on the cover and species richness of perennial grasses, perennial forbs, and annuals and biennials, and native forb diversity. In response to fire, big sagebrush cover and density has been reduced to zero and green rabbitbrush (*Chrysothamnus viscidiflorus* (Hook.) Nutt.) cover and density has increased several years after fire. Long-term vegetation plots that were not burned have exhibited declines in both big sagebrush and green rabbitbrush cover. Declines in the cover of unburned big sagebrush are likely attributable to the widespread big sagebrush mortality in the Intermountain West, including the INEEL, during the 1970's and 1980's. Spatial patterns of post-fire big sagebrush establishment followed the prevailing wind gradient in at least one fire. Unburned patches of big sagebrush were probably the greatest factor consistently affecting post-fire big sagebrush establishment patterns. The results from this study also indicate that nearly a century is required for full post-fire big sagebrush recovery at the INEEL. Temporal patterns of big sagebrush establishment are occurring cyclically at approximately 10 year intervals. Precipitation during the fall and one year prior to germination are major factors influencing successful big sagebrush establishment, although the La Niña phenomena and black-tailed jackrabbit (*Lepus californicus* Gray) population cycles may also be synergistically influencing big sagebrush establishment patterns. Precipitation is a driving force in sagebrush steppe and other arid lands, influencing vegetation cover, density, species diversity, species richness, and recruitment patterns. Slow post-fire recovery rates of big sagebrush indicate that species more tolerant to fire, such as green rabbitbrush, provide a critical role in the post-fire community until big sagebrush is able to become established.

ACKNOWLEDGEMENTS

I am grateful to many people for their assistance in this research project. I appreciate the guidance provided by my committee, Dr. Stephen Bunting, Dr. Karen Launchbaugh, and Dr. Gerald Wright. I am grateful for Dr. Bunting's patience, encouragement, and resourcefulness throughout the duration of this project. A special thanks to the staff at the S.M. Stoller Corporation, especially Dr. Roger Blew and Amy Forman, who provided logistical support, as well as advice and background knowledge that improved this research project. My deepest appreciation goes to Dr. Jay Anderson, Dr. Karl Holte, and other researchers who have studied the long-term vegetation plots at the Idaho National Engineering and Environmental Laboratory (INEEL). This research project would not have been possible without the foresight, ingenuity, and perseverance of these predecessors. I am grateful to Dr. Anderson for providing background information for the long-term vegetation study and INEEL, and for stimulating new ideas.

Many technicians assisted throughout the duration of this research project, and I appreciate their energy, hard work, and dedication. Thank you to Steven Albano, Lucy Gammett, Marika Kearsley, and Georjanna Pokorney. I am especially grateful to Georjanna Pokorney, who provided field assistance during both the 2001 and 2002 field seasons. Thank you to Dr. Holte and Brian Schuetz, who verified identification of plant specimens. Thank you to Dr. Richard Folk, Dr. Tom Gorman, and Jan Pitkin of Forest Products for training me and allowing me to use their industrial belt sander. Thank you to Dr. Kenneth Newman and Dr. Kirk Steinhorst of Statistics for statistical consultation. Thank you to Kathy Mallory and Eva Strand for their technical support and positive attitudes. Finally, I am grateful to Tim Weekley for his friendship and support—to him I give the greatest thanks.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER I

INTRODUCTION

Big sagebrush (*Artemisia tridentata* Nutt.) occupies vast expanses of treeless, arid lands in the western United States. Big sagebrush is a native, late-seral shrub species that forms the dominant community structure throughout many of these areas, commonly described as sagebrush steppe. Many sagebrush steppe communities have become highly fragmented, largely due to anthropogenic disturbances, agricultural conversion, introduction of invasive annuals and biennials, and the increasing frequency of large wildfires (Noss et al. 1995, Knick and Rotenberry 1997, Knick 1999). The loss of sagebrush steppe has negative repercussions for biodiversity because it provides habitat and forage to many species in an otherwise inhospitable environment (Maser et al. 1984, Moritz 1988, Updike et al. 1990, Noss et al. 1995, Knick and Rotenberry 1997, Sveum et al. 1998).

Fires in sagebrush steppe are stand-replacing, and post-fire establishment of big sagebrush may occur on the scale of decades or centuries, depending on the big sagebrush subspecies (Harniss and Murray 1973, Wright and Bailey 1982). Fires serve an important role by increasing nutrient availability, creating space for seedling establishment, and enhancing habitat complexity across the landscape (Bunting 1996). Fires shift the dominance from big sagebrush to early- or mid-seral species that are better adapted to fire effects and the post-fire environment (Humphrey 1984). Secondary succession progresses slowly in the most arid regions of the sagebrush steppe distribution and fires may become imprinted on the landscape for centuries (Anderson et al. 1996).

Understanding vegetation dynamics and post-fire recovery of sagebrush steppe will improve our knowledge of how to best manage these communities. Our objectives for this research were the following: 1) Continue monitoring of long-term vegetation plots and evaluate vegetation dynamics after several recent, large fires; 2) Study spatial patterns of post-fire big sagebrush establishment within several older fires; 3) Study temporal patterns of big sagebrush establishment in relation to environmental influences.

LITERATURE REVIEW

Big Sagebrush Taxonomy and Distribution

Basin big sagebrush (*Artemisia tridentata* Nutt. spp. *tridentata*) and Wyoming big sagebrush (*A. t. wyomingensis* Beetle & Young) belong to the order Asterales, family Asteraceae. Four additional subspecies of big sagebrush occur in the United States, including: big sagebrush (*A. t. parishii* (Gray) Hall & Clements), big sagebrush (*A. t. spiciformis* (Osterhout (Kartesz & Gandhi))), mountain big sagebrush (*A. t. vaseyana* (Rydb.) Beetle), and foothill big sagebrush (*A. t. xericensis* Winward ex R. Rosentreter & R. Kelsey). All plant nomenclature comes from the USDA Plants Database (USDA, NRCS 2002).

Basin big sagebrush is the tallest of all the big sagebrush subspecies, normally growing 1.2-m to 1.8-m tall, and can attain heights of 2.4-m (Tisdale and Hironaka 1981). The main stem of basin big sagebrush usually remains a single trunk with age (Winward 1970). The leaves are generally longer than 2 cm and leaf margins are straight. Basin big sagebrush grows at elevations of 1220 to 2410 m in deep, well-drained soils where there is 250 to 460 mm of annual precipitation. Basin big sagebrush is distributed from Washington east to Montana and south to California and New Mexico (Winward 1970, Barker and McKell 1983, Bunting et al. 1987, Welsh et al. 1987).

Wyoming big sagebrush grows 0.4 to 1.0 m tall (Tisdale and Hironaka 1981). The main stem of Wyoming big sagebrush usually separates into two to three segments as the shrub ages (Winward 1970). The leaves are approximately 1.2 cm long (Welsh et al. 1987) and the leaf margins are curved outward (Winward 1970). Wyoming big sagebrush grows at elevations of 1525 to 1980 m in fine-textured soils that normally receive 180 to 310 mm of annual precipitation. Wyoming big sagebrush is distributed from Wyoming and Idaho south to Colorado (Beetle 1960, Barker and McKell 1983, Bunting et al. 1987, Welsh et al. 1987). Basin and Wyoming big sagebrush frequently intergrade where their ranges overlap (Barker and McKell 1983, Shumar and Anderson 1986, Welsh et al. 1987).

Big Sagebrush Life History and Reproduction

Big sagebrush reproduces by seed upon reaching maturity at two to four years of age, and does not resprout after fire or other disturbances. Big sagebrush forms inconspicuous, yellow flowers between August and September, and the seeds mature between late October and early November. Mature individuals annually produce greater than 300,000 seeds that are dispersed by wind, water, and animals (Daubenmire 1975, Tisdale and Hironaka 1981, Welsh et al. 1987, Young and Evans 1989, Taylor 1992).

Big sagebrush seeds are small—1.25 mm long and 0.5 mm wide (Chambers 2000). Seed dispersal is strongly correlated with the direction of the prevailing winds, particularly near edges in big sagebrush cover, where plants experience higher wind velocities (Goodwin 1956, Young and Evans 1989, Wagstaff and Welch 1990). Seeds generally disperse within 30 m of the parent plant, which limits the ability of big sagebrush to disperse into large unvegetated areas, such as those produced by wildfire (Goodwin 1956, Johnson and Payne 1968, Daubenmire 1975, Knick and Rotenberry 1997). Basin and Wyoming big sagebrush seeds can remain in the seedbank up to four years (Tisdale and Hironaka 1981), but are generally short-lived and do not survive fires (Young and Evans 1989). Successful big sagebrush seedling germination occurs in pulses, or widespread recruitment events, and depends on years of elevated precipitation levels. Recruitment pulses are common to many plants of arid lands (West et al. 1979, Maier et al. 2001). Cohorts of big sagebrush occur during years when multiple plants survive beyond germination and become established (Daubenmire 1975, Young and Evans 1989, Maier et al. 2001, Perryman et al. 2001).

Big Sagebrush Age

The well-defined annual growth rings on big sagebrush can be used to determine age accurately (Keeley 1993). The rings are formed by the development of a secondary xylem layer during spring when high precipitation levels promote the highest growth rates. An interxylary cork layer also forms between the new growth ring and that of the previous year, which suberizes the growth of the previous year, and clearly delimits each xylem layer (Ferguson 1964).

Ring width is correlated with annual precipitation levels and can be used to compare shrub cross-sections. Missing rings may occur during years of low rainfall, and false rings may occur if significant precipitation occurs after summer drought conditions (Ferguson 1964, Keeley 1993). False and entirely missing rings have not yet been found on big sagebrush growing in northerly latitudes and higher elevations (Ferguson 1964, Perryman and Olson 2000, Maier et al. 2001).

Height of big sagebrush tends to be more associated with site conditions and subspecies than the age of the plant. In addition, as big sagebrush ages, the main trunk tends to split into multiple branches, reducing the shrub’s height (Ferguson 1964, Daubenmire 1975). Splitting of the main trunk and increased decay with big sagebrush maturity prevents accurate age determination in some older individuals, which may affect age distributions (Roughton 1972, Daubenmire 1975).

Big Sagebrush Demographics

Demographic patterns of big sagebrush are influenced by disturbances that open up space for new recruitment sites (Owens and Norton 1992), and years of high precipitation that promote favorable conditions for pulse events (Cawker 1980, Maier et al. 2001, Perryman et al. 2001). Disturbances may include fire, herbivory, flooding, and insect and disease outbreaks (Perryman et al. 2001, Wambolt and Hoffman 2001). The oldest age class in a big sagebrush stand may indicate the length of time since the last fire occurred (Roughton 1972, Daubenmire 1975), due to the stand replacing character of fires in big sagebrush. Fires leave their imprint on arid lands for long time periods (Anderson et al. 1996) and recovery of big sagebrush may take many decades (Daubenmire 1975, Wambolt et al. 1999, Wambolt et al. 2001).

Big sagebrush can live more than 100 years (Ferguson 1964), although Wambolt and Hoffman (2001) found that the mean age of dead mountain big sagebrush in southwestern Montana was 41 years. Mountain big sagebrush has a mean stand age of 32 years in the northern Yellowstone region (Wambolt and Hoffman 2001) and 26 years throughout the state of Wyoming. In Wyoming, Wyoming big sagebrush has a mean stand age of 32 years, and basin big sagebrush has a mean stand age of 17 years. Demographic data from the same study indicate that basin big sagebrush also has a more frequent recruitment interval than Wyoming big sagebrush (Perryman et al. 2001).

Fire Ecology

Fire intervals in basin and Wyoming big sagebrush communities are variable and range between 20 to 100 years (Houston 1973, Wright and Bailey 1982, Wambolt et al. 2001). These estimates are based on analysis of fire scars on trees growing adjacent to big sagebrush communities (Houston 1973) and the length of time resprouting shrubs are able to survive without successive fires (Wright and Bailey 1982). Perennial grasses growing with big sagebrush provide discontinuously spaced fuels, making it difficult for fires to spread great distances (Whisenant 1990, Peters and Bunting 1994). Of the two subspecies, basin big sagebrush has a shorter mean fire interval due to the higher productivity and greater fuel loads of perennial grasses in these systems. The more arid, less productive habitats of Wyoming big sagebrush often do not produce enough fuels to carry a fire except under more extreme fire conditions (Bunting et al. 1987).

Fires are stand-replacing events in big sagebrush communities and recovery may take a long time under such arid conditions. Green rabbitbrush (*Chyrsothamnus viscidiflorus* (Hook.) Nutt.) and rubber rabbitbrush (*Ericameria nauseosa* (Pallas ex Pursh) Nesom & Baird), native shrubs that resprout after fire, often form the dominant vegetation community well after a wildfire has occurred in areas formerly inhabited by big sagebrush (Wright and Bailey 1982, Bunting et al. 1987, Wambolt et al. 1999). Given enough time without fire, big sagebrush eventually regains its dominance over green rabbitbrush (Harniss and Murray, Wright and Bailey 1982, Humphrey 1984).

Native perennial grasses of sagebrush steppe have variable responses to fire, depending on fire severity, season of fire, and growth form of plant. Leafy bunchgrasses, such as needle-and-thread grass (*Hesperostipa comata* (Trin. & Rupr.) Barkworth) are generally most susceptible to fire, particularly if the fire occurs early in the summer while the plant is still growing (Wright and Klemmedson 1965, Wright and Bailey 1982). Other perennial grasses, such as bottlebrush squirreltail (*Elymus elymoides* (Raf.) Swezey), are less susceptible to fire because they are composed of coarser materials and have less foliage, preventing smoldering and damage to plant tissues. Bottlebrush squirreltail is one of the few native grasses that can successfully compete with cheatgrass (*Bromus tectorum* L.; Hironaka and Tisdale 1963).

Cheatgrass

Cheatgrass, an introduced winter annual from Eurasia, competes for resources by starting vegetative growth earlier in spring than other plants and by rapidly proliferating after fires (Stewart and Hull 1949, Young and Evans 1978). The dense accumulation of dried cheatgrass tillers in mid-summer creates conditions suitable for subsequent fires. In some areas, fire frequency has increased to less than five years, and the fire season has been lengthened in areas formerly inhabited by sagebrush steppe (Whisenant 1990, Peters and Bunting 1994). Cheatgrass invades big sagebrush most readily where some type of disturbance produces the initial conditions for cheatgrass establishment. These disturbances may include roads and other transportation corridors, excessive livestock grazing, construction, agriculture, and off-road vehicle use (Young and Evans 1978, Peters and Bunting 1994, Knick and Rotenberry 1997, Knick 1999, Stylinski 1999). Excessive livestock grazing may increase shrub density and reduce perennial grass density and diversity (Whisenant 1990), creating openings for cheatgrass and exacerbating post-fire recovery of perennial grasses.

Cheatgrass-dominated fires are larger, more intense, and less patchy than historical fires. They homogenize and simplify community composition and vertical structure (Whisenant 1990, Peters and Bunting 1994), affecting plant and animal species that inhabit and depend on sagebrush steppe (Wambolt et al. 2001). In Idaho, many sagebrush steppe communities are highly fragmented because of the continuous nature of cheatgrass (Knick and Rotenberry 1997). Big sagebrush cannot re-establish once cheatgrass takes over because big sagebrush seedlings are killed by the larger, more frequent fires before they can mature and reproduce (Young and Evans 1978, Whisenant 1990). Re-establishment of big sagebrush becomes unlikely as distances between big sagebrush remnants and burned areas increase due to larger fire sizes and greater connectivity of cheatgrass-dominated lands (Knick and Rotenberry 1997).

Restoration of Sagebrush Steppe

Basin big sagebrush in the Snake River Plain in Idaho is considered to be one most critically endangered ecosystems in the United States, as a result of conversion of more than 99% of basin big sagebrush to agriculture (Winward 1970, Noss et al. 1995). Cheatgrass, altered fire regimes, and overgrazing have detrimentally impacted the biodiversity and sustainability of sagebrush steppe. Loss of habitat is correlated with species extinction (Noss et al. 1995). Degradation and fragmentation of sagebrush steppe, particularly those of basin and Wyoming big sagebrush, negatively impact the plant and animal species that occupy these communities (Knick and Rotenberry 1995).

Greater sage grouse (*Centrocercus urophasianus* (Bonaparte)) has had widespread population declines associated with big sagebrush habitat loss (Maser et al. 1984). Sage grouse nesting sites are correlated with big sagebrush that has high vertical structure and perennial bunchgrass cover (Sveum et al. 1998). Many other species of birds inhabit big sagebrush communities, and have been affected by the fragmentation and loss of big sagebrush (Knick and Rotenberry 1995). Big sagebrush also provides forage and habitat for mule deer (*Odocoileus hemionus* (Rafinesque)), pronghorn antelope (*Antilocarpa americana*

Ord), elk (*Cervus elaphus* Erxleben), and Rocky Mountain bighorn sheep (*Ovis canadensis* Shaw; Moritz 1988, Updike et al. 1990, Welch and Criddle, unpublished data).

Restoration of sagebrush steppe is being used to prevent the further conversion of big sagebrush to cheatgrass and to preserve biodiversity, particularly after wildfires (Downs et al. 1995, Ratzlaff and Anderson 1995, Chambers 2000). Sagebrush steppe that retains adequate levels of native plant composition is less susceptible to cheatgrass and other introduced species after wildfires, and may be able to naturally reseed itself (Ratzlaff and Anderson 1995, Anderson and Inouye 2001). Restoration in this situation may be most effective if certain pressures, such as livestock grazing, are temporarily alleviated. Highly degraded big sagebrush communities require a more active approach to restoration (McIver and Starr 2001), although revegetation of cheatgrass-dominated lands in the Snake River Plain and the Great Basin has been largely unsuccessful (Whisenant 1990).

STUDY AREA

The study area is located primarily on the Idaho National Engineering and Environmental Laboratory (INEEL), situated along the Upper Snake River Plain in southeastern Idaho, USA (Fig. 1.1). Livestock grazing ceased on most of the INEEL land in 1950 and 1957. This is delineated by the grazing exclusion boundary shown in Fig. 1.1. Before 1950, the land within the study area was used as a Naval gunnery range during World War II, and it is unlikely that livestock grazing occurred (Harniss 1968, Harniss and West 1973). Part of the study area is located on lands managed by the Bureau of Land Management (BLM) and the Idaho Department of Lands (IDL), which are open to livestock grazing. Much of the INEEL-owned portion of the study area has remained relatively free of anthropogenic disturbances since 1950.

During the late-1870's, large herds of cattle were driven across trails within the study area (Harniss 1968). The grazing history within the study area before 1950 is largely unknown, but widespread use of the Snake River Plain by sheep and cattle since the late 1800's coupled with low perennial grass cover at the onset of this study has led multiple investigators to conclude that the study area was seriously overgrazed prior to 1950 (Harniss and West 1973, Anderson and Holte 1981, Anderson and Inouye 2001).

Extensive historical wildfires and bombing scars associated with the WWII-era Naval gunnery range are visible on aerial photographs. Many of the bombing scars were reseeded with crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.) during the 1950's and 1960's (McBride et al. 1978). Several large wildfires have occurred at the INEEL since 1994, dramatically altering the landscape and plant communities (Fig. 1.2).

The elevation is approximately 1500 m throughout much of the study area, with little overall elevation change except at Twin Buttes and Big Southern Butte, to the south and southeast, respectively, and the Lost River Range along the western boundary. The Lost River Range creates a rain shadow across the study area. The climate is continental and typified by hot summers with high evaporation rates and cold, snowy winters. The mean annual temperature and mean annual precipitation are 5.6ºC and 215 mm (Fig. 1.3), respectively (NCDC 2003 as accessed by INSIDE Idaho 2002). There are two main peaks of precipitation during the year, a larger peak in May and June and a smaller peak in December and January. The prevailing winds are predominantly southwesterly across the INEEL (Harniss 1968, Harniss and West 1973).

The INEEL has a complex geologic history and most of the soils come from silicic volcanic flows and Paleozoic sedimentary rocks of aeolian origin. Basalt is the primary material from the volcanic flows at the surface, and the depth to the basalt layers affects the vegetation composition and water-holding capacity over relatively short distances. The soils within the study area are dominated by orthidic Aridisols and include the following groups: sands and clays over basalt, exposed basalt layers, glacial and alluvial deposits, and irrigable soils composed of clays, sands, and humus (McBride et al. 1978, Shumar and Anderson 1986, Anderson and Inouye 2001).

Basin big sagebrush (*Artemisia tridentata* Nutt. ssp. *tridentata*) and Wyoming big sagebrush (*A. t. wyomingensis* Beetle & Young) form the dominant vegetative structure throughout the study area, and frequently hybridize in this transitional zone between the two subspecies (Shumar and Anderson 1986). Wyoming big sagebrush and the hybridization between basin and Wyoming big sagebrush are the most frequently occurring representatives of the big sagebrush subspecies. All big sagebrush subspecies sampled throughout this study were lumped together for the analysis component, and are collectively referred to as big sagebrush. Other dominant shrub species include: green rabbitbush (*Chrysothamnus viscidiflorus* (Hook.) Nutt.), rubber rabbitbrush (*Ericameria nauseosa* (Pallas ex Pursh) Nesom & Baird), shadscale saltbush (*Atriplex confertifolia* (Torr. & Frém.) Wats.), spineless horsebrush (*Tetradymia canescens* DC.), and three-tip sagebrush (*Artemisia tripartita* Rydb.). Perennial grasses grow in the shrub understory and commonly include: bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve), bottlebrush squirreltail (*Elymus elymoides* (Raf.) Swezey), crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.), Indian ricegrass (*Achnatherum hymenoides* (R. & S.) Barkworth), needle-and-thread grass (*Hesperostipa comata* (Trin. & Rupr.) Barkworth), Sandberg bluegrass (*Poa secunda* J. Presl.), thickspike wheatgrass (*Elymus lanceolatus* (Scribn. & Smith) Gould), and bluestem wheatgrass (*Pascopyrum smithii* (Rydb.) A. Löve). Perennial forbs comprise a smaller component of the shrub understory, and common species include: Hood's phlox (*Phlox hoodii* Rich.), freckled milkvetch (*Astragalus lentiginosus* Dougl.), basalt milkvetch (*Astragalus filipes* Torr.), and cushion buckwheat (*Eriogonum ovalifolium* Nutt.). Wildlife species found within the study area include: elk (*Cervus elaphus* Erxleben), mule deer (*Odocoileus hemionus* (Rafinesque)), pronghorn antelope (*Antilocarpa americana* Ord), greater sage grouse (*Centrocercus urophasianus* (Bonaparte)), black-tailed jackrabbit (*Lepus californicus* Gray), and many other bird and small mammal species (Anderson and Inouye 2001).

CHAPTER II

Vegetation Dynamics and Fire Effects in Long-term Vegetation Plots

INTRODUCTION

Big sagebrush (*Artemisia tridentata* Nutt.) occupies vast expanses of treeless, arid lands in the western United States. Big sagebrush is a native, late-seral shrub species that often forms the dominant community structure in these areas, commonly described as sagebrush steppe. Native forb, perennial grass, and other shrub species coexist with big sagebrush, and disturbances, such as fire, rapidly shift the dominance from big sagebrush to early- to mid-seral plant species. Fires serve an important role by increasing nutrient availability, creating space for seedling establishment, and enhancing habitat complexity across the landscape (Bunting 1996).

Wright and Bailey (1982) estimate that fire frequency of the most arid sagebrush steppe habitats could be greater than 100 years. Recovery of big sagebrush after fire or other disturbance requires long periods of time, and takes at least several decades before mature big sagebrush stands are able to return (Harniss and Murray 1973, Wright and Bailey 1982, Bunting et al. 1987, Wambolt et al. 1999). Early- to mid-seral species quickly reestablish a burned area by vegetatively resprouting, seed dispersal, germination of fire-tolerant species in the seed bed, or a combination of these mechanisms (Humphrey 1984). Green rabbitbrush (*Chrysothamnus viscidiflorus* (Hook.) Nutt.) and spineless horsebrush (*Tetradymia canescens* DC.) are shrub species that reinvade burned areas using vegetative resprouting and prolific seed dispersal from the resprouted plants, and infrequent disturbances maximize their reproductive capabilities and persistence. Bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve), bottlebrush squirreltail (*Elymus elymoides* (Raf.) Swezey), crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.), and Sandberg bluegrass (*Poa secunda* J. Presl) are relatively tolerant to burning because of their coarse stems and low proportion of leafy material. Thickspike wheatgrass (*Elymus lanceolatus* (Scribn. & Smith) Gould) and bluestem wheatgrass (*Pascopyrum smithii* (Rydb.) A. Löve) have rhizomes that promote rapid post-fire regrowth. Forbs are generally unaffected by fire as long as active growth is not occurring when they are burned (Wright and Bailey 1982, Humphrey 1984).

The post-fire environment is also optimal for the invasion and establishment of introduced annual and biennial species, particularly cheatgrass (*Bromus tectorum* L.). Cheatgrass, a winter annual originally from Eurasia, competes for resources by growing earlier in spring than other plants and by rapidly proliferating after fires (Stewart and Hull 1949, Young and Evans 1978). Dense accumulation of dried cheatgrass tillers in mid-summer increases fire susceptibility, decreasing the fire frequency to every three to five years in some areas formerly inhabited by sagebrush steppe (Young and Evans 1978, Whisenant 1990, Peters and Bunting 1994). Fires in cheatgrass-dominated vegetation homogenize and simplify the compositional and vertical community structure (Whisenant 1990, Peters and Bunting 1994), negatively affecting the diversity of plant and animal species that inhabit sagebrush steppe (Noss et al. 1995, Knick 1999, Wambolt et al. 2001).

Concerns about cheatgrass invasion into recently burned sagebrush steppe have prompted massive reseeding projects. However, multiple studies indicate that these reseeding projects are unnecessary where sagebrush steppe was in good condition before burning (West and Hassan 1985, Ratzlaff and Anderson 1995). Permitting the natural revegetation of native early- to mid-seral species is more practical and sustainable, particularly in arid environments where succession occurs on a longer temporal scale than the careers of land managers.

The goal of this research was to monitor vegetation trends of relatively pristine sagebrush steppe and its response to several recent, large fires at long-term vegetation plots within the study area. Our hypotheses were: 1) No overall change in cover, density, and richness of perennials compared to surveys since 1985; 2) An overall decrease in cover, density, and richness of annuals and biennials in 2001 compared to the 1995 survey, resulting from the exceptionally dry year in which the most recent study was conducted; 3) Increased native forb diversity at recently burned plots, reflecting the decreased dominance of shrubs at these plots; and 4) No overall change in cover, density, and richness of introduced annuals and biennials, specifically cheatgrass.

METHODS

Ninety-nine long-term vegetation plots were established in 1950 when what is currently known as the Idaho National Engineering and Environmental Laboratory (INEEL) opened in southeastern Idaho. This study was originally initiated to investigate the influence of atomic energy reactor plants in natural ecosystems (Harniss 1968). Harniss and West (1973) later concluded that reactor-produced radioactivity did not detectably affect the surrounding vegetation.

The plots are aligned along two perpendicular macrotransect lines running southwest to northeast and southeast to northwest, both radiating from what used to be the center of the reactor testing activity in 1950 (43º N, 112º W; labeled "2001 plots"; Fig. 1.1; Harniss 1968). The majority of the plots are systematically spaced at 1.6-km (1.0-mi) intervals. Plots located at the foothills of the Lost River Range, and across both Big Southern Butte and East Butte occur at shorter intervals, 0.4 to 0.8 km (0.25 to 0.50 mi) apart, to compensate for changes in vegetation composition due to greater elevation differences (Harniss and West 1973). While these plots are not randomly located, they continue to provide valuable information due to the large number of plots and the long duration of the study.

Although 99 plots were initially established, plots 61 to 65 were never sampled due to conversion to agriculture. Plots 58 and 59 were last sampled in 1950 and plot 60 was never sampled; these plots were also likely lost to agricultural conversion. Plots 12 and 16 were lost before 1965 when they were destroyed by road construction. Plot 6 was destroyed by a landslide on Big Southern Butte and was last sampled in 1985. In 1983 and 1990, plot 28 was not sampled due to its inaccessibility, although it has been sampled in latter years. The result is that 89 of the 99 original plots were sampled in 2001.

The plots are differentiated between core and peripheral plots. Core plots refer to the plots located within the livestock exclusion boundary (Harniss 1968), and there are currently 43 core plots. Peripheral plots refer to those located outside of the grazing exclusion boundary, and currently number 46. The peripheral plots include the plots located at Big Southern Butte, East Butte, and the Lost River Range foothills, and subsequently have plots at higher and more variable elevations and with less aridity than the core plots (Anderson and Inouye 1999, Anderson and Inouye 2001). Both the core and peripheral plots were sampled in 1950, 1957, 1965, 1975, 1985, 1995, and 2001. In addition, a subset of 35 or 36 of the core plots was sampled in 1978, 1983, and 1990.

The peripheral plots are not consistent with regards to the type of grazing animal and season of use, and may not be representative of grazing practices within all the peripheral plots. Some of the peripheral plots may have little or no recent history of livestock grazing. The heterogeneity of grazing practices suggests that the core and peripheral plots should not be considered as ungrazed versus grazed plots (Anderson and Inouye 2001).

Each plot is marked by a steel fence post and rock crib along the macrotransect line (Fig. 2.1). Attached to the steel fence post is a 2.5-cm round metal identifying tag labeled as "VEG TRANS" and the plot number (Anderson et al. 1978). In 1950, each plot consisted of two 15.24-m (50-ft) long transects parallel to the macrotransect line. The first transect was 15.24 m from the macrotransect line and the second transect was 4.57 m (15 ft) away from the first transect. In 1985, a third 20-m long transect was established 4.57 m from the second transect on all of the core plots and approximately one-half of the peripheral plots (plots 13 to 15, 17 to 57, and 71 to 98), for purposes described later in this section. Plots along the southwest to northeast macrotransect have transects located northwest from the macrotransect line. Plots along the southeast to northwest macrotransect line have transects located southwest from the macrotransect line. The transect origin and end points are marked by steel rebar posts (Fig. 2.1).

Vegetation sampling of the plots occurred during June, July, and August of 2001 and corresponded with the timing of vegetation sampling in previous years. A metal measuring tape calibrated in both centimeters and inches was tautly fastened between the steel rebar posts along each transect.

The first procedure was to take two black and white and two color photos at each plot, one of the 1 x 1-m photo plot (Fig. 2.1) and a landscape photo. From where the photo plot was photographed, a landscape photo was taken to capture the horizon and the endpoints of the two 15.24-m transects. A standard form was used to record the film roll and exposure number, film speed, date, time, weather conditions, photographer and the photographer's height, and any pertinent notes concerning the plot. In addition, a sketch was made of the plant species occurring within each photo plot.

Point-interception was used to estimate cover at all plots that had a third transect marked by steel rebar posts, and was measured on both the two 15.24-m transects and the 20-m transect. This method was first used in 1985 (Anderson and Inouye 1988), and was repeated in 1990 (core plots only), 1995 (Anderson and Inouye 2001), and 2001. Point-interception was initiated to allow an objective estimate of cover of all growth forms that more proportionately records their abundance (Anderson and Inouye 2001).

Point-interception involved using a point-frame, a 1.0 x 0.5-m sighting frame with 36 points spaced at 0.1-m intervals when viewed from above. The plant species or entity (e.g., bare ground) that the point intercepted was recorded. If multiple plant species were intercepted at the same point, both or all species were recorded. Other entities were recorded only if the point did not also intercept a plant species or formerly living entity (e.g., dead shrub). The point was counted for a shrub or forb species as long as it intercepted some part other than the inflorescence. The point was only counted for an annual or perennial grass species if it intercepted the basal area of the grass. In 2001, as a result of several large, recent fires that burned through several plots, a new category, “burnt stump”, was created to describe burnt stumps still rooted in the ground. Point-interception is further described and compared to other vegetation sampling methods in Floyd and Anderson (1982).

The point-frame was centered over the tape lengthwise and read at one-meter intervals (Floyd and Anderson 1982, Anderson and Inouye 2001). Point-interception began at the zero point of the tape for all three transects. Fifteen point-frames were read on each of the two 15.24-m (50-ft) transects, so that the last point-frame ended at the 15-m mark on the tape, just before the steel rebar post marking the endpoint. Twenty point-frames were read on the third 20-m transect line, and the last point-frame ended at the 20-m mark on the tape, where the steel rebar post was located. Point-interception was sampled before the other methods to avoid trampling or disturbance of the overstory, which was more critical to this method than the methods described later in this section.

The basal area of all perennial grasses and canopy of live and dead shrubs, trees, and prickly-pear cactus (*Opuntia polyacantha* Haw.) were recorded to the nearest 0.25 cm (0.10 in) using line-interception (Canfield 1941). Line-interception has been measured on all plots since 1950 and has only been read on the two 15.24-m transects. The field technician sampled vegetation by standing on the side of the transect closest to the macrotransect line, using a plumb bob that intercepted vegetation on the opposite side of the tape. Gaps in the canopy of a shrub were not counted. For example, a shrub was recorded continuously from the beginning to the end point of the same shrub, or continued to the next shrub if there was overlap between shrubs of the same species. Dead shrubs and prickly-pear cactus have not been consistently recorded in surveys prior to 2001, but should be recorded in future surveys. Dead shrubs had to have some part still rooted in the ground to be classified as dead shrub.

Density and frequency were sampled along each of the two 15.24-m transects, for a total of 20 quadrats read per plot. This method has been used to sample all plots since 1950, although density-frequency data for plot 82 is missing for the 1950 survey and for plots 86 and 88 to 93 for the 1957 survey (Anderson and Inouye 1999). In addition, the frequency data for 1995 is currently missing.

Density-frequency quadrats were located on the transect side opposite to the macrotransect line (Fig. 2.1). The first quadrat of each transect started at the 1.52-m (5-ft) mark along the tape and the last quadrat started at the steel rebar post at the transect endpoint. The quadrats were spaced at 1.52-m intervals, and a plumb bob was used to locate the beginning point of each quadrat along the tape. Each quadrat was 1.0 x 0.3 m and the shorter side was parallel to the tape. All perennials within the quadrat were counted if at least one-half of the root or basal area fell within the quadrat. All annuals and biennials were counted in a 0.3 x 0.1-m subsection within each quadrat. The 0.3 x 0.1-m section was located at the side closest to the tape and the 0.3-m side ran parallel to the tape (Harniss 1968, Harniss and West 1973). Previous reports and publications have stated that this annuals section was a 0.1 x 0.1-m subsection (Anderson and Inouye 1988, Anderson and Inouye 1999, Anderson and Inouye 2001), but verbal correspondence revealed that these were misprints and that the subsection has been 0.3 x 0.1-m (Dr. J.E. Anderson, personal communication). The size of the annuals subsection was increased from 0.3 x 0.1-m to 0.3 x 0.3-m in 1975 (Anderson et al. 1978), and may not have been accounted for in data analysis after 1978.

Quality control procedures were used during the 2001 field season to ensure calibration between technicians and to increase the precision of vegetation sampling. The first part of the quality control procedures consisted of Dr. J.E. Anderson spending a day with the field technicians at a practice plot and making sure the sampling techniques were properly performed. Then, the field technicians sampled the first three plots as a single group. Afterwards, the field technicians split into two two-person teams and re-sampled parts of every fourth plot for the next 16 plots, and then every 15th plot until sampling all of the long-term vegetation plots was completed.

After all vegetation sampling at a plot was complete, a coin toss was used to determine re-sampling techniques. The team with heads would re-sample the density-frequency quadrats that the other team had sampled. The team with tails would re-sample the other team’s transect using line-interception. Both teams would simultaneously re-sample four point-frames randomly selected from the 15 point-frames on the two shorter transects. The quality control data were compared to the original data at the end of each field day to evaluate and correct sampling inconsistencies between technicians. All four technicians would switch teams after each plot was completed to further reduce sampling inconsistencies. The final quality control procedure involved the field technicians collecting voucher plant specimens for all species found within the plots in 2001, as this was the first year that investigators from the University of Idaho sampled the plots. Dr. Karl E. Holte, Brian K. Schuetz, and Dr. Stephen C. Bunting verified the voucher specimens and assisted in the identification of unknown plant specimens.

Perennial grass species in the formerly classified genus *Agropyron* and species in the genus *Poa* may have been misidentified by investigators over the sampling period before 1975 (Harniss 1968, Anderson and Inouye 2001), so the native species within each genus should be pooled for data analysis including years prior to 1975. The current data analysis does not cover years prior to 1985, so *Agropyron* and *Poa* were not pooled.

We were interested in studying changes in cover, species richness, species diversity, and density between plots that had burned recently and plots that had not recently burned (Fig. 1.2). Since the inception of the long-term vegetation study, 14 plots have been burned in wildfire (Fig. 1.2). Plots 95, 96, and 97, on the western side of the INEEL outside of the grazing exclusion boundary, were burned in 1994 (1994 fire), and the soils at these plots are in the Simeroi-Sparmo and Malm-Bondfarm-Matheson Complexes. Plots 78 and 79 were burned in 1996 (1996 fire #1), and the soils at these plots are in the Coffee-Nargon-Atom Complex and Atom Silt Loam, respectively. Plot 78 is located outside the grazing exclusion boundary and plot 79 is within the boundary. Plots one to four, located south of Big Southern Butte and outside the grazing exclusion boundary, were burned in a different fire that also occurred in 1996 (1996 fire #2), and the soils at these plots are primarily in the Hondo Cobbly Loam, but also in the Orthids-Rubble-Rock Outcrop Complex. Plots three and four were burned previously in 1957. Plots one to four are the only plots in the study area where cheatgrass has become dominant; density-frequency data and photographic records indicate that cheatgrass has been ubiquitous in plots one to three since at least 1950. Plots 30, 31, 85, 86, and 87, located completely within the grazing exclusion boundary, burned in 2000 (2000 fire) and the soils at these plots are in the Malm-Bondfarm-Matheson Complex. Plot 31 was excluded from the data analysis because it had numerous tire tracks associated with fire suppression activities running across the plot.

Anderson and Inouye (2001) have comprehensively analyzed and described the vegetation trends that have occurred at these plots between 1950 and 1995, using line-interception and density data. The current study is unique because we primarily use point-interception data, available since 1985 (Anderson et al. 2001), and the density data, to compare vegetation trends between burned and unburned plots, and assess the contribution of the recent fires to overall vegetation trends within the study area. Species were pooled for analysis into the following species groups: shrubs, perennial grasses, perennial forbs, native annuals and biennials, and introduced annuals and biennials. Species of interest were evaluated separately and included big sagebrush, green rabbitbrush, and cheatgrass. Density data and both relative and absolute point-interception data were used to analyze the species and species groups. Relative data represents the proportion of a species or species group out of the total live plant cover intercepted. Absolute data represents the proportion of a species or species group out of the maximum number of points intercepted (e.g., includes bare ground, litter, etc.). Statistical analyses were run separately for plots burned in the 1994 fire, 1996 fire #1, 1996 fire #2 (density data only), 2000 fire, and for unburned plots. Euclidean distance was used to determine which unburned plots were most similar to plots burned in the 1994 fire, 1996 fire #1, and 2000 fire during the survey immediately prior to burning. PC-ORD Version 4.20 was used to determine species richness, Shannon-Weiner and Simpson diversity indices, and Euclidean distance plot similarity indices. SAS for Windows 8.02 was used to evaluate if there was a significant difference between the 1985, 1990, and 2001 surveys for species and species groups.

RESULTS and DISCUSSION

Precipitation

Total annual water-year precipitation was higher during the first two surveys, 1985 (247 mm) and 1995 (360 mm), than the mean annual precipitation (215 mm; Fig. 1.3; NCDC 2003 as accessed by INSIDE Idaho 2002). The period between the late-1980's and early-1990's was typified by several years of less than average total mean precipitation, followed by an extremely wet period in the mid-1990's. Subsequent years of above-average and average precipitation continued until 2000, initiating another droughty period. The final survey, 2001, was one of the driest years on record, and the total annual precipitation (128 mm) was 60% of the mean annual precipitation. Total spring precipitation (April-June) was 35, 196, and 26 mm for 1985, 1995, and 2001, respectively, greatly diverging from the mean spring precipitation (76 mm).

Cover—All Plots

Point-interception data were relativized to show the actual proportions of vascular plant species and species groups between 1985 and 2001 (Fig. 2.2). Shrubs continued to be the dominant cover type in all plots sampled with the point-interception method (N=58). Relative shrub cover declined in 1995 and then increased in 2001, although did not return to the same extent as 1985 ($P<0.001$). The decrease in shrub cover was coupled by an increase of perennial grasses ($P<0.001$), nearly doubling from 14% in 1985 to 26% in 2001. Based on line-interception data, Anderson and Inouye (2001) also documented an increase of perennial grasses at the core plots between 1985 and 1995. Before 1985, they found perennial grass cover had increased 13-fold between 1950 and 1975, but then decreased between 1975 and

1985. Perennial forb cover was greater in 1995 than 1985 (P=0.040), corresponding with trends in perennial forb density data described in Anderson and Inouye (2001). This increase in perennial forb cover may have been driven by high precipitation in the early- to mid-1990's, although perennial forb cover did not differ between 1995 and 2001. These results are inconsistent with our hypothesis that there would be no overall change in the cover of shrubs, perennial grasses, and perennial forbs since 1985. All three perennial growth forms have exhibited significant differences in overall cover between 1985 and 2001.

Abundant precipitation in 1995 created favorable conditions for both native and introduced annuals and biennials, which were both highly significant compared to 1985 and 2001 ($P<0.001$), but introduced annuals and biennials may have gained an edge over native species after 1995 (Fig. 2.2). The number of available sites for ephemeral species likely increased in 1995 due to the enhanced moisture conditions. Once introduced annuals and biennials established themselves and produced seed in 1995, they may have been able to retain their foothold through droughty conditions in 2001 better than their native counterparts. These results concur with our hypothesis that there would be an overall decrease in the cover of annuals and biennials in the 2001 survey since the 1995 survey. We had also predicted that there would be no overall change in the cover of introduced annuals and biennials, although our results indicate that they may displace native annuals and biennials in future surveys.

The decline in shrub cover has been driven by the loss of approximately half of the big sagebrush cover in 1995 and 2001 since 1985 ($P<0.001$; Fig. 2.2). Anderson and Inouye (2001) noted widespread big sagebrush mortality had occurred in the study area since 1975, and reported analogous trends based on the line-interception data. Similar patterns of big sagebrush mortality were also observed throughout the Intermountain West between the late 1970's and 1980's, but its causes are largely unknown (McArthur et al. 1990, Anderson and Inouye 2001). The second most dominant shrub, green rabbitbrush, was not different between 1985 and 2001 (P=0.386), inconsistent with the trends based on the line-interception data up to 1995 (Anderson and Inouye 2001). In 1995, shrub cover decreased in both the core plots (P<0.001) and peripheral plots (P=0.003), following the same pattern regardless of livestock grazing management. These results are contrary to our hypothesis that there would be no overall change in shrub cover in the core or peripheral plots since 1985.

Species Richness—All Plots

Species richness of shrubs, perennial grasses, perennial forbs, and native and introduced annuals and biennials was derived from the point-interception data (Fig. 2.3). The species richness of shrubs was similar between years (P=0.984). Species richness of perennial grasses was greater in 1995 than 1985 (P=0.007), and species richness of perennial forbs was greater in 1995 than either 1985 or 2001 (P<0.001). We had not expected that species richness would change for perennials; however, species richness for perennial grasses and forbs increased in 1995. The high precipitation in 1995 combined with an increased proportion of establishment sites due to widespread big sagebrush mortality may have allowed perennial grass and forb species to expand their distribution within the study area. Anderson and Inouye (2001) found that the species richness of shrubs, perennial grasses, and perennial forbs increased between 1950 and 1995, based on both the line-interception data (for shrubs and perennial grasses only) and the density-frequency data, and that species richness was increasing at a greater rate for perennial grasses than shrubs.

The decreased species richness of native and introduced annuals and biennials since 1995 (Fig. 2.3) concurs with our hypothesis. Species richness of both native and introduced annual and biennial species was greater in 1995 than in 1985 or 2001 ($P<0.001$), likely associated with the favorable precipitation in 1995. Anderson and Inouye (2001) found that the overall species richness of annuals and biennials, based on the density-frequency data, generally increased between 1950 and 1995, although annual and biennial species richness fluctuated in relation to precipitation patterns. Discrepancies with the annuals and biennials density-frequency data indicates care should be taken when interpreting their results (See *Chapter II: Methods*).

Density—All Plots

Mean density of perennials, annuals and biennials, and dominant shrub species between 1985 and 2001 for all plots sampled with the density-frequency method ($N=87$) is shown in Fig. 2.4. Consistent with our hypothesis, there were no significant trends in the density of shrubs ($P=0.757$), perennial grasses ($P=0.080$), and perennial forbs ($P=0.375$). Shrub cover decreased in 1995, even though their species richness and density did not change between 1985 and 2001. Big sagebrush density decreased from 1.1/m^2 in 1985 and 1995 to 0.5/m^2 in 2001 ($P=0.004$). Green rabbitbrush density was not different between surveys ($P=0.309$), although the standard error was much larger in 2001 than previous years, largely due to the increased role of disturbance within the study area. Anderson and Inouye (2001) found that big sagebrush densities have been declining since approximately 1975, and that green rabbitbrush densities have been increasing since the study was initiated in 1950. Shrub densities overall appear to be unchanged between 1985 and 2001, although major shifts have been occurring between the dominant shrub species within the time scale of the current study and since 1950 (Anderson and Inouye 2001). The cover and species richness of perennial grasses and forbs has been increasing, although density has remained unchanged. Perennial grasses and forbs are likely becoming larger in size, because their increased cover was not coupled with a greater number of individuals.

Native annuals and biennials were present in significantly reduced densities in 2001 compared to 1985 (P=0.009; Fig. 2.4). Despite high precipitation in 1995, the results indicate that native annuals and biennials are becoming sparser between 1985 and 2001. Densities of introduced annuals and biennials did not change between 1985 and 2001 (P=0.098). These results are generally consistent with our observations for both cover and species richness that introduced annuals and biennials are displacing the native annuals and biennials. However, the depressed density values in 1995 are counterintuitive because 1995 was the highest total annual precipitation year on record, and based on the point-interception data, annuals and biennials positively contributed to high cover and richness values in 1995. Past discrepancies in the size of the annuals and biennials subsection in 1975 (See *Chapter II: Methods*) lead us to suspect that the size of the annuals and biennials subsection was inconsistently reduced to 0.1 x 0.1-m in 1995, and perhaps additional years (Anderson and Inouye 1988, Anderson and Inouye 2001), potentially affecting the validity of the density data for annuals and biennials. Further inquiries should be made before full weight is given to the annuals and biennials density data.

Cover—Burned Versus Unburned Plots

We subdivided the plots from the point-interception analysis (1985-2001) into plots burned in the 1994 fire (N=3), 1996 fire #1 (N=2), 2000 fire (N=4), or into unburned plots (N=49), plots that have not been burned since the initiation of the study in 1950. Relative and absolute cover of species groups and common perennial species are shown in Figs. 2.5-2.6. The small sample size and time since fire may influence the significance for some of the results, but these results show the overall differences and similarities between the burned and unburned plots.

Similar trends in relative shrub cover occurred between plots burned in the 1996 fire #1 and unburned plots, although only the unburned plots exhibited reduced relative shrub cover in 1995 ($P<0.001$; Fig. 2.5). Relative shrub cover at the 1994 fire decreased from 84% in 1985 to 5% in 1995, and has since increased to 30% in 2001 ($P=0.002$). Relative shrub cover at the 2000 fire decreased from 50% in 1985 to 36% in 1995, and then dropped to 2% in 2001 ($P=0.004$). Absolute shrub cover generally depicts the same trends for all groups, although declines observed in the 1995 relative shrub cover of the unburned plots are not significant for the absolute cover data ($P=0.056$; Fig. 2.6).

The relative cover of big sagebrush decreased since 1985 at the 1994 fire ($P<0.001$), 1996 fire #1 ($P=0.029$), and 2000 fire ($P=0.007$; Fig. 2.5). These results were expected because big sagebrush does not resprout after fire or other disturbance (Humphrey 1984). Despite excluding the burned plots, relative cover of big sagebrush at the unburned plots declined from 39% in 1985 to 20% in 1995, and has since risen to 27% ($P<0.001$). Absolute cover of big sagebrush also decreased since 1985 at 1994 fire ($P<0.001$), 2000 fire ($P=0.049$), and the unburned plots ($P=0.004$), although there were no differences in absolute cover of big sagebrush in plots burned in the 1996 fire #1 ($P=0.063$; Fig. 2.6). Absolute cover of big sagebrush at the unburned plots showed a continuing decline from 9.9 to 7.3 to 6.0% during the 1985, 1995, and 2001 surveys, respectively. These results indicate that big sagebrush cover remains in a declining stage, even without fire. The disproportionately high cover of annuals and biennials in 1995 followed by low cover in 2001 likely skewed the relative cover data so that shrub and big sagebrush cover appeared to be increasing since 1995, even though big sagebrush cover is still decreasing at the unburned plots. Declines in big sagebrush cover at the unburned plots are likely attributable to the widespread big sagebrush mortality mentioned previously.

Relative cover of green rabbitbrush in the unburned plots has remained extremely constant since 1985, and has varied between 21% and 23% (P=0.791; Fig. 2.5). Absolute cover of green rabbitbrush, however, shows that green rabbitbrush cover declined at the unburned plots in 2001 since 1995 (P=0.029; Fig. 2.6). Green rabbitbrush cover did not change at any of the burned plots for either relative or absolute cover data, although plots burned in the 1994 fire and 1996 fire #1 showed an increasing trend in green rabbitbrush cover since they were burned. Relative cover of green rabbitbrush at the 2000 fire decreased from 13% in 1985 to 11% in 1995, and then dropped to 2% in 2001 (P=0.397). Green rabbitbrush is an early- to mid-seral shrub that vegetatively resprouts after fire or other disturbance. Resprouted green rabbitbrush, stimulated by the fire, produce prolific amounts of seed after one to two years have passed (Young and Evens 1974). Unlike other early successional species, green rabbitbrush is also able to coexist with late-seral species, such as big sagebrush, although it becomes less dominant as time since fire increases (Humphrey 1984). These results suggest that the green rabbitbrush cover at the burned plots will continue to increase for many decades until its growth slows down and big sagebrush reestablishes itself in the community. Declines in the absolute green rabbitbrush cover at the unburned plots concur with related studies that green rabbitbrush requires disturbance to stimulate growth and reproduction (Young and Evans 1974, Humphrey 1984, Harniss and Murray

1973). Concomitant reductions in both absolute big sagebrush and green rabbitbrush cover at unburned plots indicate there was an increased availability of resources in 2001. Droughty conditions in 2001 may have exacerbated resource availability, as no other species appeared to benefit from the declines in the absolute cover of big sagebrush and green rabbitbrush. Future surveys, particularly in conjunction with favorable precipitation, may show a shift in the absolute cover of other species groups at the unburned plots.

Relative cover of perennial grasses at plots burned in the 1996 fire #1 followed a similar trend as the unburned plots, although only the unburned plots had greater relative perennial grass cover in 2001 than 1985 ($P=0.031$; Fig. 2.5). Absolute cover of perennial grasses at the unburned plots was different between all three surveys (1995>2001>1985; $P<0.001$), suggesting that the prolific growth of annuals and biennials in 1995 has also skewed the relative cover of perennial grasses (Fig. 2.6). In addition, although not significant, absolute perennial grass cover at the burned plots is following a similar trend as the unburned plots. Growth of perennial grasses seems to be affected more by favorable precipitation than fire. Relative perennial grass cover increased after fire, though weakly, at the 2000 fire ($P=0.050$). Comparing the relative and absolute perennial grass cover data reveals that post-fire perennial grass cover has not increased due to fire. The relative proportion of perennial grass cover has increased after fire, particularly at the 1994 fire and 2000 fire. The relative proportion of perennial grass cover at the 1996 fire #1 has also increased, although not to the same extent, likely due to the increased proportion of green rabbitbrush at these plots. Contrary to other studies (Harniss and Murray 1973, West and Hassan 1985), these results suggest that perennial grass cover does not increase after fire in sagebrush steppe for plots burned one, five, or seven years ago. In southwestern Montana, Wambolt et al. (2001) also found that fire did not substantially affect perennial grass cover in sagebrush steppe. Burning sagebrush steppe to increase livestock forage (Harniss and Murray 1973, West and Hassan 1985) may not necessarily produce the desired results (Wambolt et al. 2001).

Relative perennial forb cover was higher in 2001 than 1985 at only the plots burned in the 1994 fire (P=0.027), increasing from 8% in 1985 to 24% after the fire in 1995, and continuing to 31% in 2001 (Fig. 2.5). The absolute cover data shows that perennial forbs increased in 1995 compared to either 1985 or 2001 for both the unburned plots (P<0.001) and the plots burned in the 1996 fire #1 (P=0.019; Fig. 2.6). Fluctuations in perennial forb cover was likely driven more by the favorable precipitation in 1995 than fire. At the 1994 fire, relative perennial forb cover seems to be increasing despite the influence of favorable precipitation, and may be more related to fire. Perennial forbs are tolerant of fire in the late-summer or fall when the plant is no longer actively growing, and are typically unaffected by wildfires as long as the plant has ceased growth (Wright and Bailey 1982). Land use history, soils, time since fire, and microsite moisture availability may be driving the disparity between the plots burned in the 1994 fire and the remaining plots. The plots burned in the 1994 fire are outside the livestock grazing exclusion boundary, have soils different from other plots, and supports dense stands of Great Basin wildrye (*Leymus cinereus* (Scribn. & Merr.) A. Löve), an indicator of moist site conditions (Welsh et al. 1997).

The relative cover of native annuals and biennials was highly significant at the 1996 fire #1 (P<0.001) and unburned plots (P<0.001; Fig. 2.5). At the 1996 fire #1, native annuals and biennials cover increased from 2% in 1985 to 15% in 1995, and then decreased to nearly zero in 2001. At the unburned plots, native annuals and biennials cover rose from 4% in 1985 to 12% in 1995, then also dropped to nearly zero in 2001. Although not significant, native annuals and biennials cover jumped from nearly zero in 1985 to 17% in 1995, falling to nearly zero again in 2001 (P=0.192). Native annuals and biennials cover fluctuated in response to high precipitation in 1995 at nearly all burned and unburned plots, with the exception of the plots burned in the 2000 fire. Native annuals and biennials at the 2000 fire followed a different trend that was not significant between survey years (P=0.944), decreasing from 23% in 1985 to 16% in 1995, and then rising again to 20% one year after the fire. Neither the high precipitation that occurred in 1995 nor the fire affected the abundance of native annuals and biennials cover at the 2000 fire.

At the 1996 fire #1, the relative cover of introduced annuals and biennials cover increased from zero in 1985 to less than one percent in 1995 and 2001 (P=0.012); however, this miniscule increase is likely not biologically significant (Fig. 2.5). Although not significant, introduced annuals and biennials cover at the 1994 fire rose to 12% in 1995 from zero in 1985, and then fell to less than one percent in 2001 (P=0.434). The newly established presence of introduced annuals and biennials in these burned plots warrants close monitoring in the future. Prior to burning, introduced annuals and biennials were already established in the plots burned in the 2000 fire, and comprised nearly 9% of the relative cover in the droughty year of 2001 (P=0.246). Introduced annuals and biennials cover at the unburned plots increased from 1% in 1985 to 9% in 1995, and dropped again to 3% in 2001 (P<0.001). High precipitation years such as 1995 produce favorable conditions for ephemeral species to initially establish and reproduce. Wildfires are most common in dry years that follow wet years (Swetnam and Betancourt 1990, Knapp 1995, Miller and Rose 1999), so if a wildfire occurs the year after a high precipitation year, there is plentiful seed stored in the seed bed to take advantage of the post-fire conditions. The wet-dry cycle of wildfire ignitions might amplify the seedbank of introduced annuals and biennials, ensuring the abundant seed availability at the best possible time, immediately after a wildfire (Young and Tipton 1990).

Cheatgrass, an aggressive winter annual, is the dominant factor in most of the patterns of the introduced annuals and biennials. However, there were no significant patterns of cheatgrass cover occurring in any of the burned plots (Fig. 2.5). At the unburned plots, cheatgrass cover was higher in 1995 than 1985, and 2001 was not different than either year (P=0.013). At the 1994 fire, cheatgrass was minutely present only in 1995. Cheatgrass has never been observed at the 1996 fire—the increase in introduced annuals and biennials noted previously was derived from other species. At the 2000 fire, cheatgrass cover increased from less than one percent in 1985 to 18% in 1995, and made up 4% of the cover one year after the fire. High precipitation appears to be the driving force in cheatgrass abundance, and its presence prior to fire is related to its continued presence after fire. Cheatgrass appears to pose a greater problem at the plots burned in the 2000 fire than at the other burned plots, although it is still a relatively small part of the post-fire plant community.

Cover—Similarity Between Burned and Control Plots

Plots burned in the 1994 fire (N=3), 1996 fire #1 (N=2), and 2000 fire (N=4) were matched up to the same number of control plots, unburned plots most similar in absolute cover to the burned plots during the survey prior to burning. Plots 27, 84, and 92 were most similar to plots burned in the 1994 fire (Fig. 2.7). Plots 84 and 93 were most similar to plots burned in the 1996 fire #1 (Fig. 2.8). Plots 27, 34, 44, and 53 were most similar to plots burned in the 2000 fire (Fig. 2.9). The small sample size or time since fire may influence the significance for some of the results, but these results show the overall differences and similarities between burned and unburned plots that had strong similarities prior to burning.

Absolute shrub cover declined in 2001 since 1985 and 1995 at the plots burned in the 1994 fire and 2000 fire. Absolute shrub cover did not change between survey years at the plots burned in the 1996 fire #1 or at any of the control plots (Figs. 2.7-2.9). Absolute big sagebrush decreased in 2001 since 1985 at the 1994 control plots (P=0.047), and absolute green rabbitbrush cover did not change (P=0.526). There were no significant trends in either absolute big sagebrush or green rabbitbrush cover at the 1996 fire #1 and 2000 control plots. Consistent with previously described results, big sagebrush declined in plots without fire. Plots without fire did not exhibit any changes in green rabbitbrush cover, although they appear to be declining since 1995, consistent with the results for all unburned plots. Both the plots burned in the 2000 fire and its control plots are following nearly identical trends in green rabbitbrush cover. Green rabbitbrush does not substantially increase until one to two years after disturbance (Young and Evans 1974). Plots burned in the 2000 fire will likely diverge from the control plots within several years, and exhibit increases in green rabbitbrush cover.

Absolute perennial grass cover was relatively unaffected by fire, and there were no significant trends between years for either the burned or control plots (Figs. 2.7-2.9). These results concur with earlier statements that fire does not necessarily increase perennial grass cover. Relative and absolute perennial grass cover has been increasing between 1985 and 2001, at all unburned plots, and at all plots together. Factors other than the recent, large wildfires are contributing to the increased perennial grass cover across the study area. Recent, favorable precipitation and livestock grazing exclusion across much of the study area may be influencing the increases in perennial grass cover (Anderson and Inouye 2001). Absolute perennial forb cover was also rather unaffected by fire (Figs. 2.7-2.9). Absolute perennial forb cover was greater for both the plots burned in the 1996 fire #1 (P=0.019) and its control plots (P=0.011). The plots burned in the 1994 fire and 2000 fire and their control plots did not have any significant trends in perennial forb cover between surveys. Like perennial grass cover, favorable precipitation in 1995 appears to be the greatest factor influencing fluctuations in perennial forb cover.

Absolute cover of native annuals and biennials was higher during the 1995 survey than in 1985 or 2001 for plots burned in the 1994 fire (P=0.028) and its control plots (P=0.008; Fig. 2.7). Plots burned in the 1996 fire #1 (P=0.019) and its control plots (P=0.003) had greater absolute cover of native annuals and biennials in 1995 than in 1985 or 2001 (Figs. 2.8). There were no significant differences between years for plots burned in the 2000 fire (P=0.578) or its control plots (P=0.239; Fig. 2.9). Absolute cover of introduced annuals and biennials was not significant between years for the plots burned in the 1994 fire (P=0.433), although it was greater in 1995 than other years for the 1994 control plots (P=0.005; Fig. 2.7). Absolute cover of introduced annuals and biennials was greater in 1995 than other years at the 1996 fire #1 (P<0.001), although there were no significant differences between survey years at the 1996 fire #1 control plots (P=0.269; Fig. 2.8). Absolute introduced annuals and biennials cover was not different between survey years at either the 2000 fire (P=0.367) or its control plots (P=0.178; Fig. 2.9). There were no differences in absolute cheatgrass cover between surveys for any of the burned or control plots. There are no clear trends concerning the influence of fire on the absolute cover of introduced annuals and biennials, although this may be related to the relatively low abundance of cheatgrass across much of the study area. Favorable current year precipitation is driving the high abundance of native and introduced annuals and biennials. Site conditions at the plots burned in the 2000 fire and its control plots may be causing the dissimilarities between these plots and the other burned and unburned plots, due to the minimal response of annuals and biennials to the high precipitation in 1995.

Native Forb Diversity—Burned Versus Unburned Plots

Point-interception data were used to compare native forb diversity between burned and unburned plots. Native forb diversity was higher at unburned plots in 1995 than either 1985 or 2001 for both the Shannon-Weiner index ($P<0.001$) and modified Simpson index ($P<0.001$; Fig. 2.10). Native forb diversity was much higher at all of the burned plots in 1995 than the other surveys, but was not significant, likely associated with the small sample sizes. The Shannon-Weiner and modified Simpson indices similarly measure species diversity, although the modified Simpson index increases the importance of dominant species (Magurran 1988). We had originally hypothesized that native forb diversity would be greater in the 2001 than 1995 for plots burned since 1995, but precipitation has been more influential to native forb diversity than fire.

Density—Burned Versus Unburned Plots

We subdivided the plots from the density analysis (1985-2001) into plots burned in the 1994 fire ($N=3$), 1996 fire #1 ($N=2$), 1996 fire #2 ($N=4$), 2000 fire ($N=4$), or into unburned plots ($N=74$), plots that have not been burned since the initiation of the study in 1950. Significant differences in big sagebrush density between survey years were observed at plots burned in the 1994 fire ($P<0.001$) and 1996 fire #1 ($P=0.011$), not unexpected because big sagebrush does not resprout after fire (Fig. 2.11). Big sagebrush density at the plots burned in the 1994 fire decreased from 2.4/m^2 in 1985 to zero in both 1995 and 2001. At the plots burned in the 1996 fire #1, big sagebrush density was zero in 2001 after decreasing from 2.1/m² in 1985 and 1.6/m² in 1995. At the unburned plots, big sagebrush density also declined from 1.0/m² in 1985 and 1.1/m² in 1995 to 0.6/m² in 2001 ($P=0.036$). This decline in big sagebrush density at the unburned plots is likely due to the widespread big sagebrush mortality that also decreased big sagebrush cover in the point-interception data. At the plots burned in the 1996 fire #2, big sagebrush density was only 0.21/m² in 1985 and 0.17/m² in 1995 before declining to zero in 2001. Based on 1950 density-frequency data, we know that cheatgrass has dominated these plots since at least 1950, impeding successful big sagebrush establishment. Although not significant, big sagebrush density at the 2000 fire declined from 3.4/m² in 1985 to 1.4/m² in 1995, before dropping to zero in 2001 ($P=0.240$). The reduction of big sagebrush density by 2.0/m² suggests that many big sagebrush individuals had senesced in these plots prior to burning.

At the plots burned in the 1996 fire #1, green rabbitbrush density increased from 2.8/m² in 1985 and 1.9/m² in 1995 to 42.0/m² in 2001 ($P=0.024$; Fig. 2.11). This pronounced increase in green rabbitbrush density was only observed at the plots burned in the 1996 fire #1—none of the other plots, burned or unburned, showed any differences or trends between surveys. Increased green rabbitbrush density is often expected to occur a few years after a fire in sagebrush steppe (Young and Evans 1974). The lack of increased green rabbitbrush density at plots burned in the 1994 fire signifies that there are other factors, such as site conditions, determining species dominance during early successional stages. Although green rabbitbrush was present at the 1996 fire #2, it was at such low densities that vegetative resprouting, seed production, and seedling establishment has not occurred at nearly the same extent as the other plots—therefore, green rabbitbrush has retained a minimal role in these cheatgrass-dominated plots. Green rabbitbrush density at the 2000 fire was only 0.5/m² in both 1985 and 1995, and decreased to 0.4/m² in 2001. The pre-fire green rabbitbrush density was much lower at the 2000 fire than at the 1996 fire #1, and may reflect the proportion of available green rabbitbrush propagules available for post-fire seed production. We are not certain to what extent green rabbitbrush will become reestablished at the plots burned in the 2000 fire, based on the divergent patterns seen at the 1994 fire and 1996 fire #1.

There were no significant differences in cheatgrass density for any of the plots (Fig. 2.11), although the cheatgrass density data would have been affected by the inconsistencies in the size of the annuals and biennials subsection, as previously mentioned. We had hypothesized that cheatgrass density would not change since past surveys, but it is difficult to accurately ascertain a conclusion due to sampling inconsistencies—the following is an interpretation based on the current data set. Cheatgrass density appeared to remain exceptionally stable at the 1994 fire, 1996 fire #1, and at the unburned plots, although cheatgrass cover was higher in 1995 than 1985 at the unburned plots, as mentioned previously. Plots burned in the 1996 fire #2 dropped from 508.3/m² in 1985 to 124.2/m² in 1995, and then rebounded to 682.1/m² in 2001. Although the 1996 fire #2 likely did positively influence the cheatgrass density, it seems improbable that cheatgrass density would be so low during one of the wettest years on record. Regardless of the inconsistencies in the annuals and biennials subsection, plots burned in the 1996 fire #2 are presenting much different patterns in cheatgrass density than the rest of the plots. These plots are functionally and structurally dissimilar than the other plots, and have crossed a threshold into a cheatgrass-dominated state (Friedel 1991, Laycock 1991). At the 2000 fire, cheatgrass density increased from 133.3/m² in 1985 to 197.5/m² in 1995, but was only 15.8/m² in 2001, right after the fire. Cheatgrass does not appear to be a large component of the post-fire introduced annuals and biennials group at the 1994 fire, 1996 fire #1, and 2000 fire.

CONCLUSIONS

More than the recent wildfires, precipitation is having a greater influence on the cover and species richness of perennial grasses, perennial forbs, native annuals and biennials, and introduced annuals and biennials, and native forb diversity on the INEEL. Infrequent periods with high precipitation, such as during the mid-1990's, drive temporal fluctuations in cover, diversity, and recruitment for many species and species groups (Westoby 1979). Introduced annuals and biennials cover was greatest in 1995, simultaneously peaking with native forb diversity. Sequences of highly favorable or unfavorable precipitation can influence vegetation dynamics and species composition for many years or decades in arid environments (Westoby 1979, Griffin and Friedel 1985, Goldberg and Turner 1986, Austin and Williams 1988, Friedel et al. 1993).

Both absolute and relative perennial grass cover have followed an increasing trend between 1985 and 2001, although absolute perennial grass cover was greatest in 1995. Perennial grass cover substantially increased after the initial 1950 survey, probably due to the cessation of livestock and sheep grazing within much of the study area, although has generally fluctuated in response to precipitation in subsequent surveys (Anderson and Inouye 2001). Absolute perennial grass cover was not affected by burning, supporting similar findings in southwestern Montana that burning sagebrush steppe does not increase post-fire perennial grass cover (Wambolt et al. 2001).

The recent wildfires have substantially affected the cover and density of the two dominant shrub species, big sagebrush and green rabbitbrush, and have dramatically decreased vertical structure across the landscape. The cover of big sagebrush, a shrub that does not vegetatively resprout after fire, has been reduced to zero at plots that have burned within the past decade. Most of the recent wildfires have been extremely large, and unburned patches of big sagebrush within the wildfires will be crucial to post-fire big sagebrush establishment (Frischnecht and Bleak 1957, Johnson and Payne 1968, Wambolt et al. 1999, See *Chapter III: Results and Discussion*). Post-fire green rabbitbrush cover and density has generally increased, except at the 2000 fire, although the response has been widely variable. Green rabbitbrush becomes prolific about one to two years after fire and remains persistent for many decades (Harniss and Murray 1973, Young and Evans 1974, See *Chapter III: Results and Discussion*), and green rabbitbrush at the 2000 fire will likely show considerable growth in future surveys.

The absolute cover of both big sagebrush and green rabbitbrush has declined at the unburned plots since 1985. Anderson and Inouye (2001) found that overall relative big sagebrush cover and density has been on a decreasing trend since the 1975 survey, and attributed this decline to widespread big sagebrush mortality at the INEEL and across the Intermountain West. However, they also found that overall relative green rabbitbrush cover, but not green rabbitbrush density, has been increasing since the 1975 survey. We found that, based on the point-interception data, overall relative green rabbitbrush has remained relatively unchanged between 1985 and 2001. The differences in our results may be ascribed to the larger number of plots used, decreased temporal scale, or use of the point-interception data. Point-interception is generally more a more precise measurement of the true proportion of species cover in shrublands than the line-interception method (Floyd and Anderson 1982), so the results in the current study probably have greater sensitivity to change for the 1985 to 2001 time period. Based on our results, green rabbitbrush has been increasing at plots burned in the recent wildfires, but has been decreasing in the absence of fire. Green rabbitbrush, an early- to mid-seral species, is able to persist in late-seral big sagebrush communities, but becomes less dominant as time increases since the last fire (Harniss and Murray 1973, Young and Evans 1974, Humphrey 1984).

Big sagebrush cover decreased prior to burning at the 1996 fire #1 and 2000 fire. At the unburned plots, big sagebrush cover has similarly declined since at least 1985. Synergism between widespread big sagebrush mortality, increased herbaceous cover in the mid-1990's associated with above-average precipitation, and the pairing of dry years after wet years likely enhanced the continuity of fuels and the susceptibility of the sagebrush steppe to fire, resulting in several large wildfires at the INEEL (Swetnam and Betancourt 1990, Knapp 1995, Miller and Rose 1999). The association between large wildfires and prolific growth of introduced annuals and biennials with wet and dry years implies that introduced annuals and biennials are able to maximize the seedbank at the optimal time, the year preceding a year with synchronous wildfires (Young and Tipton 1990).

Due to the persistence of standing dead big sagebrush and the continued decline of big sagebrush and green rabbitbrush cover on unburned plots, large areas within the study area may be highly prone to fire if there is an appropriate combination of future climatic events. Climate change predictions suggest that fires will become more frequent in the western United States, creating substantial implications for post-fire vegetation dynamics (Delcourt et al. 1983, Romme and Turner 1990, Sprugel 1991, Bartlein et al. 1997). Many interacting factors have the potential to shift the role of introduced annuals, particularly cheatgrass, in relatively healthy stands of sagebrush steppe in the western United States (Peters and Bunting 1994, Knick 1999).

Sagebrush steppe provides vertical structure and habitat for many plant and animal species in an otherwise inhospitable environment (Tisdale and Hironaka 1981, Maser et al. 1984, Moritz 1988, Updike et al. 1990, Knick and Rotenberry 1995, Sveum et al. 1998). Interactions between altered disturbance regimes, introduced annuals, and anthropogenic pressures have resulted in the conversion of extensive tracts of sagebrush steppe in the western United States to cheatgrass-dominated grasslands (Knick and Rotenberry 1997, Knick 1999). Monitoring both vegetation trends and climate records is essential to understanding sagebrush steppe communities.

CHAPTER III

Post-Fire Spatial Patterns of Big Sagebrush Establishment

INTRODUCTION

Big sagebrush (*Artemisia tridentata* Nutt.) occupies vast expanses of treeless, arid lands in the western United States. Big sagebrush is a native, late-seral shrub species that often forms the dominant community structure in these areas, commonly described as sagebrush steppe. Habitat loss and degradation of big sagebrush due to agricultural conversion, excessive livestock grazing, cheatgrass invasion, and altered fire regimes is threatening the ecological integrity and biodiversity of these ecosystems (Noss et al. 1995, Knick 1999). Spatial patterns of big sagebrush establishment after large disturbances are not well-known and most studies date from the "range improvement" era when the primary purpose was to increase short-term livestock forage capacity through big sagebrush eradication (Mueggler 1956, Frischknecht and Bleak 1957, Johnson and Payne 1968). Recovery of big sagebrush after wildfires or other disturbances requires long time periods, often taking many decades or a century before mature stands of big sagebrush are able to return, depending on big sagebrush subspecies (Harniss and Murray 1973, Wright and Bailey 1982, Bunting et al. 1987, Wambolt et al. 1999).

Big sagebrush produces very small, wind-dispersed seeds that disperse up to 33 m away from the parent plant (Goodwin 1956, Mueggler 1956, Young and Evans 1989). The majority of big sagebrush seeds are short-lived and usually do not persist in the seed bank beyond the first growing season (Mueggler 1956, Young and Evans 1989, Meyer 1994). Big sagebrush establishment is associated with sequences of favorable precipitation prior to germination, and widespread, cyclical recruitment pulses may only occur once a decade (See

Chapter IV: Results). Big sagebrush age is determined by counting the number of annual growth rings (Ferguson 1964), and is well-correlated with stem diameter (Perryman and Olson 2001).

Several studies indicate that big sagebrush seed dispersal is greatest on the windward side of the parent plant, following the direction of the prevailing winds (Mueggler 1956, Johnson and Payne 1968, Wagstaff and Welch 1990). Prevailing winds have an influential role in both seed dispersal and seed arrival, and generally translate into directional, nonrandom landscape patterns of adult establishment (Auspurger and Franson 1987, Schupp 1995, Schupp and Fuentes 1995). Wind dispersal of seeds has been strongly linked with spatial establishment patterns for many plant species, including sweet birch (*Betula lenta* L.; Matlack 1989), Russian thistle (*Salsola tragus* L.; Stallings et al. 1995), and plomo (*Tachigalia versicolor* Standley & L.O. Williams; Augspurger and Kitajima 1992). Spatial patterns of seed arrival, also known as seed shadows, generally have the greatest seed densities closest to the parent plant and reduced densities with increasing distances from the parent plant (Young and Evans 1989, Wagstaff and Welch 1990, Schupp and Fuentes 1995).

The few studies that have explored the relationship between big sagebrush seed dispersal from adjacent, unburned stands and spatial establishment patterns after large disturbances have generally had inconclusive results (Mueggler 1956, Johnson and Payne 1968). Surviving big sagebrush individuals within burned or disturbed areas are believed to provide the most rapidly available seed source after a stand-replacing disturbance (Mueggler 1956, Johnson and Payne 1968, Meyer 1994).

The goal of this research was to compare four fires in a sagebrush steppe predominantly inhabited by Wyoming and basin big sagebrush hybridizations and answer the following questions: 1) What is the post-fire age structure of big sagebrush along the prevailing wind gradient within several older fires?; 2) Is post-fire big sagebrush establishment greater in the southwestern than the northeastern area of each fire?; 3) Are there any consistent spatial patterns of big sagebrush establishment between the four fires?; and 4) How does perennial grass cover respond to shrub densities representing different seral stages?

METHODS

Aerial photographs from 1949, 1954, 1976, 1982, and 1991 were used to identify and map fires that burned within the study area before 1994. The Quick and Dirty Image Reference Version 1.0 extension (Poston 2000) for ArcView 3.2 was used to trace the outline of fires from digitized aerial photographs. Four fires were selected for the study, and were chosen for similarity in soils, topography, and plant composition within each fire and between fires; these represent different time periods and fire sizes (Fig. 3.1).

The Seven-Mile Road Fire (Fig. 3.2) was ignited by a lightning strike and occurred during June 1985. This fire burned approximately 125 ha and exhibited a high degree of patchiness, likely due to the presence of rocky fissures and the early season of the fire. The soils are of the Malm-Bondfarm-Matheson Complex and occur as sands over basalt. The Fire Station Fire (Fig 3.3) is listed as a controlled fire that occurred during September 1974, burning about 71 ha. This fire occurred in a floodplain, and the soils are Typic Torrifluvents. The B & B County Fire (Fig. 3.4) is not dated, but appears on 1949 aerial photographs and probably occurred between 1910 and 1949. This fire burned 1345 ha and its tail end stops at the Principle Lineament, a very large fire that may have occurred in the early 1800's (Anderson et al. 1996, Dr. J.E. Anderson, personal communication). The soils are predominantly loess, of the Coffee-Nargon-Atom Complex, although the soils towards the northeastern tip of the fire are of the Malm-Bondfarm-Matheson Complex. The Tractor Flats Fire (Fig 3.5) occurred in 1910 and was 3696 ha. The soils are of the Aecet-Outcrop Rock Complex, Berreniceton-Terreton-Rock Outcrop Complex, and Pancheri Silt Loam throughout the majority of the fire, and occur as sands over basalt or loess (Moritz 1988).

All sampling for this study occurred in June, July, and August of 2002 and the associated plots are labeled “2002 plots” in Figs. 1.1-1.2 and Figs. 3.1-3.5. One macrotransect was established linearly across each fire from southwest to northeast (Figs. 3.2-3.5) to account for the wind gradient. The macrotransect width was 140 m for all four fires. The first plot on both ends of the macrotransect started 50 m within the fire perimeter. At the 50-m mark, a plot was randomly located along the macrotransect width using a random point generator. From this point, distance intervals varied depending on fire size. Plots within the two smaller fires, the Seven-Mile Road Fire and Fire Station Fire, were placed at 50-m intervals for the next four plots in either direction, and then at 100-m intervals or until the plots merged in the middle of the macrotransect. Plots within the two larger fires, the B & B County Fire and Tractor Flats Fire, continued at 200-m intervals from the first plot for the next three plots. After the fourth plot from either direction, plots would be similarly selected at 400-m intervals until the plots merged in the middle of the macrotransect. Plots were not used if there was a rocky fissure, soils less than 10 cm in depth, evidence of crested wheatgrass seedings, or an unburned patch, or if there was a road, railroad track, or evidence of a historical sheep camp within the plot. If any of these occurred, a substitute plot would be randomly selected. If the plot was still unacceptable, another substitute plot was randomly selected, sometimes just outside of the macrotransect width or at a wider distance interval. Control plots were established at least 200 m outside of the fire perimeter. The control plots were also randomly selected along the same macrotransect as the main plots, except where unacceptable site conditions prevented this. The control plot associated with the southwestern perimeter of the Tractor Flats Fire could not be sampled due to extensive crested wheatgrass seedings, and was relocated to the east of the fire perimeter. The substitute control plot that was sampled was in an area that had big sagebrush mortality due to flooding, and had site conditions atypical of the other plots; thus, this control plot was excluded from statistical analysis.

Each plot consisted of two 50-m transects spaced 10 m apart that were parallel to the macrotransect. Transects were sampled from the southwest to northeast direction. Two horizontal landscape photos were taken down the transect lines of each plot from both the southwest and northeast directions. Cover of shrubs and perennial grasses was measured on both transects using the line-interception method (Canfield 1941).

We developed a gauge to estimate big sagebrush age using the maximum stem diameter at the base of the shrub (Fig 3.6). Several studies indicate that big sagebrush age and stem diameter are correlated (Winward 1970, Perryman and Olson 2000). I used age prediction equations for basin big sagebrush from southwestern Wyoming (Perryman and Olson 2000) to determine the stem diameter sizes that correspond with chosen age classes. Using the age classes of 0 to 10, 10 to 20, 20 to 30, and greater than 30 years, the stem diameter size classes were 0 to 13, 13 to 44, 44 to 75, and greater than 75 mm, respectively. The gauge was inexpensively constructed from a Plexiglas sheet and allowed quick determination of stem diameter size class to estimate the age class of big sagebrush.

Belt transects were used to count the number of live and dead big sagebrush, green rabbitbrush, and other shrubs at each plot. The stem diameter size class of all live big sagebrush was also recorded for this tally. All shrubs rooted within one meter of both transects were used to determine the overall shrub density of each plot (Bunting et al. 1987, Bonham 1989). Shrub demographics within the smaller fires, the Fire Station Fire and Seven-Mile Road Fire, were assessed with histograms showing density of the big sagebrush size classes and overall shrub density across the length of each macrotransect. The same analysis was performed for the two larger fires, the B & B County Fire and Tractor Flats Fire, using SAS for Windows 8.02. Analysis of variance and simple linear regression were used to compare correlation of each size class across the length of the macrotransect from the southwest to northeast direction. The same procedures were done on the total distribution of big sagebrush and green rabbitbrush across each of the larger fires. In addition, I ran an analysis of variance for a two-way factorial without interaction to determine the significance of plot and size class within the larger fires. Line-interception data were compared to the densities of big sagebrush and green rabbitbrush using simple linear regression. Thickspike wheatgrass (*Elymus lanceolatus* (Scribn. & Smith) Gould) and bluestem wheatgrass (*Pascopyrum smithii* (Rydb.) A. Löve) were pooled for analysis because of their similar growth form and life history.

Three big sagebrush cross-section sampling points were established at the 0-, 25-, and 50-m points along each transect, for a total of six per plot. The closest big sagebrush of each of the four size classes to the point was sampled, and the maximum distance from the point was 5-m. Sampling representatives of each big sagebrush size class included measuring the height and stem diameter of the big sagebrush, and subsequent removal of the cross-section.

Cross-section sampling and analysis follow methods previously used for big sagebrush (Ferguson 1964, Perryman and Olson 2000, Maier et al. 2001, Perryman et al. 2001, Wambolt and Hoffman 2001). The cross-sections were cut with a saw or clippers at ground level so that the pith and first annual growth ring would be included. The second cut was made 10 cm above the first cut, so that each cross-section was 10 cm long. Sampling of deteriorated big sagebrush (i.e., damaged stem, dead) was avoided because their age is less likely to be accurately assessed. The bottom portion of each big sagebrush cross-section was sequentially sanded with 100, 150, and 320 grit sanding belts. Annual growth rings were counted through a dissecting scope and cross-dated to maintain accuracy in counting the growth rings.

The cross-section data were used to perform the following analyses, so that they could be extrapolated to the belt transect data. Stem diameter was the dependent variable and age and height, separately and together, were the independent variables in simple and multiple linear regressions, to assess which relationship is the best age predictor. The stem diameter and age regression equation was used to predict the true big sagebrush age class intervals for each of the size classes. An analysis of variance and Tukey's Studentized Range Test were used to test the variance and significance between size class and age.

RESULTS AND DISCUSSION

Predicting Big Sagebrush Age from Stem Diameter and Height

Stem diameter proved to be a better predictor of big sagebrush age than height before and after natural log transformation (Figs. 3.7-3.8). Natural log transformation improved the coefficient of determination between stem diameter and age from $R^2=0.68$ to $R^2=0.81$. Natural log transformation improved the coefficient of determination between height and age from $R^2=0.58$ to $R^2=0.72$. All correlations were significant ($P<0.001$). The coefficient of determination between stem diameter and age are consistent with results in a statewide big sagebrush demographic study in Wyoming (Perryman and Olson 2000).

The oldest big sagebrush found was at least 120 years old; approximately 10 years of its pith was missing, so this individual was about 130 years old. Interestingly, the same plot, the control plot southwest of the B & B County Fire, also had six additional long-lived big sagebrush individuals ranging in age from 73 to 96 years old. Big sagebrush is known to live for up to 200 years (Ferguson 1964), but it is relatively uncommon to find big sagebrush at ages greater than 80 years old (Perryman et al. 2001).

Table 3.1 shows the big sagebrush size classes used to measure stem diameter, the original age classes used to develop the size classes, and the new age classes that were determined from the regression equation between age and stem diameter (after natural log transformation). The new age classes were 0-13.5, 13.5-29.1, 29.1-42.9, and greater than 42.9 years for the associated size classes of 0-13, 13-44, 44-75, and greater than 75-mm, respectively. The age range for the new age classes was greater than predicted by the original age classes that were based on big sagebrush in southwestern Wyoming.

Box plots depicted in Fig. 3.9 show the relationship between big sagebrush age and each stem diameter size class without natural log transformation. The stem diameter size classes correspond well with their prediction of big sagebrush age ($R^2=0.64$; $P<0.001$). Tukey’s Studentized Range Test determined that all age and stem diameter size class pairwise relationships were different from each other. Although all pairwise relationships were different from each other, the 45-75 and 75+ mm stem diameter size classes show some overlapping of age prediction, likely because of the tendency of big sagebrush stems to rot with increasing age. Many older big sagebrush individuals had missing piths, so one could only determine that they were very old and had attained some minimum age. Reducing the number of size classes or the minimum size of the oldest size class would likely improve the predictive ability of the big sagebrush size class gauge. We would not suggest increasing the minimum size of the oldest size class because of increased variance and reduced predictive ability with increasing big sagebrush age.

These results indicate that the big sagebrush size class gauge (Fig. 3.6) is an effective, non-destructive tool to quickly assess big sagebrush age. The big sagebrush size class gauge is probably most accurate and predictable for similar stands in southeastern Idaho with basin and Wyoming big sagebrush hybridization. However, the technique could be used in sagebrush steppe throughout the western United States if big sagebrush were concurrently sampled to develop an accurate baseline regression equation.

Post-Fire Spatial Patterns of Vegetation—Seven-Mile Road Fire and Fire Station Fire

After 17 and 28 years, big sagebrush has not substantially returned to either the Seven-Mile Road Fire or Fire Station Fire (Figs. 3.11-3.12). The greatest total live big sagebrush density for any plot in either of these two fires was 11/200 m², a small fraction of the mean total live big sagebrush cover in the control plots (355/200 m²; Fig. 3.10). Although the Fire Station Fire was much smaller and 11 years older than the Seven-Mile Road Fire, the two fires have relatively equivalent levels of big sagebrush establishment. The presence of big sagebrush in the 29-43 year size class in both fires indicates that these individuals survived the fires in unburned patches. The Seven-Mile Road Fire appears to be very patchy on both aerial photographs and in the field, likely due to the fire's relatively early occurrence in June and the widespread presence of rocky fissures within the study area. The higher proportion and wider distribution of big sagebrush in the 29-42 year size class appears to have improved the probability of big sagebrush establishment on the Seven-Mile Road Fire, relative to the Fire Station Fire.

Post-fire big sagebrush establishment at these two fires was too small for statistical analysis. Visual inspection indicates that big sagebrush establishment is not necessarily associated with distance from the fire perimeter edges, although wind dispersal of big sagebrush seeds along the gradient of the prevailing winds may be occurring along the southwestern edge of the Seven-Mile Road Fire (Fig. 3.11). However, the limited dispersal range of big sagebrush seed (Goodwin 1956) does not account for the presence of big sagebrush distributed within the fires.

In southwestern Montana, Johnson and Payne (1968) studied the rate of big sagebrush establishment after 20 control treatments that entailed either herbicide application or plowing to remove big sagebrush from large areas. They found that big sagebrush that were not killed were important seed sources for revegetating the disturbed sites and greatly influenced the rate of post-disturbance big sagebrush recovery. Big sagebrush seed dispersal from sagebrush steppe adjacent to the treated areas did occur, although inconsistently. The authors concluded that surviving big sagebrush within the treated areas had a faster and more widespread effect than adjacent sagebrush steppe in the revegetation of big sagebrush across large areas. These results are consistent with our own results that unburned patches of big sagebrush are important to post-fire establishment of big sagebrush. Seed dispersal from adjacent areas may have some importance, although it may be negligible across very large fires or shorter time scales (Johnson and Payne 1968). Decreasing the distance between a seed source and a recently burned safe site improves the rate and probability of successful big sagebrush establishment (Knick and Rotenberry 1997).

Green rabbitbrush was present in most plots across the Seven-Mile Road Fire, but occurred at high densities at only three plots (Fig. 3.12). Green rabbitbrush was well-distributed at relatively high densities across the Fire Station Fire. Fires stimulate vegetative growth and seed productivity of green rabbitbrush, allowing this species to rapidly increase after fire (Young and Evans 1974, Humphrey 1984). We are uncertain why the post-fire green rabbitbrush densities were so limited at most plots at the Seven-Mile Road Fire, but it may be related to pre-fire green rabbitbrush abundance, fire intensity, or soils. At the INEEL, post-fire big sagebrush establishment appears to be on the scale of many decades and rapid post-fire green rabbitbrush establishment may serve a critical intermediary role by providing vertical structure, reducing soil erosion, and preventing a shift to cheatgrass during such a susceptible time period (Whisenant 1999).

Post-Fire Spatial Patterns of Vegetation—B & B County Fire and Tractor Flats Fire

After at least 53 years, big sagebrush establishment is relatively limited within the B & B County Fire, although big sagebrush was present at every plot (Figs. 3.11-3.12). In Dubois, Idaho, Harniss and Murray (1973) found that 30 years were required for mountain big sagebrush to return to its pre-fire abundance. Clearly, well over 53 years are required for big sagebrush to return to its pre-fire abundance at the INEEL. Longer post-fire recovery periods should be expected for Wyoming big sagebrush (and related hybridizations) than mountain big sagebrush because Wyoming big sagebrush is distributed at more arid sites (Wright and Bailey 1982, Bunting et al. 1987). Only three of the 17 plots have total big sagebrush densities at or above total mean big sagebrush density at the control plots (Fig. 3.10). One plot, located 1400 m from the southwestern edge of the fire, had a high number of big sagebrush in the 0-13 year size class (523/200 m^2), although the proportion of big sagebrush in the older size classes at this plot was average compared to the other plots (Fig. 3.11). Favorability of site conditions can be temporally dynamic, and the last major recruitment event (See *Chapter IV: Results*) may have generated conditions that were temporarily better at one site than another site (Grubb 1977, Eldridge et al. 1991, Schupp and Fuentes 1995). Plots located within the northeastern part of the fire appear to have relatively abundant stands of big sagebrush, possibly due to increased patchiness observed in the tail ends of fires as they lose momentum and become patchier.

After 92 years, big sagebrush has completely established within the Tractor Flats Fire (Figs. 3.11-3.12), and only one plot had a lower big sagebrush density (208/200 m^2) than the control plots (Fig. 3.10). Densities of the 0-13 year size class were astronomically high throughout the Tractor Flats Fire (averaging 556/200m^2; Fig. 3.11), and more than half the plots had densities of this size class that were greater than twice the density of the control plots (Fig. 3.10). This suggests that many safe sites are currently available for big sagebrush seedlings (Grubb 1977) and that the big sagebrush stands are still actively recruiting new individuals. Long-term survival of these big sagebrush seedlings may depend on their proximity to adult big sagebrush, intraspecific competition, and the availability and size of gaps (Owens and Norton 1989, Reichenberger and Pyke 1990, Meyer 1994). In central Utah, long-term big sagebrush seedling survival was more likely in seedlings with larger resource areas than those with smaller resource areas (Owens and Norton 1989).

Green rabbitbrush densities are uniformly high across the B & B County Fire (Fig. 3.12), and all plots had green rabbitbrush densities at least three times those at the control plots (Fig. 3.10). Green rabbitbrush densities were much greater than big sagebrush densities at nearly every plot. Despite abundant big sagebrush establishment, green rabbitbrush densities were also very high across the Tractor Flats Fire (Fig. 3.12). Some plots had green rabbitbrush densities matching those found at the control plots (Fig. 3.10), although there were a few plots with green rabbitbrush densities more than five times those found at the control plots. In addition, big sagebrush and green rabbitbrush densities were nearly equivalent throughout much of the Tractor Flats Fire (Fig. 3.12). Interestingly, the plot with the greatest density of big sagebrush in the 0-13 year size class (1675/200 m^2) also had the highest densities of green rabbitbrush (1098/200 m^2), indicating highly favorable local site conditions. It was unexpected that green rabbitbrush densities would remain so high at a fire that occurred 92 years ago. Nearby in Dubois, Idaho, Harniss and Murray (1973) found that green rabbitbrush returned to its pre-burn abundance after approximately 30 years, although spineless horsebrush, another resprouting shrub, persisted. We are not sure why green rabbitbrush continues to be so productive at both of these older fires because green rabbitbrush generally requires infrequent disturbances to stimulate vegetative growth and improve seed productivity (Young and Evans 1974).

Spatial analysis was performed for both of these larger fires using big sagebrush size class and distance across fire in a two-way factorial analysis without interaction. Size class remained highly significant at both fires ($P<0.001$). Distance across fire was not significant at the B & B County Fire ($P=0.534$), but was significant at the Tractor Flats Fire ($P=0.0184$). The same analysis was repeated using only the closely spaced plots located at either the southeastern or northeastern edges of each fire perimeter. At the B & B County Fire, plots at the southeastern edge (0 to 600 m) and northeastern edge (3750 to 4550 m) were not significantly different from each other ($P=0.378$, $P=0.265$). At the Tractor Flats Fire, plots at the southeastern edge (0 to 600 m) and northeastern edge (3400 to 4000 m) were also not significantly different from each other ($P=0.188$, $P=0.302$).

Simple linear regression was also performed to analyze the trend of each big sagebrush size class across the B & B County Fire (Fig. 3.13) and Tractor Flats Fire (Fig. 3.14). None of the size classes were different at the B & B County Fire. At the Tractor Flats Fire, the 0-13 ($P=0.032$), 29-43 ($P=0.005$), and 43+ year ($P=0.0014$) size classes were all different; the 13-29 year size class was not different ($P=0.1074$). The distribution of the size classes at the Tractor Flats Fire were all skewed with the highest big sagebrush densities occurring at the southwestern edge of the fire. Simple linear regression was also used to compare trends between total live and dead big sagebrush and green rabbitbrush across the two fires (Figs. 3.15-3.16). Again, there were no significant relationships for total live or dead big sagebrush or green rabbitbrush at the B & B County Fire. Both total live ($P=0.0346$) and dead ($P=0.0213$) big sagebrush had the greatest densities at the southwestern edge of the Tractor Flats Fire. Neither live nor dead green rabbitbrush was significantly distributed across the Tractor Flats Fire.

Obviously, two very different patterns are occurring at the B & B County Fire and Tractor Flats Fire. As hypothesized, big sagebrush establishment is greatest at the southwestern edge of the Tractor Flats Fire. Contrary to our hypothesis, there are no spatial patterns of big sagebrush establishment at the B & B County Fire. These results indicate that wind dispersal of big sagebrush seeds from adjacent unburned areas is not consistently influencing the spatial patterns of big sagebrush establishment. While both fires have relatively similar sizes, the Tractor Flats Fire appears to be older than the B & B County Fire; comparisons of the age structures across each fire indicate that the Tractor Flats Fire is probably about 20 to 30 years older than the B & B County Fire. Also, the sampling scale may not have been sufficient to detect more subtle spatial patterns of big sagebrush establishment. Analysis of distance and size class at the southwestern and northeastern edges of the fires did not indicate any spatial big sagebrush establishment patterns for either fire at this finer scale, although this could have been limited by the small sample size.

We believe that wind dispersal along the southwestern prevailing winds may be affecting spatial patterns of post-fire big sagebrush, but that there are probably other factors that often override and obscure these spatial patterns. As discussed above for the smaller fires, persistence of unburned patches of big sagebrush provides important seed sources within a fire (Frischknecht and Bleak 1957, Johnson and Payne 1968, Wambolt et al. 1999). Most big sagebrush seedlings establish during infrequent recruitment pulses, and the presence of unburned patches may help big sagebrush establish more quickly after a fire or other disturbance (Johnson and Payne 1968) when establishment is so temporally limited (See *Chapter IV: Results*). The loss of unburned patches of big sagebrush combined with increasing homogeneity for cheatgrass-dominated fires reduces the ability for big sagebrush to establish after disturbances (Knick and Rotenberry 1997). Big sagebrush may be at a competitive disadvantage after fires that have left relatively few unburned patches, drastically decreasing the rate of post-fire establishment.

Wind is the primary dispersal mechanism for big sagebrush (Mueggler 1956), but secondary dispersal mechanisms, those mechanisms occurring after wind dispersal, may also influence the spatial distribution of big sagebrush establishment. Relatively little is known about secondary dispersal mechanisms, although they appear to have considerable influence on seed dispersal (Matlack 1989). In Delaware, Matlack (1989) studied sweet birch seed dispersal by wind and across snow after the initial wind dispersal. They found that secondary dispersal of seeds across snow increased the distance from the parent more than three times greater than by wind dispersal alone. Big sagebrush likely uses this combination of primary and secondary dispersal mechanisms, particularly because the seasonal timing of seed dispersal is correlated to snowfall, and that both wind and snow are dominant factors in sagebrush steppe (Young et al. 1989). Secondary seed dispersal across the snow is also influenced by the prevailing winds, so we would not expect directional patterns to differ from those created by wind dispersal alone, but big sagebrush seed could be dispersing farther distances than has been documented (Goodwin 1956, Mueggler 1956). Tertiary seed dispersal mechanisms, such as when the snow melts, may also be important in determining seedling establishment patterns at finer spatial scales (Matlack 1989). Seed arrival may have a greater effect than wind dispersal on the spatial patterns of big sagebrush establishment (Schupp and Fuentes 1995).

Heterogeneity of site conditions may also affect post-fire spatial patterns (Schupp 1995). Some sites are simply more favorable than others for successful establishment (Young et al. 1990), as can be seen within the Tractor Flats Fire. Favorable site conditions may be related to climate patterns (Grubb 1977, Eldridge et al. 1991, Schupp and Fuentes 1995) and heterogeneous patterns of fire intensity within a fire. Fire intensity influences post-fire nutrient availability and the presence of mycorrhizal fungi (Harvey et al. 1989). Mycorrhizal fungi improve big sagebrush seedling survival by increasing seedling drought tolerance (Stahl et al. 1998). Depending upon temporally dynamic climate patterns, site conditions may also be temporarily better at one site than another site (Grubb 1977, Eldridge et al. 1991, Schupp and Fuentes 1995), synergistically affecting the heterogeneity of site conditions. Availability of safe sites may be limited if they are already saturated by other species (e.g., annual and perennial grasses, forbs, resprouting shrubs) before big sagebrush has its first post-fire recruitment pulse. The length of time between the fire and the first recruitment pulse, which could be up to 10 years (See *Chapter IV: Results*), may explain why the Tractor Flats Fire, and not the B & B County Fire, exhibited spatial patterns of big sagebrush establishment related to wind dispersal. If big sagebrush doesn't get established quickly, pending the availability of a favorable climate window, other species may establish first and reduce the wind effect.

Response of Perennial Plant Cover to Densities of Big Sagebrush and Green Rabbitbrush

Total perennial grass cover (P=0.0260) and thickspike/bluestem wheatgrass cover (P=0.0015) were the only cover components that changed in response to green rabbitbrush (*Chrysothamnus viscidiflorus* (Hook.) Nutt.) density, and both decreased with increasing green rabbitbrush densities (Fig. 3.17). Sandberg bluegrass (*Poa secunda* J. Presl) was the only species for which total cover changed (P=0.0432) in response to big sagebrush density, increasing with increasing densities of big sagebrush (Fig. 3.18). Based on these results and plant associations, thickspike/bluestem wheatgrass and perennial grasses overall have an early-seral role, and Sandberg bluegrass has a late-seral role in this sagebrush steppe. In southeastern Idaho, Humphrey (1984) found that thickspike/bluestem wheatgrass and bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve) were ubiquitously abundant through all post-fire stages of vegetation development, although thickspike/bluestem wheatgrass had a slight preference for early-seral sites. He found that Sandberg bluegrass was the only perennial grass consistently associated with late-seral stands of big sagebrush.

Bluebunch wheatgrass, bottlebrush squirreltail (*Elymus elymoides* (Raf.) Swezey), Sandberg bluegrass, and rhizomatous grasses such as thickspike/bluestem wheatgrass are generally tolerant to fire. Needle-and-thread grass (*Hesperostipa comata* (Trin. & Rupr.) Barkworth) is not tolerant to fire and requires at least three to eight years for recovery to pre-fire abundance (Wright and Klemmedson 1965, Wright and Bailey 1982). If Sandberg bluegrass and thickspike/bluestem wheatgrass are both tolerant to fire, why are their associations with post-fire seral stages so divergent? Perhaps the rhizomatous growth of thickspike/bluestem wheatgrass is stimulated by disturbances and like green rabbitbrush, requires some disturbance to maintain active growth (Young and Evans 1974). The small stature of Sandberg bluegrass allows it to survive fires (Wright and Klemmedson 1965, Wright and Bailey 1982), and might also allow it to acquire resources better than other grasses as big sagebrush re-establishes on the site.

CONCLUSIONS

Big sagebrush stem diameter and age were well correlated, and our results agree with Perryman and Olson (2000) that stem diameter can be effectively used for predicting big sagebrush age. The big sagebrush size class gauge, based on the relationship between stem diameter and age, was developed for this study, and proved to be an inexpensive, non-destructive tool for quickly determining big sagebrush age in stands dominated by Wyoming big sagebrush and hybridizations between this subspecies and basin big sagebrush. The big sagebrush size class gauge could help land managers meaningfully assess the age structure of sagebrush steppe.

Comparing two relatively younger fires, the Seven-Mile Road Fire and Fire Station Fire, to two older fires, the B & B County Fire and Tractor Flats Fire, helped us understand some of our conflicting results. Directional, nonrandom patterns of seed dispersal from adjacent, unburned big sagebrush stands southwest of the fire perimeter are influencing spatial patterns of big sagebrush establishment at the Tractor Flats Fire, and possibly the Seven-Mile Road Fire. The B & B County Fire and probably the Fire Station Fire are not exhibiting spatial patterns of big sagebrush establishment associated with unburned big sagebrush stands adjacent to the fire perimeter. Spatial patterns of post-fire big sagebrush establishment are influenced by many factors, although unburned patches of big sagebrush are probably the most dominant factor consistently affecting these spatial patterns.

Another apparent trend is that big sagebrush is taking much longer than previously documented to regain its pre-fire abundance levels after fire, although their results were based on mountain big sagebrush communities (Harniss and Murray 1973). Our results indicate that 53 to 92 years are required for full post-fire recovery of big sagebrush at the

INEEL. Wyoming big sagebrush and hybridizations of Wyoming and basin big sagebrush are most prevalent at the INEEL (Shumar and Anderson 1986), so these results would be most applicable to big sagebrush stands with similar subspecies combinations.

Wright and Bailey (1982) have suggested that fire frequencies for Wyoming big sagebrush are minimally 100 years because these arid sagebrush steppe communities would otherwise be ubiquitously dominated by resprouting shrubs, such as green rabbitbrush and spineless horsebrush. The large fires that swept across the INEEL between 1994 and 2000 will likely not be substantially established by big sagebrush for about a century. While big sagebrush slowly establishes within these fires, perennial grasses and green rabbitbrush will play key roles in creating vertical structure, reducing soil erosion, and resisting a shift to cheatgrass during this vulnerable time period.

Perennial grasses overall and thickspike/bluestem wheatgrass were most abundant during early-seral stages, and Sandberg bluegrass was most abundant during late-seral stages. Although most perennial grass species individually are growing autonomously of seral stage in vegetation development, perennial grasses overall, particularly thickspike/bluestem wheatgrass, grow best during the early-seral stage when there is greater resource availability.

The results from this study should improve the knowledge base for large-scale management decisions in sagebrush steppe, and help land managers prioritize the location and type of post-fire restoration treatments, if any, within large disturbances. Furthermore, this research should advance our understanding of spatial and temporal patterns of recovery across sagebrush steppe landscapes.

CHAPTER IV

Temporal Patterns of Big Sagebrush Establishment

INTRODUCTION

Big sagebrush (*Artemisia tridentata* Nutt.) occupies vast expanses of treeless, arid lands in the western United States. Big sagebrush is a native, late-seral shrub species that often forms the dominant community structure in these areas, commonly described as sagebrush steppe. Habitat loss and degradation of big sagebrush due to agricultural conversion, excessive livestock grazing, cheatgrass invasion, and altered fire regimes is threatening the ecological integrity and biodiversity of these ecosystems (Noss et al. 1995, Knick 1999). Careful monitoring and restoration of sagebrush steppe is important to preventing further habitat loss, but big sagebrush population dynamics and establishment patterns are not well-known (Roughton 1972, Daubenmire 1975, Maier et al. 2001, Wambolt and Hoffman 2001). Recovery of big sagebrush after wildfires or other disturbances requires long periods of time, often taking many decades or a century before mature stands of big sagebrush are able to return (Wright and Bailey 1982, Bunting et al. 1987, Wambolt et al. 1999). Understanding big sagebrush seedling establishment patterns is eminently important to preserving sagebrush steppe, particularly after large-scale disturbances.

Rare events of highly favorable precipitation result in large cohorts of shrubs that have successfully established, and the effects of these rare events persist on the landscape for many decades (Williams and Hobbs 1989). Cohorts of big sagebrush that successfully establish are the consequence of recruitment pulses, periodic events associated with favorable precipitation when large numbers of organisms are recruited into a population (West et al. 1979). In Dubois, Idaho, West et al. (1979) did not find statistically significant evidence indicating the occurrence of recruitment pulses in species associated with sagebrush-grass communities, although they did not specifically study big sagebrush. Observations of several distinct cohort years of widespread big sagebrush establishment throughout Wyoming indicate that recruitment pulses are important to big sagebrush seedling establishment (Maier et al. 2001). Being able to predict the probability of recruitment pulses, resulting in large cohorts of successfully established big sagebrush, could be used by managers to improve the effectiveness of big sagebrush seeding projects and maximize the probability of naturally occurring big sagebrush establishment.

The goal of this research was to assess the relationship between big sagebrush establishment and total seasonal precipitation, total annual precipitation for the current year, total annual precipitation for one, two, three, four, and five years prior to establishment, and combinations of two consecutive years of total annual precipitation. The results from this study are valuable to the management and restoration of sagebrush steppe and the native plant and animal species dependent on these lands.

METHODS

Plots were located at the INEEL within and immediately outside the perimeters of four fires that burned at least 17 years ago (see *Chapter III: Methods*). The fires selected for the study were chosen for similarity in soils, topography, and plant composition within each fire and between fires, and represent different time periods and fire sizes (Fig. 3.1). All sampling for this study occurred in June, July, and August of 2002 and these plots are labeled "2002 plots" in Figs. 1.1-1.2 and Figs. 3.1-3.5. Each plot consisted of two 50-m transects spaced 10 m apart and followed the southwest to northeast direction.

We developed a gauge to estimate big sagebrush age using the maximum stem diameter at the base of the shrub (Fig 3.6). Several studies indicate that big sagebrush age and stem diameter are correlated (Winward 1970, Perryman and Olson 2000). We used age prediction equations for basin big sagebrush from southwestern Wyoming (Perryman and Olson 2000) to determine the stem diameter sizes that correspond with chosen age classes. Using the age classes of 0 to 10, 10 to 20, 20 to 30, and greater than 30 years, the stem diameter size classes were 0 to 13, 13 to 44, 44 to 75, and greater than 75 mm, respectively. The true age classes associated with these size classes for big sagebrush within the study area are 0 to 13.5, 13.5 to 29.1, 29.1 to 42.9, and greater than 42.9 years (see *Chapter III: Results and Discussion*). The gauge was inexpensively constructed from a Plexiglas sheet and allowed quick determination of stem diameter size class to estimate the age class of big sagebrush.

Three big sagebrush cross-section sampling points were established at the 0-, 25-, and 50-m points along each transect, for a total of six per plot. The closest big sagebrush of each of the four size classes to the point was sampled, and the maximum distance from the point was 5 m. Sampling representatives of each big sagebrush size class included measuring the height and stem diameter of the big sagebrush, and subsequent removal of the cross-section.

Cross-section sampling and analysis follow methods previously used for big sagebrush (Ferguson 1964, Perryman and Olson 2000, Maier et al. 2001, Perryman et al. 2001, Wambolt and Hoffman 2001). The cross-sections were cut with a saw or clippers at ground level so that the pith and first annual growth ring would be included. The second cut was made 10 cm above the first cut, so that each cross-section was approximately 10 cm long. Sampling of deteriorated big sagebrush (i.e., damaged stem, dead) was avoided because their age is less likely to be accurately assessed. The bottom portion of each big sagebrush cross-section was sequentially sanded with 100, 150, and 320 grit sanding belts. Annual growth rings were counted through a dissecting scope and cross-dating was used to maintain accuracy in counting the growth rings. Applying a light coat of mineral oil to each cross-section improved the contrast between the annual growth rings.

Precipitation records for the INEEL are available between 1953 and 2002, and precipitation records between 1906 and 1952 were extrapolated from three nearby weather stations (Arco, Blackfoot, Idaho Falls) using methods described by Anderson and Inouye (2001). Total annual and seasonal precipitation was based on water-year precipitation (October-September; NCDC 2003 as accessed by INSIDE Idaho 2002). Big sagebrush cross-sections with complete age records (e.g., cross-section was intact and allowed for ascertaining establishment date) were used to build a histogram showing the number of big sagebrush that established each year and the total annual precipitation.

We used SAS for Windows 8.02 to develop regression models and analyze the relationship between big sagebrush establishment and precipitation. The regression models had one dependent variable, the natural log of the total number of big sagebrush that established each year plus one (LOG [N+1]). The regression model had two to three dependent variables, the age of the big sagebrush that established each year (AGE), and one or two precipitation variables. The AGE variable was used to adjust the model for increased senescence of big sagebrush with greater time since establishment. The precipitation variables included: total current year seasonal precipitation (October-December precipitation [OCT-DEC], January-March precipitation [JAN-MAR], April-June precipitation [APR-JUN], and July-September precipitation [JUL-SEP]), total annual precipitation (current year precipitation [CURRYR], and precipitation for one year [ONEYR], two years [TWOYR], three years [THREEYR], four years [FOURYR], and five years [FIVEYR] prior to big sagebrush establishment. Combinations of two precipitation variables were also used to analyze the relationship between big sagebrush establishment and consecutive years of total annual precipitation (current year precipitation and precipitation one year prior to establishment, precipitation one and two years prior to establishment, and precipitation two and three years prior to establishment). All seasonal and annual precipitation variables were based on the October-September water-year precipitation.

The regression models were weighted by the relative density of the big sagebrush size classes at all plots to reflect changes in big sagebrush density over time. The relative density of the 0-13, 13-29, 29-43 and 43+ year size classes was 0.74, 0.19, 0.05, and 0.02, respectively. The 0.74 weight was used to weight big sagebrush that were between 0-13 years; the 0.19 weight was used to weight big sagebrush that were between 14-29 years; the 0.05 weight was used to weight big sagebrush that were between 29-43 years; and the 0.02 weight was used to weight big sagebrush that were greater than 43 years old.

RESULTS AND DISCUSSION

Big sagebrush establishment occurred in a cyclical, multimodal pattern, peaking at approximately 9-13 year intervals (Figs. 4.1-4.2). We assumed that big sagebrush successfully establishing in a certain year had an equal probability of survivorship compared to big sagebrush that successfully established in adjacent years. Using this assumption, there are at least six distinct cycles apparent in Figs. 4.1-4.2. The cycles appear to be right-skewed and occur in a relatively predictable pattern. The observation of a cyclical, multimodal distribution in big sagebrush establishment is not unprecedented (Roughton 1972, Cawker 1980, Wambolt and Hoffman 2001). In Colorado, Roughton (1972) noted a similar pattern between age classes at four-year intervals for big sagebrush, antelope bitterbrush (*Purshia tridentata* (Pursh) DC.), and mountain-mahogany (*Cercocarpus montanus* Raf.) and described this pattern as "appearing almost cyclic." Cawker (1980) observed a multimodal distribution of big sagebrush age (subspecies not identified) over an 80-year time period in southern British Columbia, although it was more irregularly distributed than in the current study. This irregular big sagebrush age distribution may be attributed to their inclusion of all big sagebrush individuals sampled, including those with missing or rotten piths. Cawker (1980) found that big sagebrush establishment was well-correlated with multiple climate factors, particularly moisture and temperature; however, sampling methodology limitations precluded analysis of lag effects. Wambolt and Hoffman (2001) observed a bimodal pattern in the age distribution of mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana* (Rydb.) Beetle) in southwestern Montana, although the temporal scale was limited to only 15 years (1978-1992). Wambolt and Hoffman (2001) attributed the bimodal peaks in mountain big sagebrush establishment to major reductions in elk populations, although statistical analysis was not performed to test this assumption.

The only current year seasonal precipitation variable positively correlated with big sagebrush establishment was October-December precipitation immediately prior to establishment (Partial P-value=0.003; Fig. 4.3). The only total annual precipitation variable positively correlated with big sagebrush establishment was precipitation one year prior to establishment (Partial P-value=0.023; Fig. 4.4). Total annual precipitation for five years prior to establishment was negatively correlated with big sagebrush establishment (Partial P-value=0.031; Fig. 4.4), likely due to the cyclical nature of big sagebrush establishment. Total current year precipitation was negatively correlated with big sagebrush establishment (Partial P-value=0.040) when analyzed together with total annual precipitation for one year prior to establishment (Fig. 4.5). Total annual precipitation for two years prior to establishment was positively correlated with big sagebrush establishment (Partial P-value=0.044) when analyzed together with total annual precipitation for one year prior to establishment (Fig. 4.5). Total annual precipitation for two and three years prior to establishment was not correlated with big sagebrush establishment (Fig. 4.5).

Based on the current analysis, precipitation occurring during the October-December immediately prior to establishment, precipitation one year prior to establishment, and combinations of total annual precipitation for one year prior to establishment and adjacent years positively influences big sagebrush establishment. Several studies have found that high springtime precipitation positively influences widespread big sagebrush seedling germination (Young and Evans 1989, Young et al. 1990) and establishment (Maier et al. 2001). Young and Evans (1989) observed high densities of big sagebrush seedlings after a high precipitation event, although none survived into the summer months. Maier et al. (2001) found that five Wyoming big sagebrush cohort years were associated with higher precipitation in December and January than were five years with no establishment, although they did not compare big sagebrush establishment to prior years' precipitation. Several studies have recorded big sagebrush establishment despite low or average current year precipitation (Johnson and Payne 1968, Wagstaff and Welch 1990). In northern Utah, Wagstaff and Welch (1990) observed extraordinarily high rates of mountain big sagebrush seedling survival in 1988, one of the driest years on record. The year 1988 also corresponded with widespread big sagebrush establishment in southwestern Montana (Wambolt and Hoffman 2001) and in the current study. Our results concur with Maier et al. (2001) that recruitment pulses occur in big sagebrush populations, although our results indicate that the recruitment pulses are associated with high precipitation during the year prior to establishment and up to the October-December immediately prior to establishment, and that precipitation during the year of establishment is not as important.

The lag effect in big sagebrush establishment is distinctive in Figs. 4.1-4.5, but it is not unprecedented. Anderson and Inouye (2001) observed that big sagebrush cover at the INEEL had a three-year lag effect with water-year precipitation (October-September). In the same study, the cover of perennial grass species and other shrub species also exhibited two- to four-year lag effects. They theorized these lag effects may be caused by the amount of time required for seedlings to grow large enough so they have a greater probability of being measured. Our results suggest that the lag effects observed in the current study are not related to sampling methodology; lag effects appear to be a pattern inherent to big sagebrush establishment. The influence of lag effects after specific weather sequences on vegetation dynamics and widespread seedling establishment has been observed in arid lands worldwide (Crisp 1978, Griffin and Friedel 1985, Goldberg and Turner 1986, Austin and Williams 1988, Friedel et al. 1993). Wiegand et al. (1995) discussed the importance of incorporating lag effects into simulation models for shrublands in South Africa, and suggested that drought periods increase the proportion of safe sites for seedlings, enabling successful seedling establishment once favorable precipitation conditions return. Lag effects may also be related to increased big sagebrush seed production during high precipitation years (Young et al. 1989), so that the seed bank is greater in the subsequent year.

Precipitation is a driving force in big sagebrush seedling establishment, although there may be additional factors that affect big sagebrush establishment (Cawker 1980). In Australia, high precipitation related to the El Niño/Southern Oscillation phenomenon has been associated with shrub recruitment pulses (Austin and Williams 1988). Similarly, the La Niña phenomenon is the dominant, cyclical weather pattern that produces high precipitation years in the Pacific Northwest. La Niña events were recorded in the following years: 1920, 1924, 1928, 1931, 1938, 1942, 1949, 1954, 1964, 1970, 1973, 1975, 1988, 1998, and 1999 (Kiladis and Diaz 1989, Smith and Sardeshmukh 2000, NOAA 2003). Most of these La Niña events appear to be positively associated with big sagebrush establishment, particularly 1924, 1938, 1942, 1949, 1975, and 1988, although there are exceptions (Fig. 4.1).

Black-tailed jackrabbit (*Lepus californicus* Gray) populations are also cyclical and peak at approximately 10-year intervals (Anderson and Shumar 1986). High population densities of black-tailed jackrabbits exhibit either a population crash or a gradual reduction in density in successive years (Johnson and Peek 1984). Black-tailed jackrabbits preferentially forage on perennial grasses, but forage on shrubs, including big sagebrush, in the fall and winter, and substantially impact all vegetation (Currie and Goodwin 1966, Anderson and Shumar 1986). Peak densities of black-tailed jackrabbits for the INEEL and eastern Idaho were recorded in the following years: 1896-1897, 1913-1914, 1920-1921, 1935-1936, 1948-1949, 1959, 1971, 1981-1982 (Johnson and Peek 1984). There may have been additional peak years of black-tailed jackrabbit densities prior to 1950 that were not recorded. Black-tailed jackrabbit surveys at the INEEL since 1980 indicate that there were also peaks in 1992 and 2000, but they are barely a fraction of the high densities occurring in 1981-1982 (Sue Vilford, unpublished data). Peak densities of black-tailed jackrabbits corresponding with big sagebrush establishment are indicated in Fig. 4.2, and they appear to be associated with the onset of each big sagebrush establishment cycle, when big sagebrush establishment is low, although this is difficult to ascertain for earlier years. Herbivory of big sagebrush seedlings may suppress successful big sagebrush establishment in years with peak densities of black-tailed jackrabbits. However, the relatively small peaks of black-tailed jackrabbit densities in 1992 and 2000 indicate that precipitation remains the main factor determining big sagebrush establishment. Both precipitation and black-tailed jackrabbit cycles may be synergistically influencing the long-term survival of big sagebrush seedlings.

Our results indicate that, in general, big sagebrush establishment occurs after, and not during wet periods. Wet periods create favorable conditions for annual and biennial plant species (see *Chapter II: Results and Discussion*), increasing competition between annual and biennial plant species and big sagebrush seedlings. In California, Williams and Hobbs (1989) found that seedlings of a shrubs species, chaparral broom (*Baccharis pilularis* DC.), had much slower root elongation rates than annual grasses, causing reduced shrub seedling establishment. In the current study, big sagebrush seedling establishment may be most successful after a wet period because they are able to avoid direct competition with annual and biennial plant species for resources, with which big sagebrush seedlings are poor competitors (Goodwin 1956, Daubenmire 1975), but still have reliable soil moisture through the summer. Wet periods reduce the depth of permanently moist soil, increasing the probability that the roots of shrub seedlings will reach permanently moist soil before the summer drought (Westoby 1979, Williams and Hobbs 1989). In a greenhouse study, Welch and Jacobsen (1988) found that Wyoming big sagebrush seedlings had roots long enough to access the entire soil profile after only 45 to 50 days following germination, in time for summer drought conditions. Hydraulic lift from adult big sagebrush also increases soil moisture availability higher in the soil profile (Richards and Caldwell 1987), thus facilitating seedling establishment of piñyon pine (*Pinus monophylla* Torr. & Frém.; Callaway et al. 1996). Hydraulic lift may also facilitate big sagebrush seedling establishment, although big sagebrush seedling growth and survival is often limited with increasing proximity to adult big sagebrush (Reichenberger and Pyke 1990). Soil moisture, perhaps more than precipitation, is important to big sagebrush seedling establishment (Cawker 1980, Reichenberger and Pyke 1990) and water relations (Campbell and Harris 1977, Sturges 1977, Richards and Caldwell 1987). Increased soil moisture persisting following a wet period may be a key factor determining the long-term survival of big sagebrush seedlings.

CONCLUSIONS

We conclude that big sagebrush establishment occurs cyclically at approximately 10 year intervals and is largely influenced by high precipitation during the October-December and one year prior to germination. The La Niña phenomena, also associated with high precipitation years, may be influencing big sagebrush establishment. Population cycles of black-tailed jackrabbits also correspond with years with low big sagebrush establishment, although it is unknown whether black-tailed jackrabbit population dynamics and big sagebrush establishment are directly correlated with each other. The best explanation for the lag effect of big sagebrush establishment may be related to the increased availability of soil moisture following high precipitation years. The cyclical pattern of big sagebrush establishment appears to be relatively predictable and could be used by managers to optimally time big sagebrush seeding projects or adjust land use practices (e.g., livestock grazing) to maximize and protect big sagebrush establishment (Crisp 1978, Owens and Norton 1992). Likewise, big sagebrush seeding projects after a drought year may be futile and economically unwise. Conditions favorable for widespread big sagebrush seedling success are infrequent (~1 in 10 years) and managers should monitor both climate and vegetation data so that they can recognize the likelihood of a widespread big sagebrush recruitment event (Wiegand et al. 1995). Future studies of big sagebrush establishment using shorter time scales should be cautious about making broad assumptions concerning the effectiveness of treatments (Wagstaff and Welch 1990) or one-time events (Wambolt and Hoffman 2001) on big sagebrush seedling success when results may be better attributed to factors driving widespread recruitment pulses. Additional studies are needed to evaluate the causes of lag effects between big sagebrush establishment and precipitation.

CHAPTER V

MANAGEMENT IMPLICATIONS

Our results indicate that climate plays a dominant role in the vegetation dynamics of sagebrush steppe. In arid environments, water resources are highly limited and plant communities accordingly respond to sequences of favorable or unfavorable precipitation (Westoby 1979). At the INEEL, favorable precipitation during the mid 1990's was associated with the increased abundance and richness of perennial grasses, perennial forbs, native annuals and biennials, and introduced annuals and biennials, and native forb diversity during the 1995 survey. Sequences of favorable precipitation have also instigated widespread big sagebrush establishment at a one-year time lag. Understanding the relationships between climate and vegetation can improve the predictability of future vegetation dynamics. Annuals and biennials respond most readily to favorable precipitation, while other growth forms may require one or more years before the precipitation effects become discernable (Anderson and Inouye 2001). Climate change will likely influence vegetation dynamics in both the absence and presence of fire, particularly in plant communities that are especially sensitive to climate patterns (Delcourt et al. 1983, Romme and Turner 1990, Sprugel 1991, Bartlein et al. 1997).

Declines in big sagebrush cover were initially observed during the 1975 survey, and have been attributed to the widespread big sagebrush mortality that occurred throughout the Intermountain West (McArthur et al. 1990, Anderson and Inouye 2001). Our results also found that big sagebrush cover has not only declined on plots recently burned in fire, but that big sagebrush cover has been substantially declining on unburned plots. Widespread big sagebrush mortality, increased herbaceous biomass associated with high precipitation in the mid-1990's, and the pairing of dry years after wet years likely enhanced the continuity of fuels, resulting in several large wildfires at the INEEL (Swetnam and Betancourt 1990, Knapp 1995, Miller and Rose 1999). The persistence of standing dead big sagebrush may contribute to additional fires when there is an appropriate combination of future climatic events.

Big sagebrush establishment has proceeded very slowly at the older fires that we studied, regardless of fire size, and our results indicate that recovery approximating pre-fire conditions may take up to 92 years. Such slow recovery rates are typical of sagebrush steppe communities dominated by Wyoming big sagebrush (Wright and Bailey 1982), and management plans should account for post-disturbance structural losses that will persist on the landscape for many decades (Wambolt et al. 2001). Unburned patches of big sagebrush within the Seven-Mile Road Fire appeared to contribute to improved big sagebrush establishment. Wind dispersal from adjacent, unburned sagebrush steppe may or may not be a dominant pattern affecting post-fire big sagebrush establishment. Post-fire spatial patterns of big sagebrush establishment may depend on the length of time before the next recruitment pulse. Other species may become well-established in the intervening period before the next recruitment pulse, up to 10 years after the fire, reducing site availability for big sagebrush seedlings.

The large areas that have burned at the INEEL within the past decade may take nearly a century until they approximate pre-fire big sagebrush age structure. Many areas within these recent fires are at least five kilometers to the nearest fire perimeter, and the short dispersal range of big sagebrush seed from sagebrush steppe adjacent to the fire will not expediently reseed the fires. The presence of unburned patches of big sagebrush within the fires will be critical to seed distribution throughout areas that would otherwise not be seeded for decades. Management strategies that increase the patchiness of fires should positively influence post-fire big sagebrush establishment.

Plots burned in the recent fires have exhibited a reduction of green rabbitbrush cover in the years immediately after burning, followed by increases of both green rabbitbrush cover and density in subsequent years. Fires stimulate green rabbitbrush cover and density, although several years may be required until this response is apparent (Young and Evans 1974). After the initial post-fire recovery, green rabbitbrush continues growth and remains a community dominant until big sagebrush becomes established or another fire stimulates the green rabbitbrush (Harniss and Murray 1973, Young and Evans 1974). Other researchers have questioned an increasing trend in green rabbitbrush cover across the INEEL since 1950 prior to the most of the recent fires (Anderson and Inouye 1988, Anderson and Inouye 2001). Our results suggest that overall green rabbitbrush cover has remained relatively constant throughout the study area since 1985. Further analysis revealed that green rabbitbrush cover has been declining since the 1995 survey at unburned plots, and has increased at most plots that were burned within the past decade. Green rabbitbrush is well-adapted for infrequent disturbances, but can persist long after the initial disturbance and eventually coexist with big sagebrush as a subdominant species in later seral stages (Young and Evans 1974, Humphrey 1984). Nearly a century after the Tractor Flats Fire, green rabbitbrush still occurs at densities similar to those of big sagebrush. The longer post-fire big sagebrush establishment rates in the most arid regions of its distribution likely increases the temporal scale that green rabbitbrush and other early seral species retain their post-fire dominance.

Early seral species may be especially important in sagebrush steppe dominated by Wyoming big sagebrush, where transitions to later seral stages may be upwards of a century.

During this time period, green rabbitbrush or other vegetatively resprouting shrubs, and perennial grasses provide the primary vertical structure across expansive landscapes, replacing the functional role of big sagebrush until big sagebrush is able to establish itself. These early seral species not only provide vertical structure, but also reduce soil erosion and resist invasions from introduced annuals and biennials (Whisenant 1999). Introduced annuals and biennials, particularly cheatgrass, aggressively respond to disturbances such as fire and reduce ecological diversity (Stewart and Hull 1949, Whisenant 1990, Peters and Bunting 1994). Rapid establishment of native, early seral species may preclude transition to cheatgrass-dominated grasslands (Whisenant 1999). Understanding the intermediary role of early seral species during the long time period between fire and big sagebrush establishment is important to managing recently burned sagebrush steppe, and could be used to restore sagebrush steppe that has been degraded.

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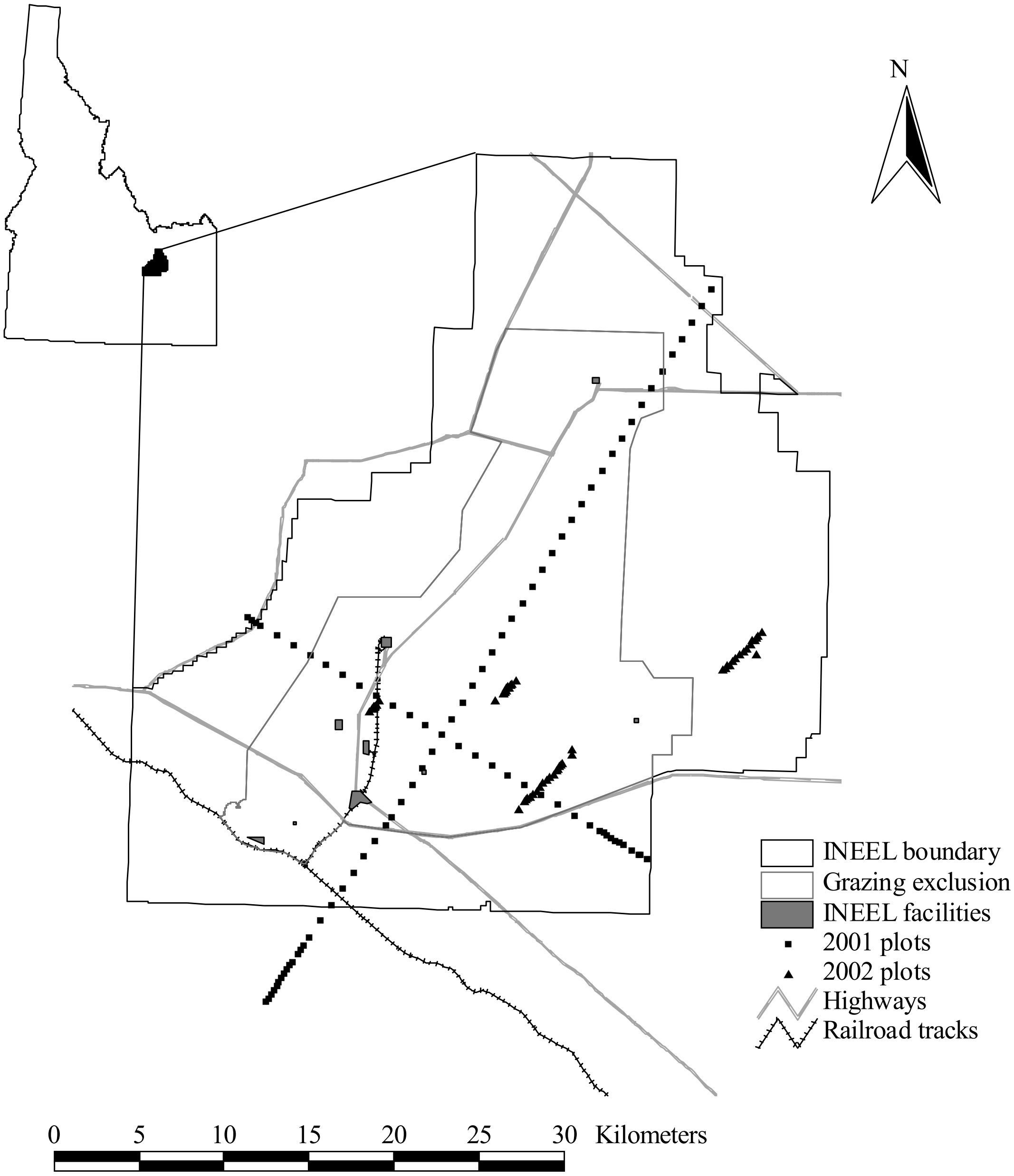

Figure 1.1. Map of the Idaho National Engineering and Environmental Laboratory (INEEL) showing the locations of plots sampled in 2001 and 2002. The boundary within the INEEL boundary is closed to livestock grazing. The inset in the upper-left corner shows the placement of the INEEL in Idaho (USA).

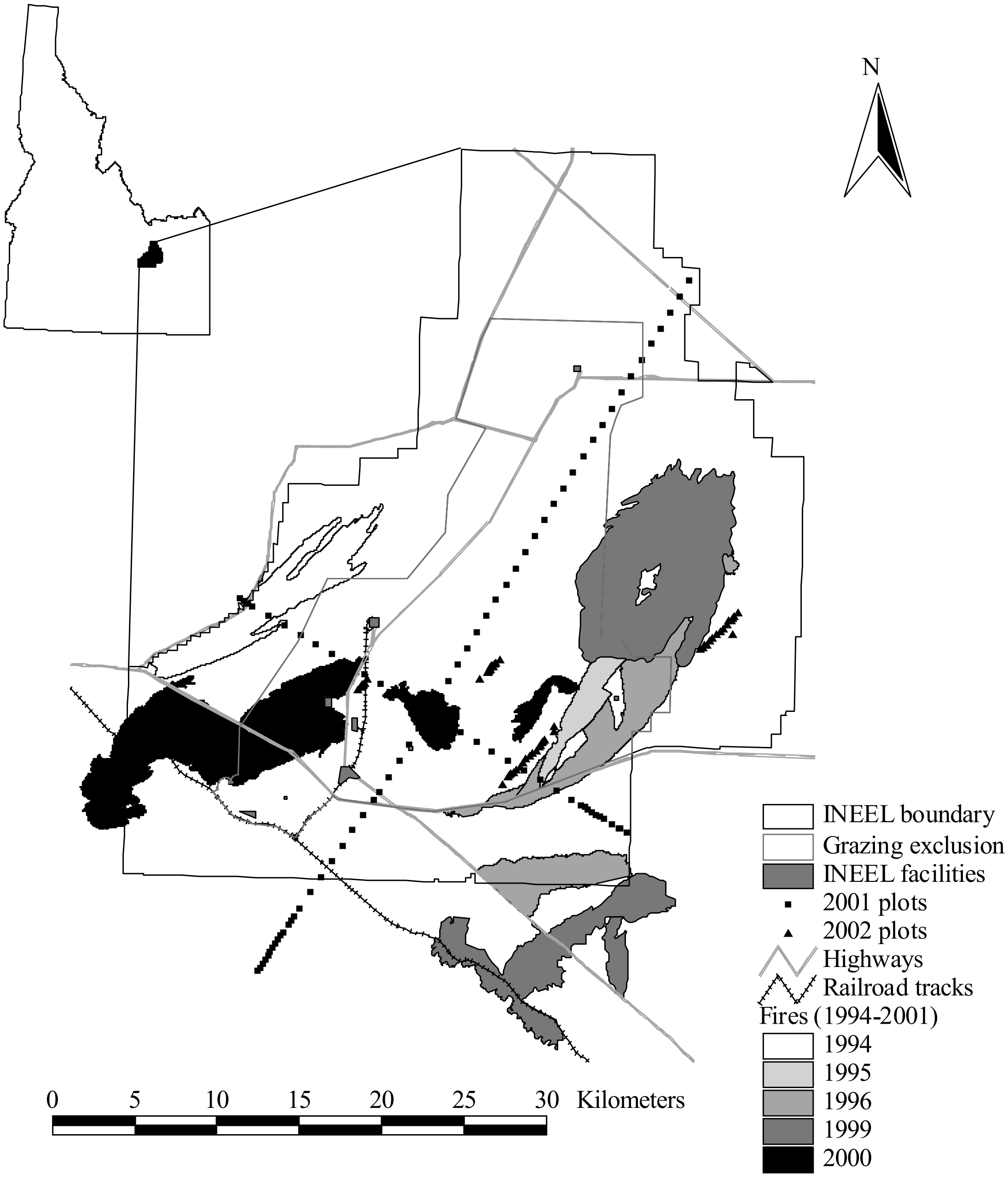

Figure 1.2. Map of the Idaho National Engineering and Environmental Laboratory (INEEL) showing the locations of recent fires that have occurred since 1994. The Cox's Well Fire (1996) was a fire that burned 2001 plots south of Big Southern Butte and is not shown in this map.

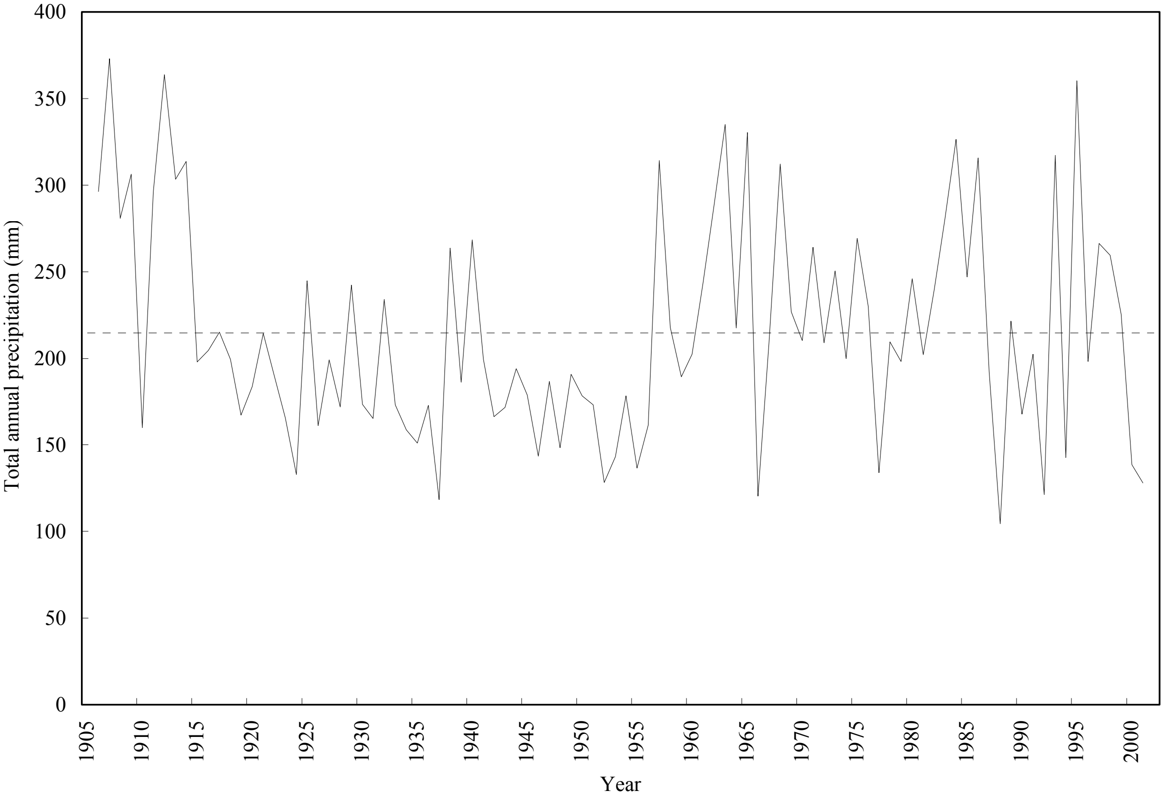

Figure 1.3. Figure showing total annual precipitation for the Idaho National Engineering and Environmental Laboratory (INEEL). The solid line represents total annual precipitation for the water-year (October-September). The dashed line represents total mean precipitation based on the total annual precipitation data. Precipitation data before 1950 were estimated by correlating precipitation data between the INEEL and 1-3 nearby locations.

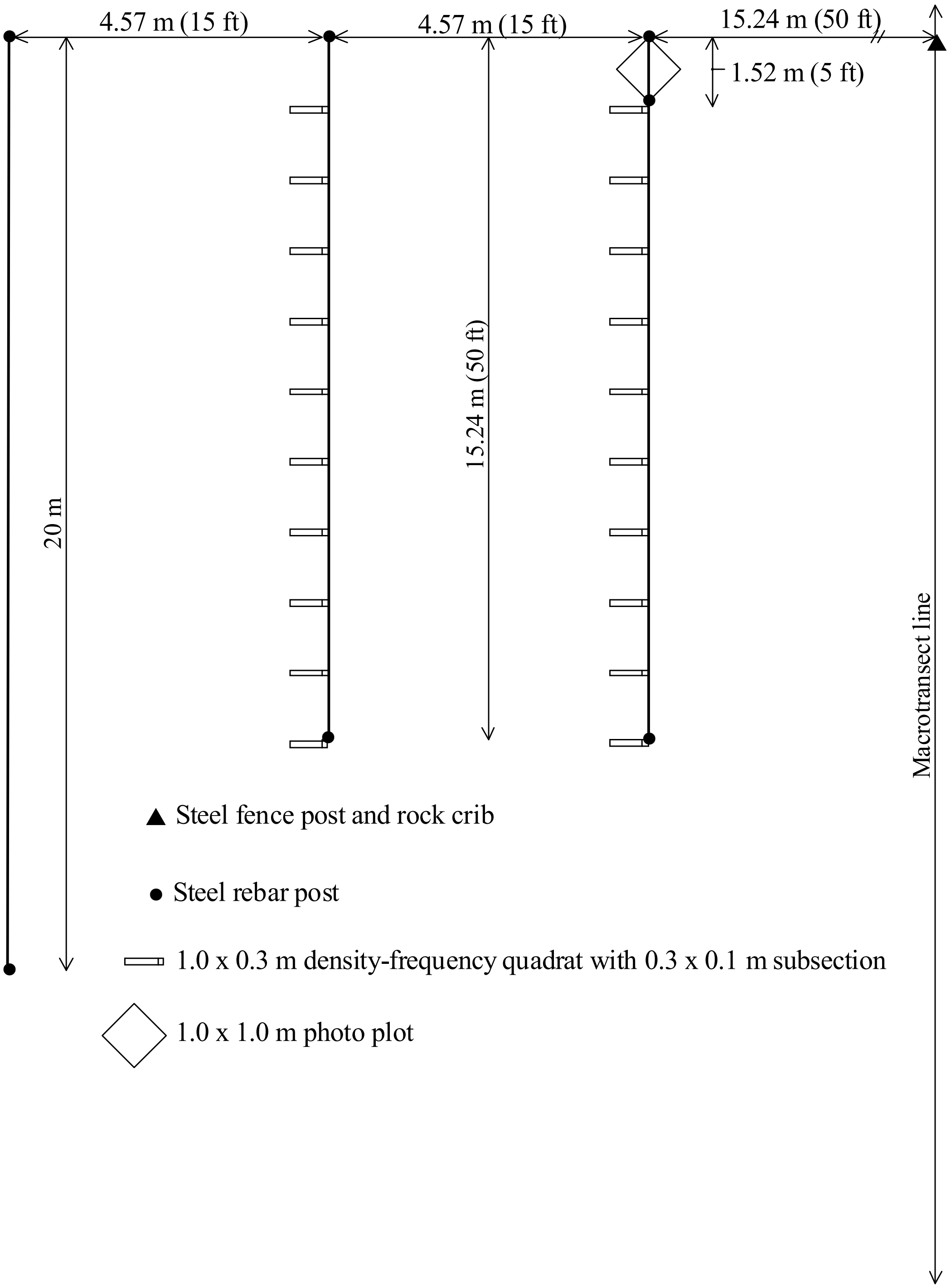

Figure 2.1. Long-term vegetation plot layout showing the two 15.24-m (50-ft) long transects and 20-m long transect that are parallel to the macrotransect line. This figure is adapted from Anderson et al. (1978).

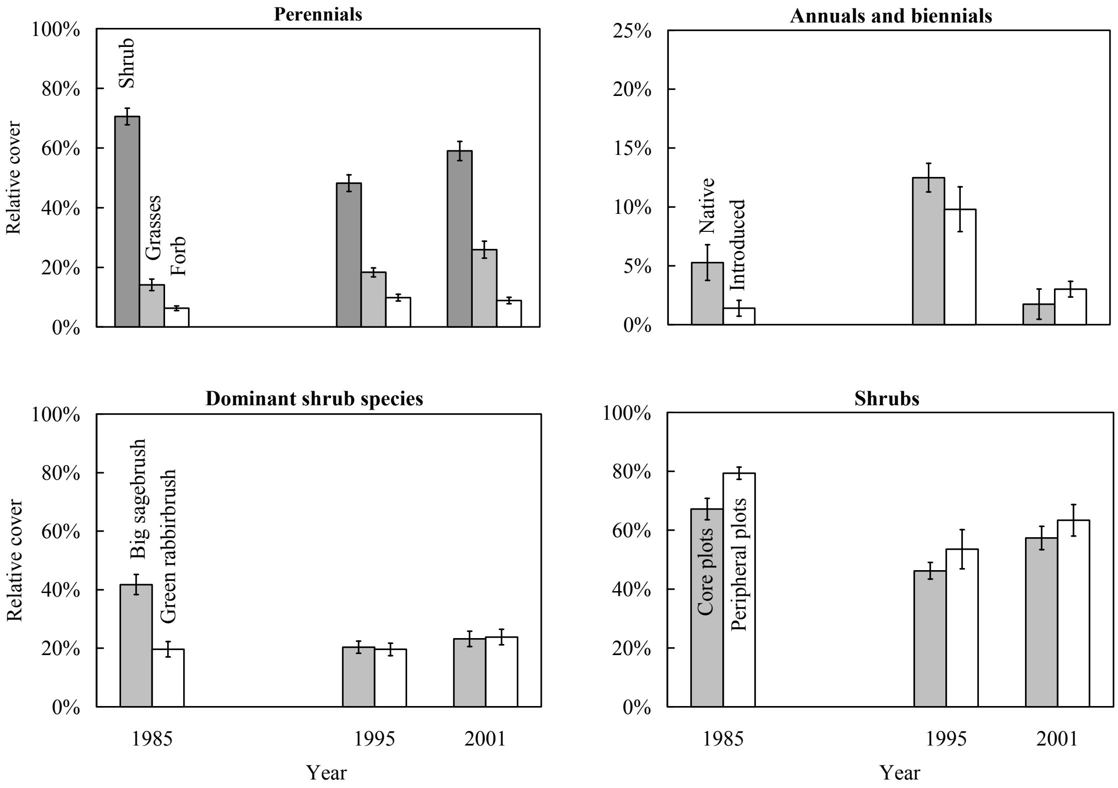

Figure 2.2. Relative cover of perennials, annuals and biennials, and dominant shrub species at all plots (N=58), and relative cover of shrubs between the core plots (N=42) and peripheral plots (N=16). Data are mean values of point-interception data and error bars are ±1 standard error.

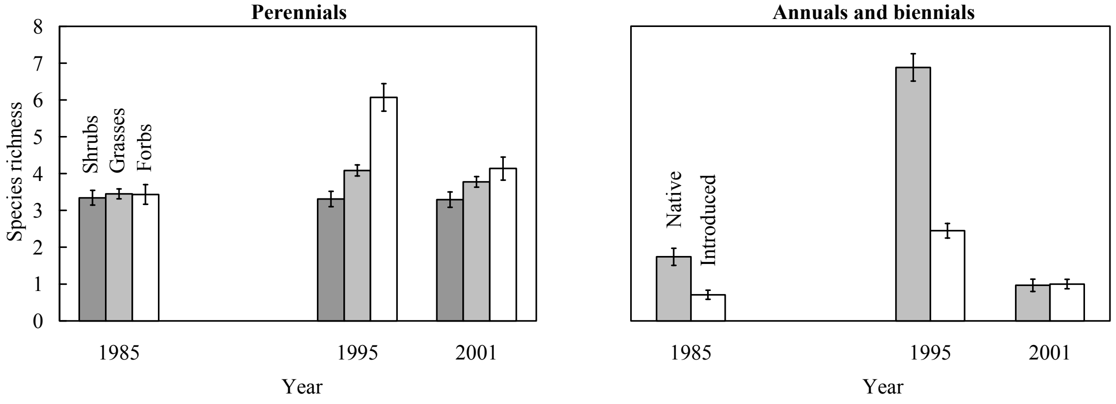

Figure 2.3. Species richness of perennials and annuals and biennials at all plots (N=58). Data are mean values of point-interception data and error bars are ±1 standard error.

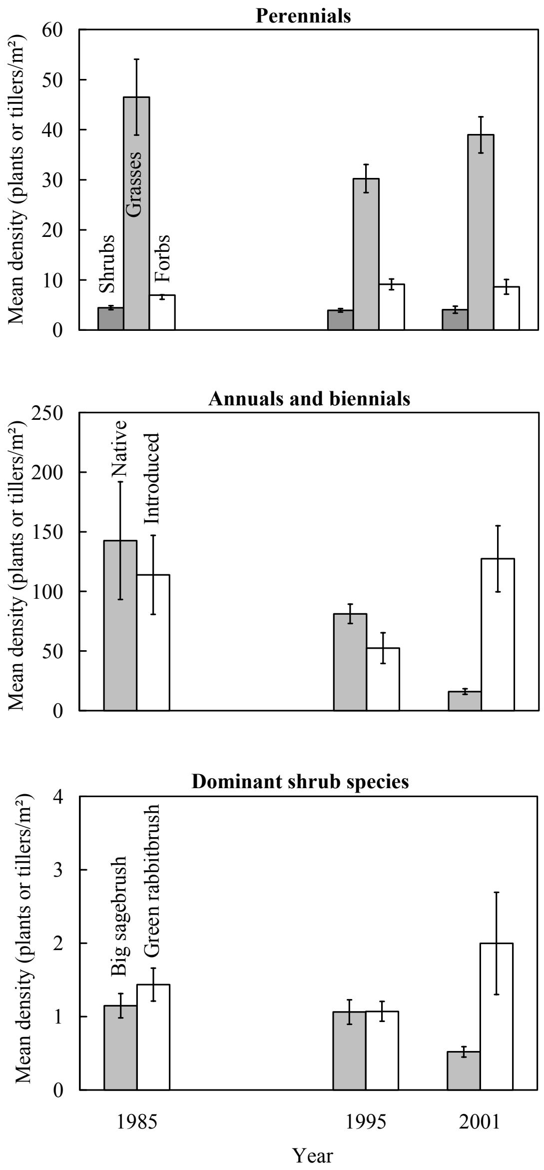

Figure 2.4. Mean density of perennials, annuals and biennials, and dominant shrub species at all plots (N=87). Data are mean values of density data and error bars are ±1 standard error.

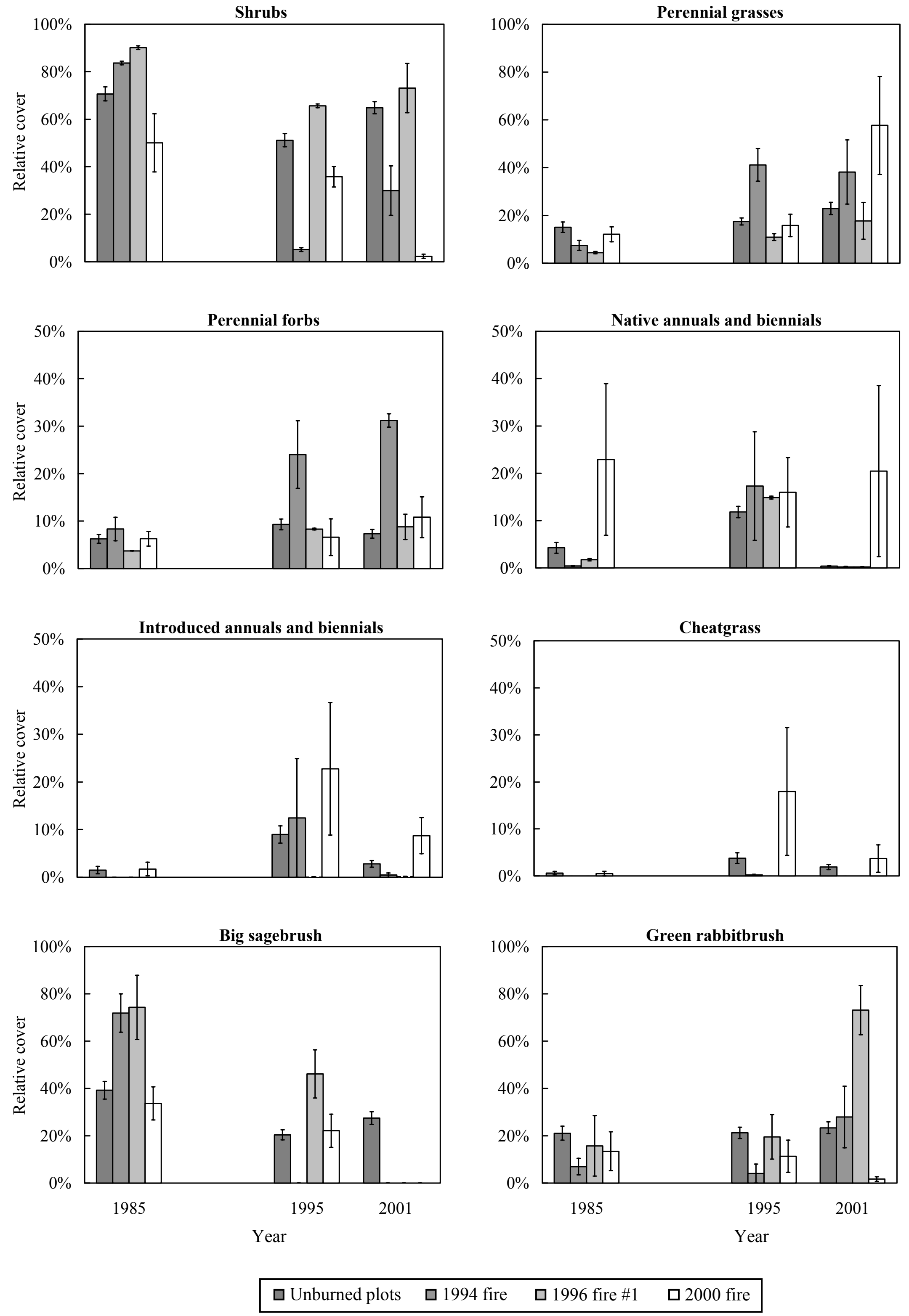

Figure 2.5. Relative cover of species groups and common perennial species at unburned plots (N=49), and plots burned in the 1994 fire (N=3), 1996 fire #1 (N=2) and 2000 fire (N=4). Data are mean values of point-interception data and error bars are ±1 standard error.

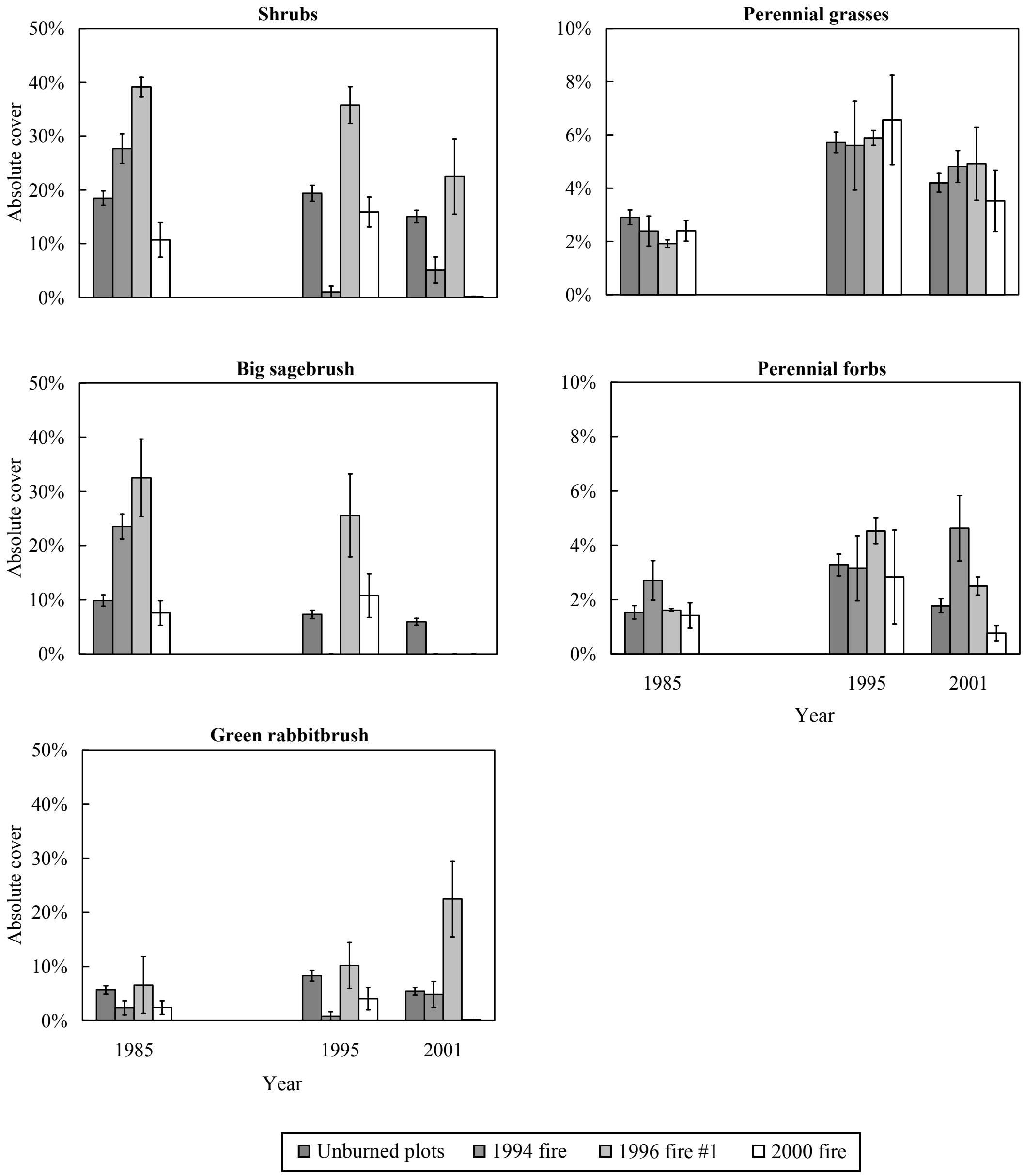

Figure 2.6. Absolute cover of species groups and common perennial species at unburned plots (N=49), and plots burned in the 1994 fire (N=3), 1996 fire #1 (N=2) and 2000 fire (N=4). Data are mean values of point-interception data expressed in percentages and error bars are ±1 standard error.

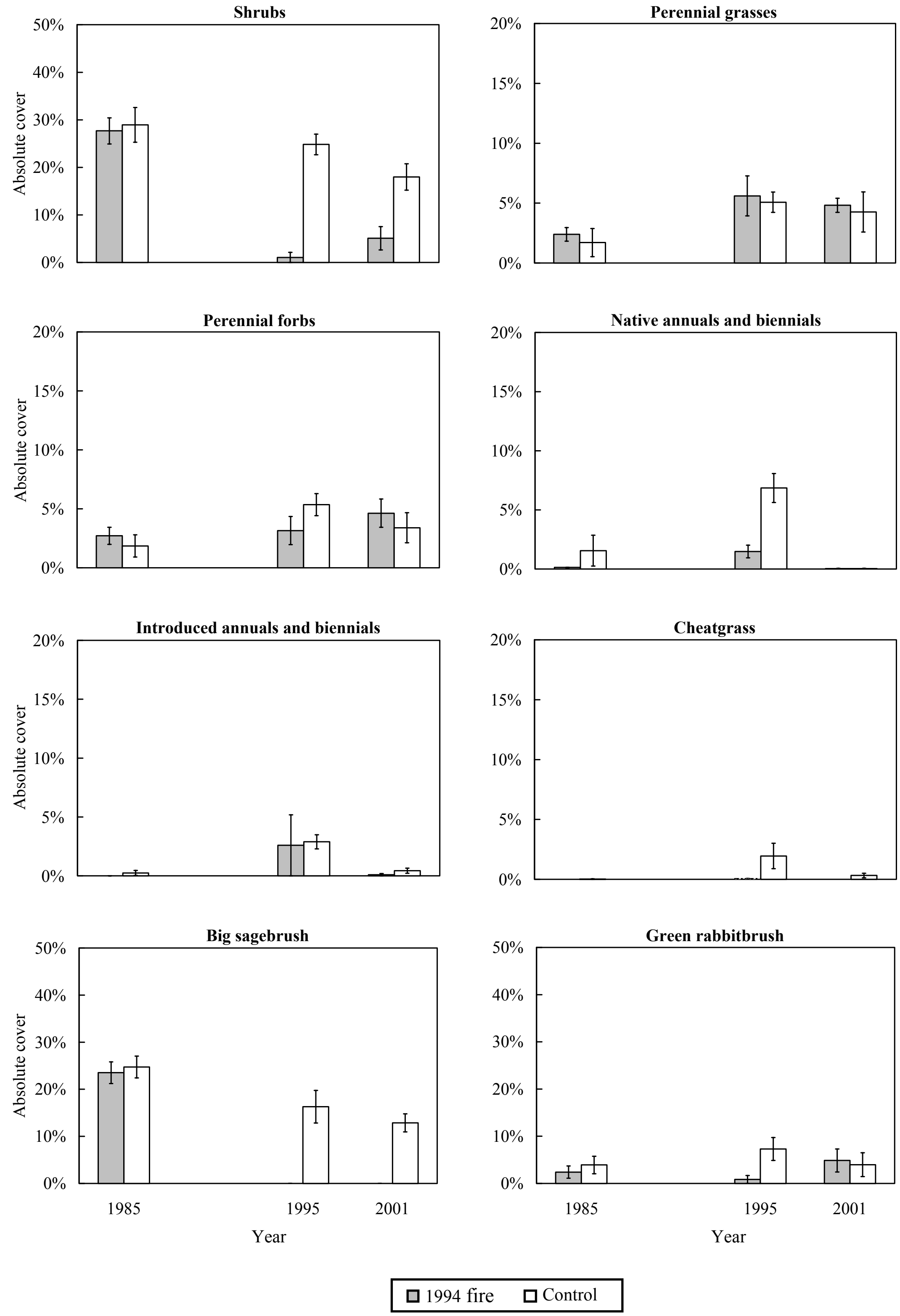

Figure 2.7. Absolute cover of species groups and common perennial species at plots burned in the 1994 fire (N=3) and control plots (N=3). Data are mean values of point-interception data expressed in percentages and error bars are ±1 standard error.

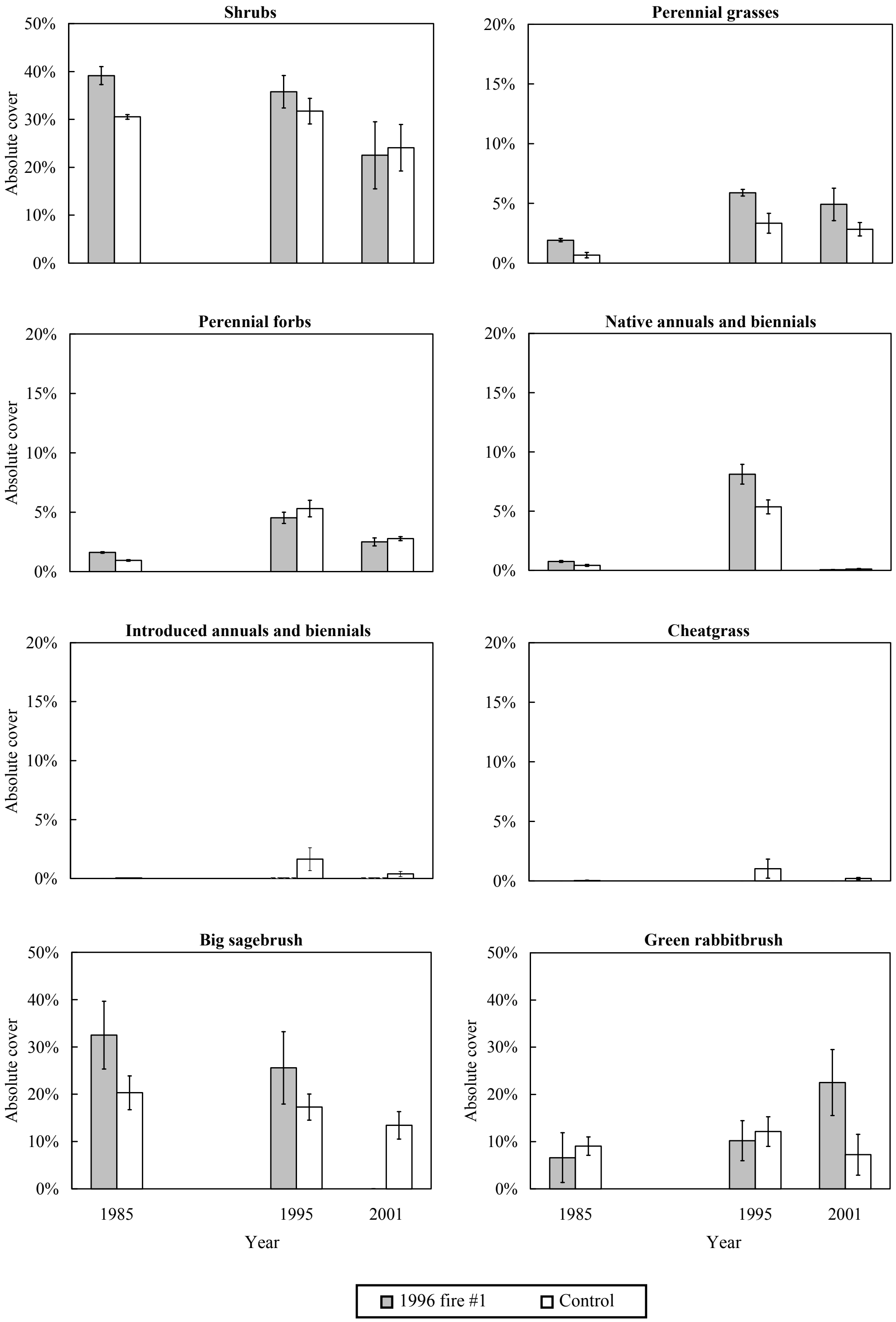

Figure 2.8. Absolute cover of species groups and common perennial species at plots burned in the 1996 fire #1 (N=2) and control plots (N=2). Data are mean values of point-interception data expressed in percentages and error bars are ±1 standard error.

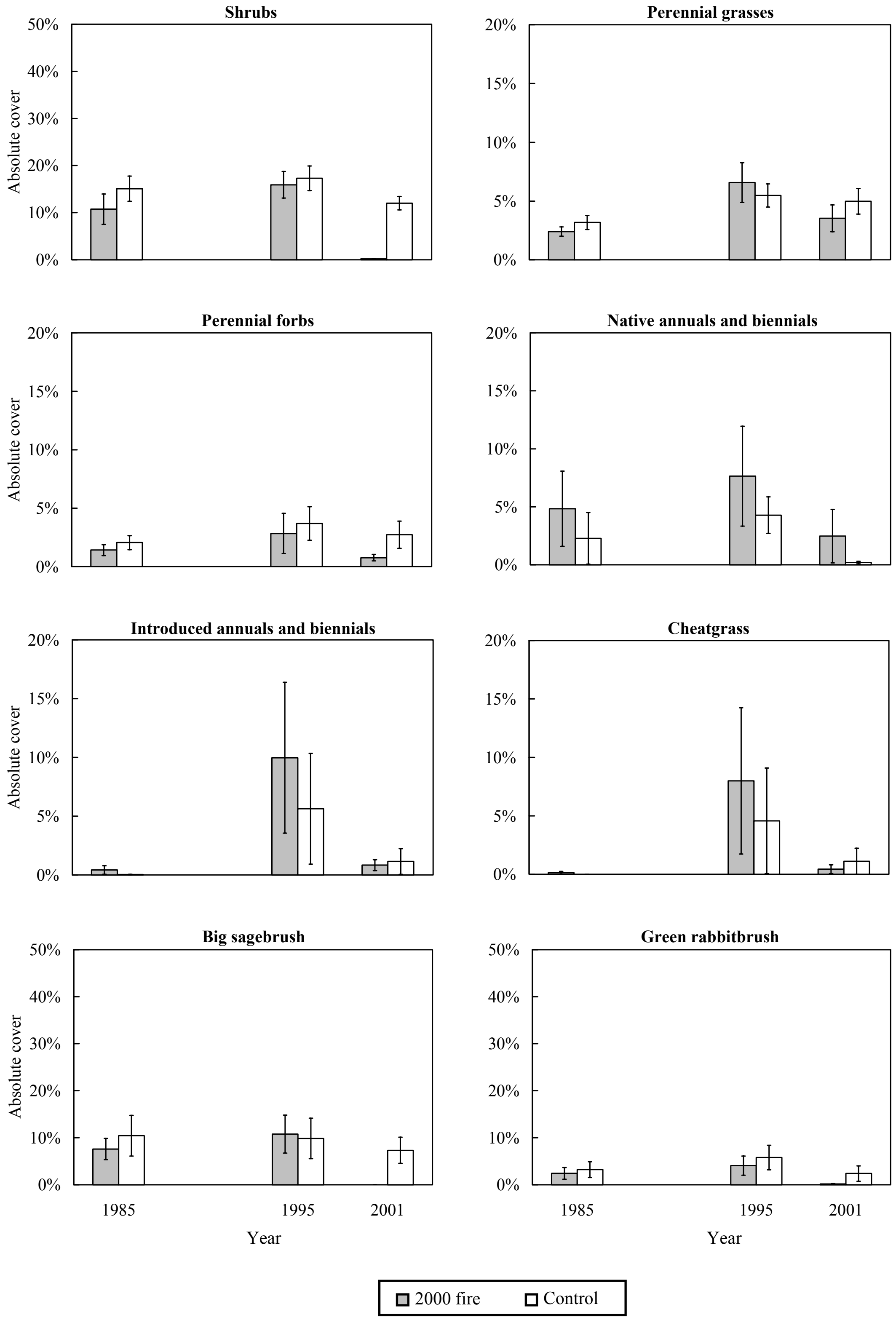

Figure 2.9. Absolute cover of species groups and common perennial at plots burned in the 2000 fire (N=4) and control plots (N=4). Data are mean values of point-interception data expressed in percentages and error bars are ±1 standard error.

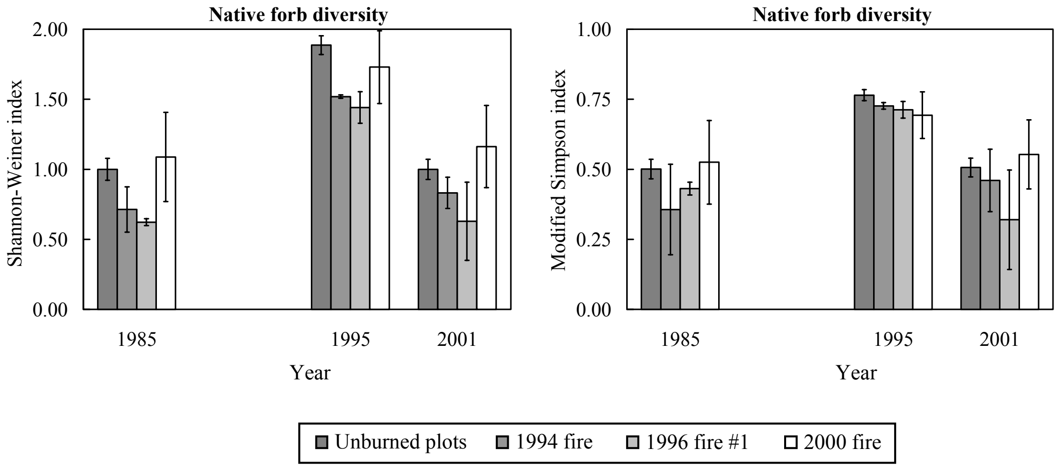

Figure 2.10. Native forb diversity using the Shannon-Weiner (H') and modified Simpson (D) indices at unburned plots (N=49), and plots burned in the 1994 fire (N=3), 1996 fire #1 (N=2) and 2000 fire (N=4). Data are based on mean values of point-interception data and error bars are ±1 standard error.

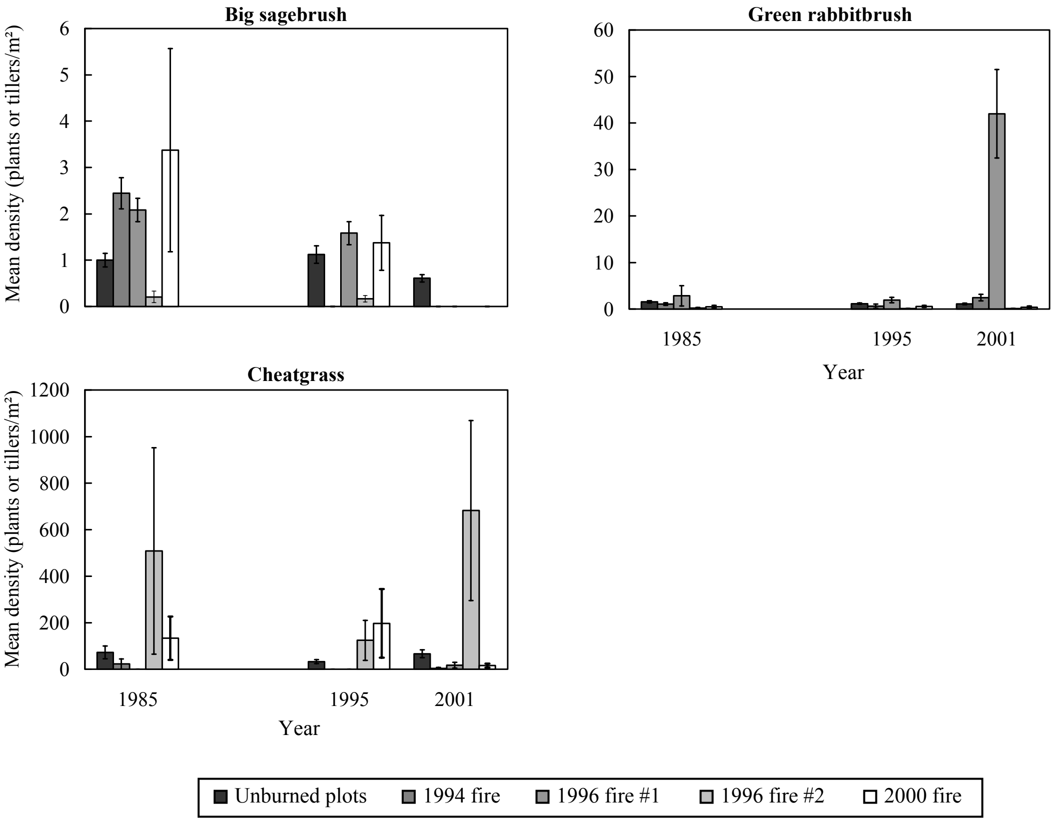

Figure 2.11. Mean density of common perennial species at unburned plots (N=74), and plots burned in the 1994 fire (N=3), 1996 fire #1 (N=2), 1996 fire #2 (N=4), and 2000 fire (N=4). Data are based on mean values of density data and error bars are ±1 standard error.

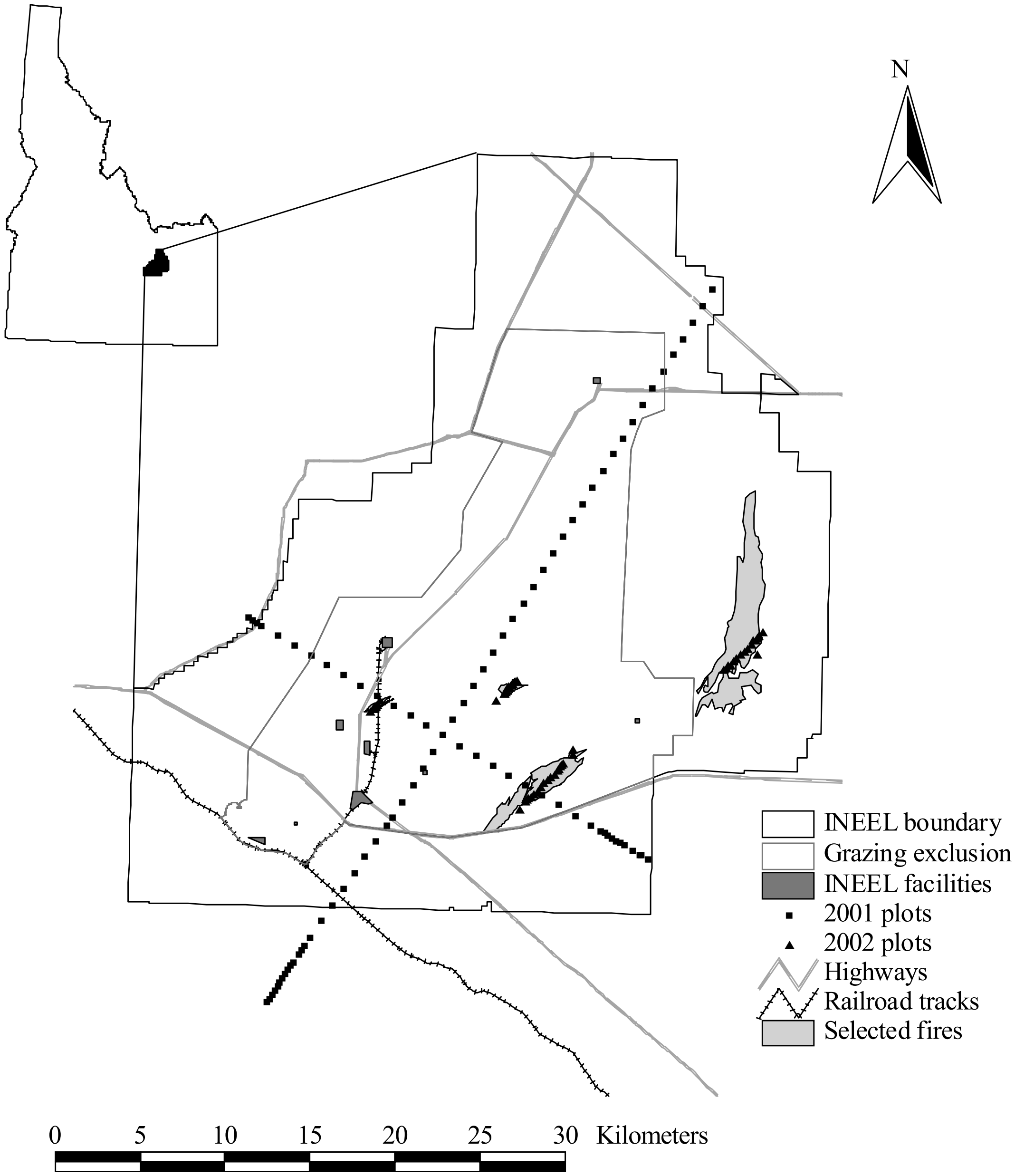

Figure 3.1. Map of the Idaho National Engineering and Environmental Laboratory (INEEL) showing the locations of plots sampled in 2001 and 2002, and the fires associated with the 2002 plots.

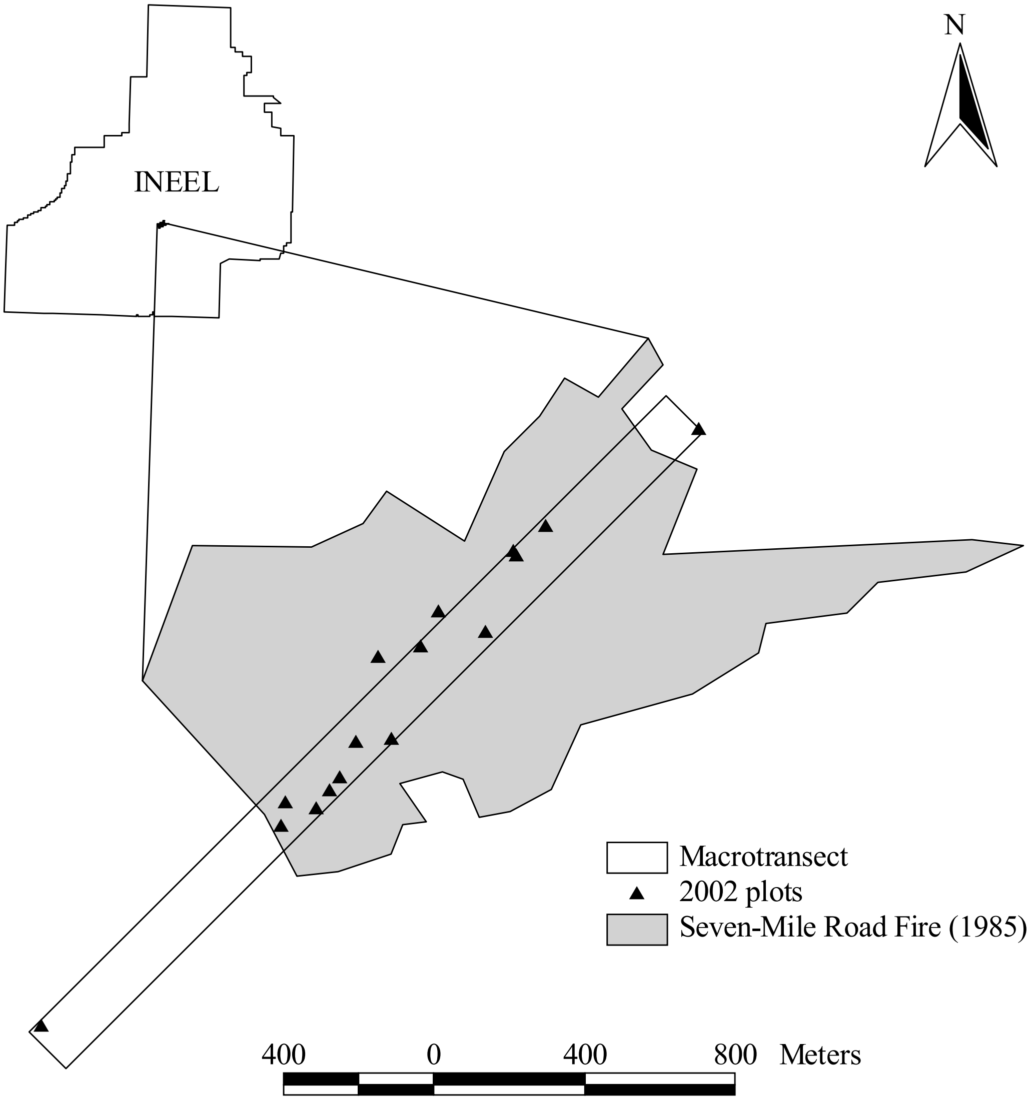

Figure 3.2. Map of the Seven-Mile Road Fire (1985) showing the locations of the macrotransect and 2002 plots associated with this fire. The inset in the upper-left corner shows the placement of this fire within the Idaho National Engineering and Environmental Laboratory (INEEL).

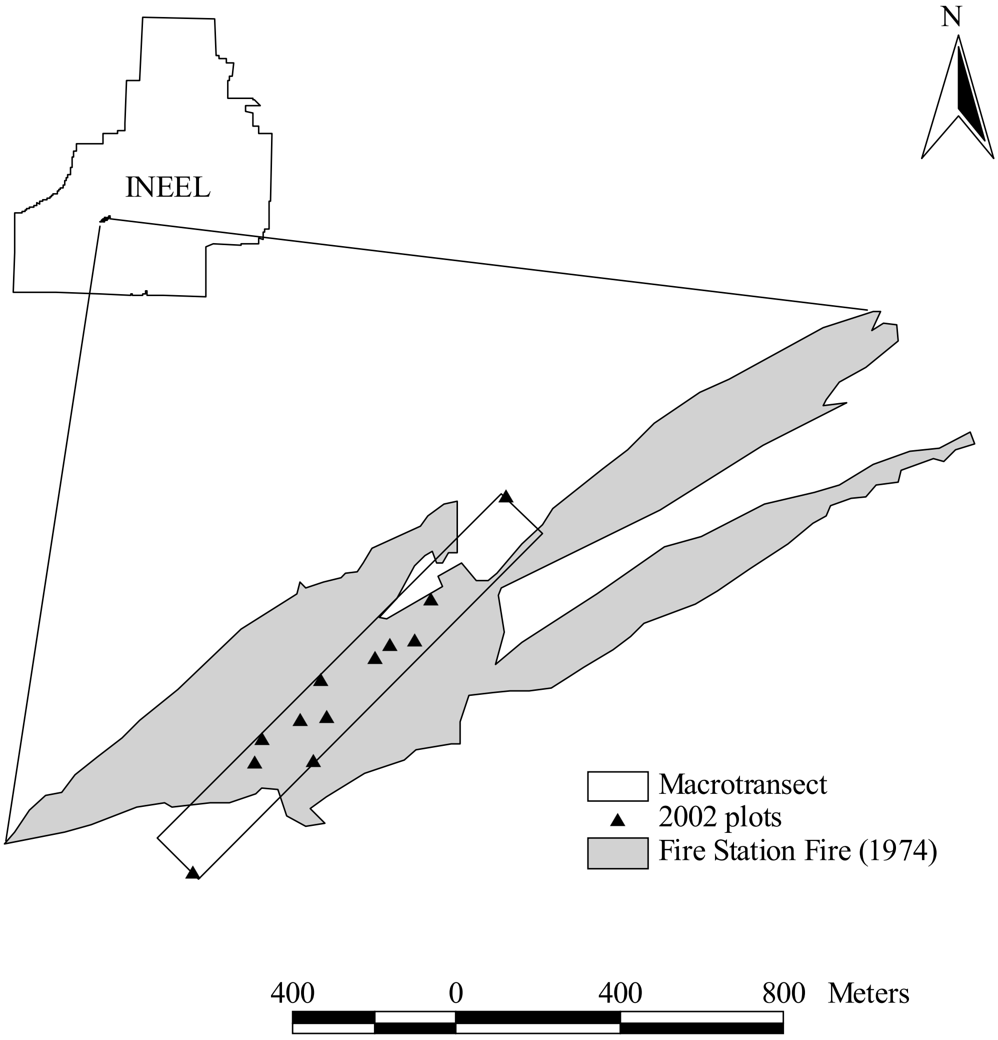

Figure 3.3. Map of the Fire Station Fire (1974) showing the locations of the macrotransect and 2002 plots associated with this fire. The inset in the upper-left corner shows the placement of this fire within the Idaho National Engineering and Environmental Laboratory (INEEL).

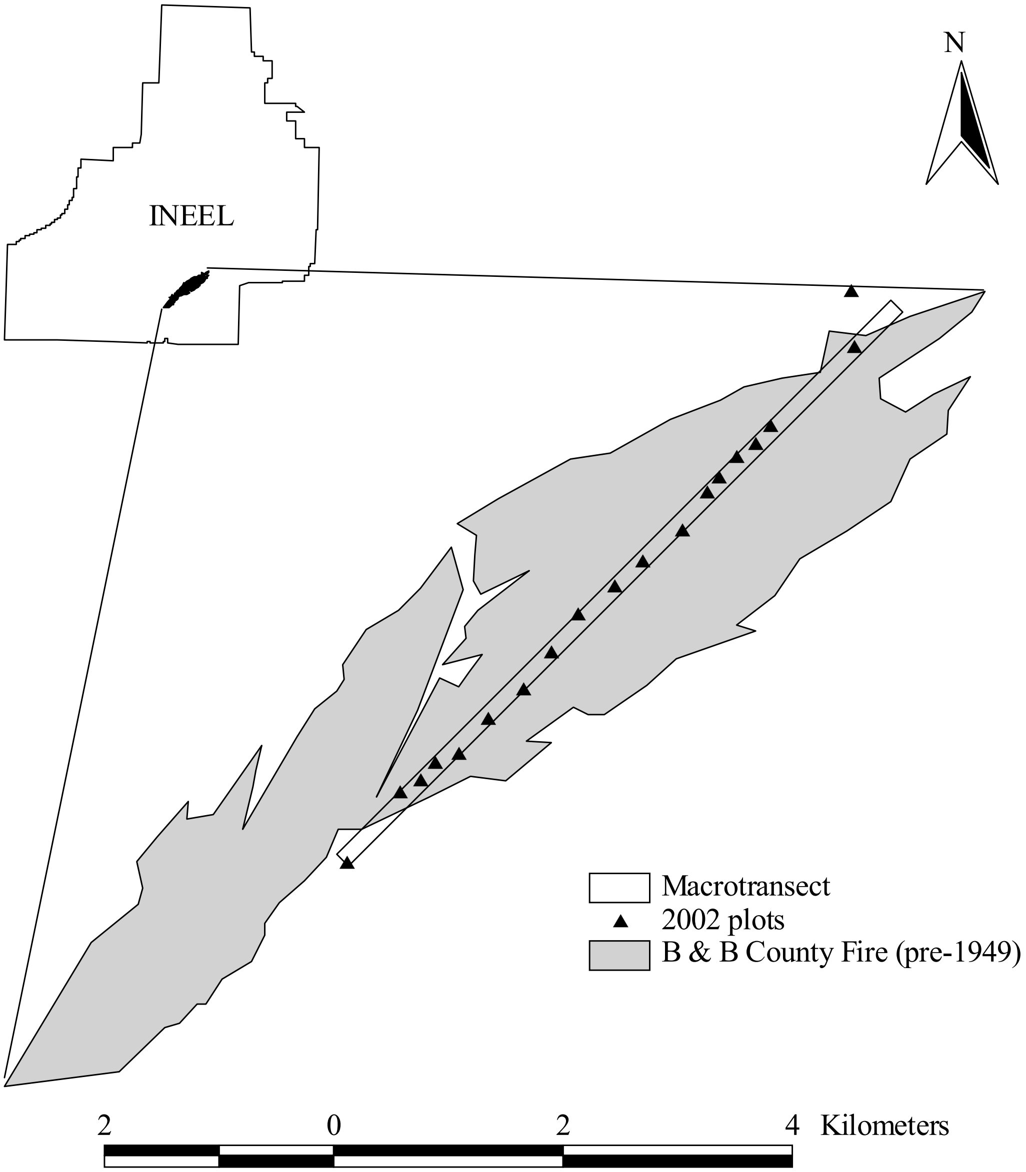

Figure 3.4. Map of the B & B County Fire (pre-1949) showing the locations of the macrotransect and 2002 plots associated with this fire. The inset in the upper-left corner shows the placement of this fire within the Idaho National Engineering and Environmental Laboratory (INEEL).

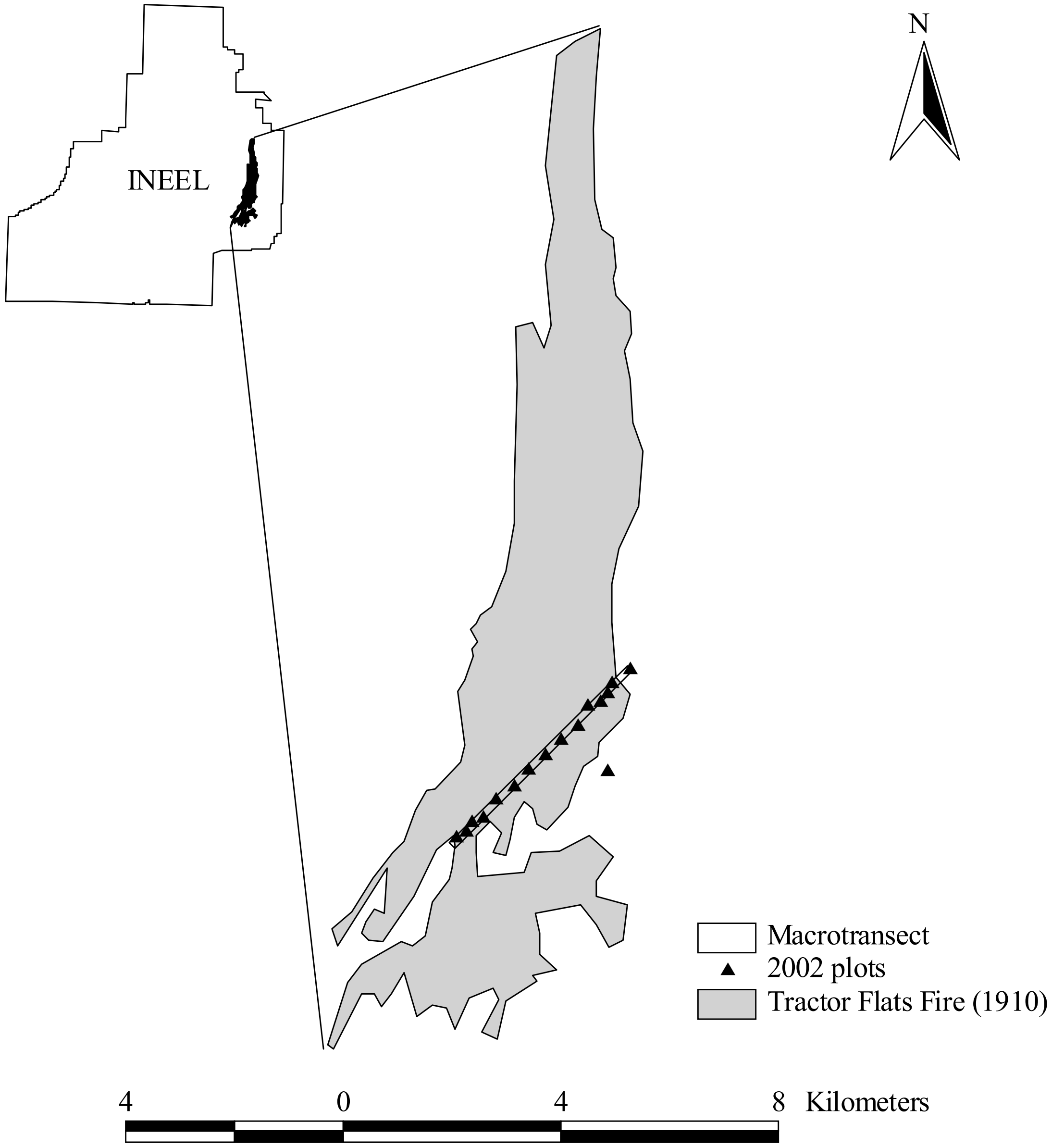

Figure 3.5. Map of the Tractor Flats Fire (1910) showing the locations of the macrotransect and 2002 plots associated with this fire. The inset in the upper-left corner shows the placement of this fire within the Idaho National Engineering and Environmental Laboratory (INEEL).

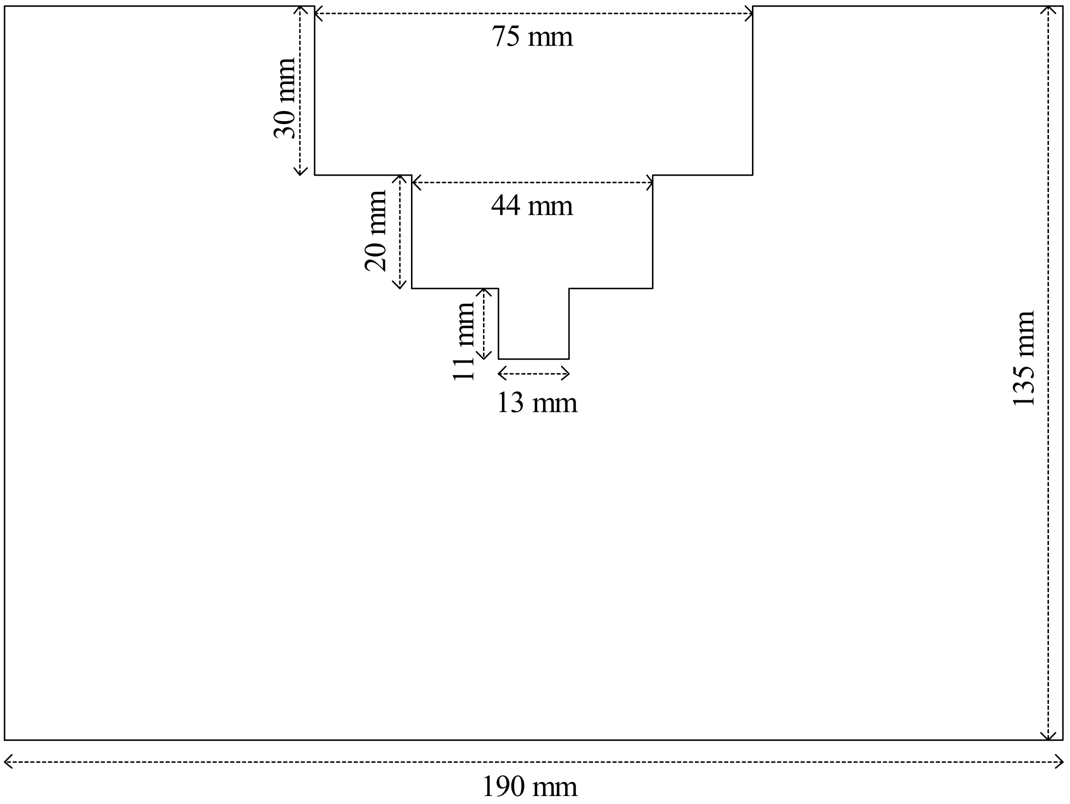

Figure 3.6. Diagram showing the big sagebrush size class gauge used to measure big sagebrush stem diameter for age estimation (reduced size).

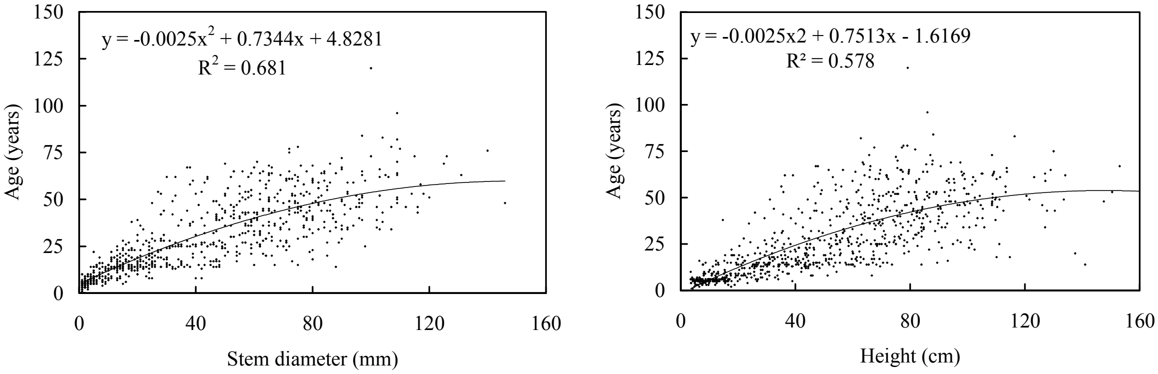

Figure 3.7. Figure showing correlation between big sagebrush age and stem diameter (N=818) on the left and height (N=802) on the right without natural log transformation. Regression equation and R² value are located in upper-left corner.

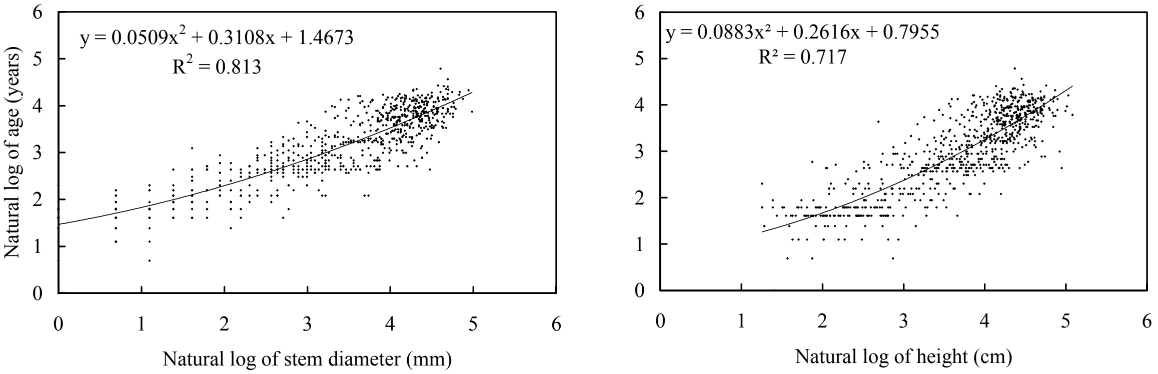

Figure 3.8. Figure showing correlation between big sagebrush age and stem diameter (N=818) on the left and height (N=802) on the right with natural log transformation.

Table 3.1. Table showing big sagebrush stem diameter size classes, original age classes used to determine size classes, and true age classes determined from cross-sections sampled in current study (after natural log transformation).

Size classes (mm)	0-13	13-44	44-75	75+
Original age classes (years)	0-10	10-20	20-30	30+
New age classes (years)	0-13.5	13.5-29.1	29.1-42.9	42.9+

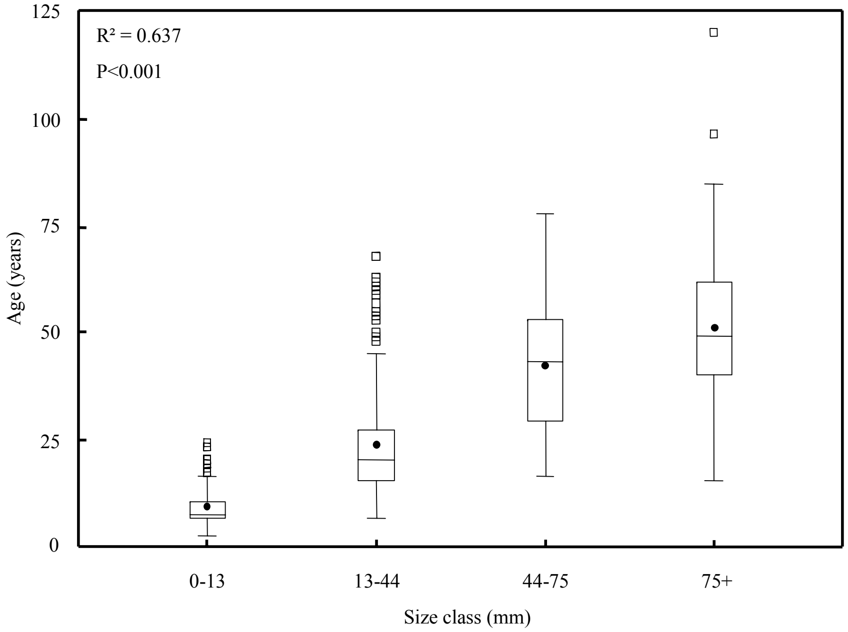

Figure 3.9. Figure showing box plots for big sagebrush stem diameter size classes (N=818).

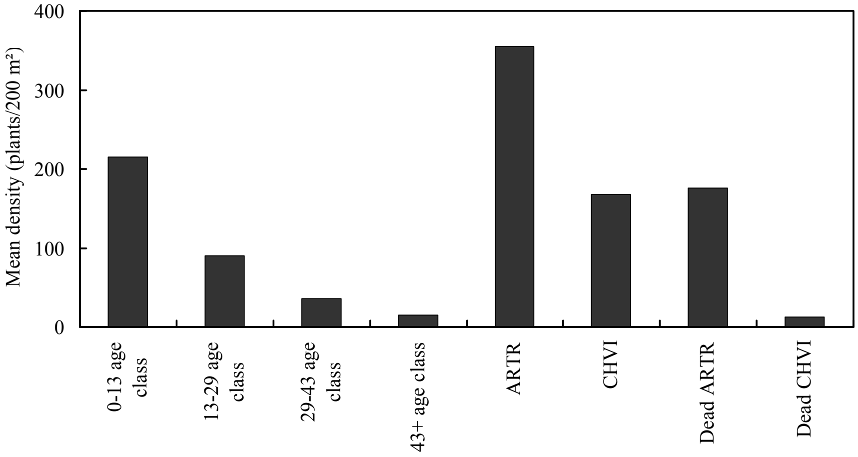

Figure 3.10. Figure showing mean density at control plots (N=7) for the four big sagebrush size classes, live and dead big sagebrush (ARTR), and green rabbitbrush (CHVI).

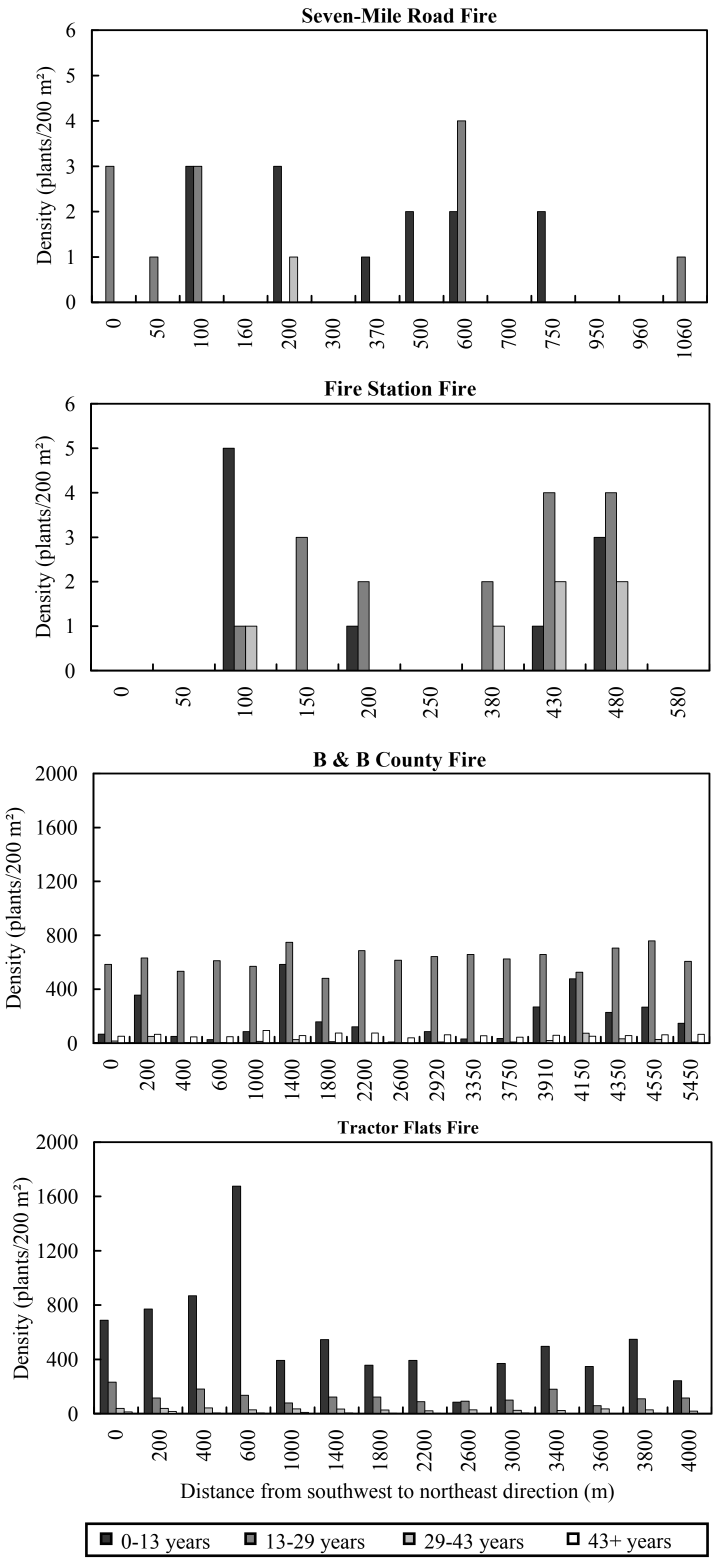

Figure 3.11. Figure showing density of the four big sagebrush size classes at plots within the Fire Station Fire (N=10), Seven-Mile Road Fire (N=14), B & B County Fire (N=17), and Tractor Flats Fire (N=14).

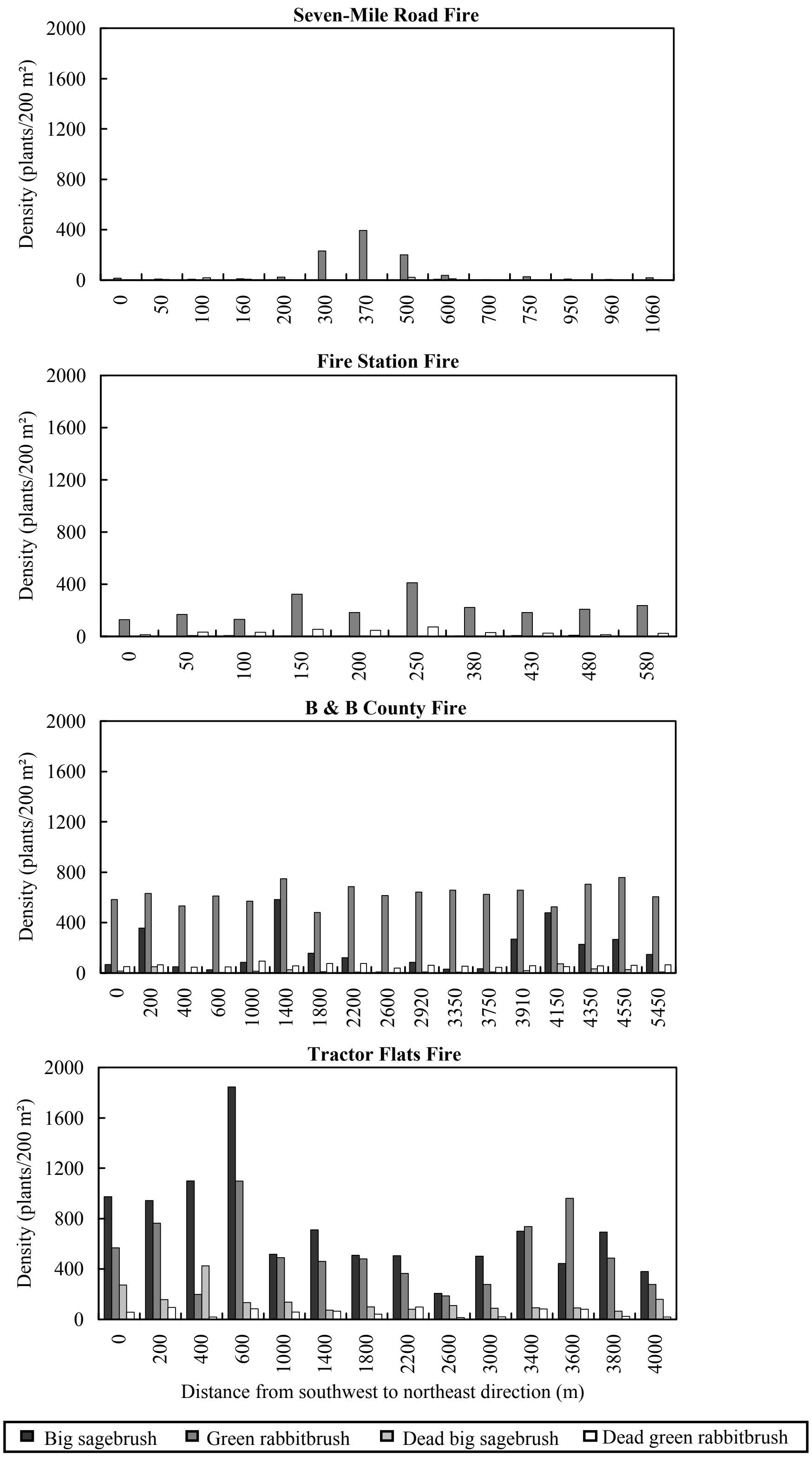

Figure 3.12. Figure showing density of live and dead big sagebrush and green rabbitbrush at plots within the Fire Station Fire (N=10), Seven-Mile Road Fire (N=14), B & B County Fire (N=17), and Tractor Flats Fire (N=14).

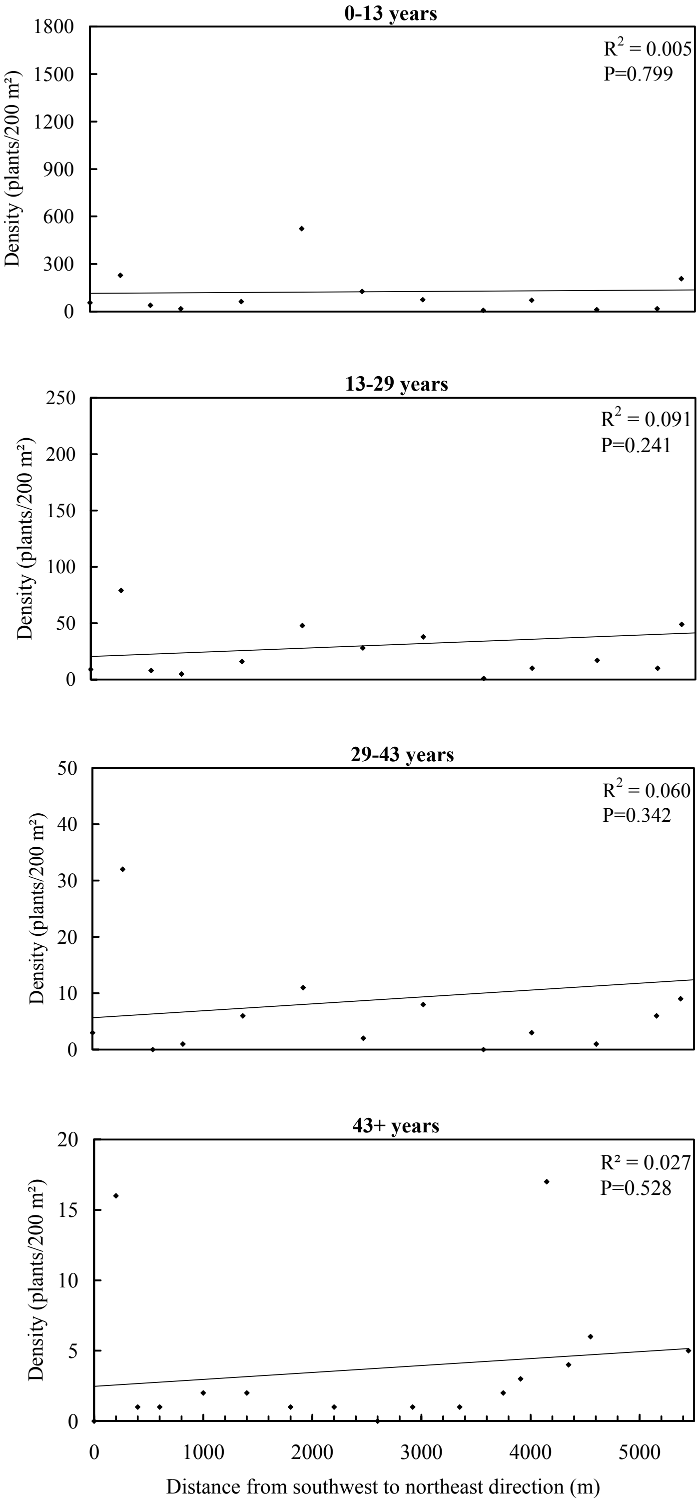

Figure 3.13. Figure showing density of big sagebrush for the 0-13, 13-29, 29-43, and 43+ year size classes at plots across the B & B County Fire (N=17) from the southwest to northeast direction.

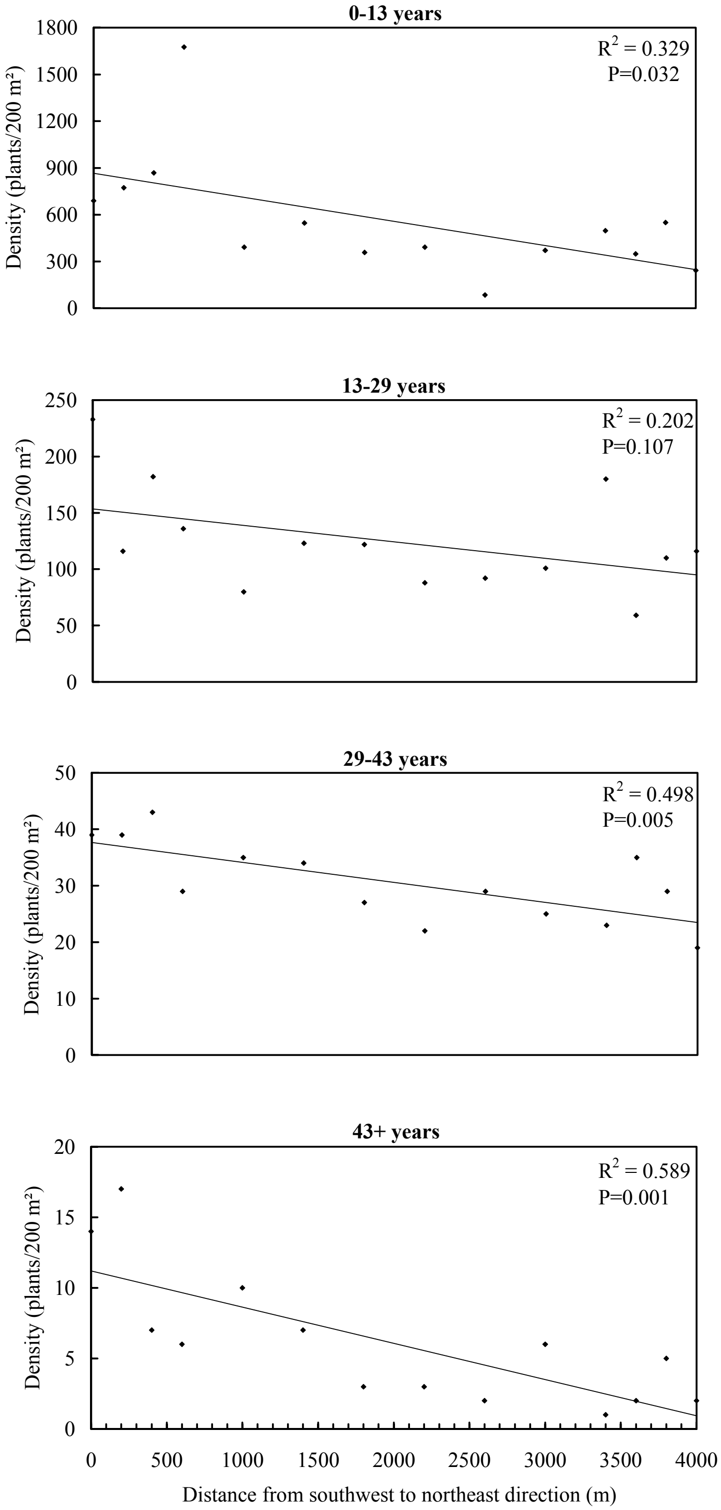

Figure 3.14. Figure showing density of big sagebrush for the 0-13, 13-29, 29-43, and 43+ year size classes at plots across the Tractor Flats Fire (N=14) from the southwest to northeast direction.

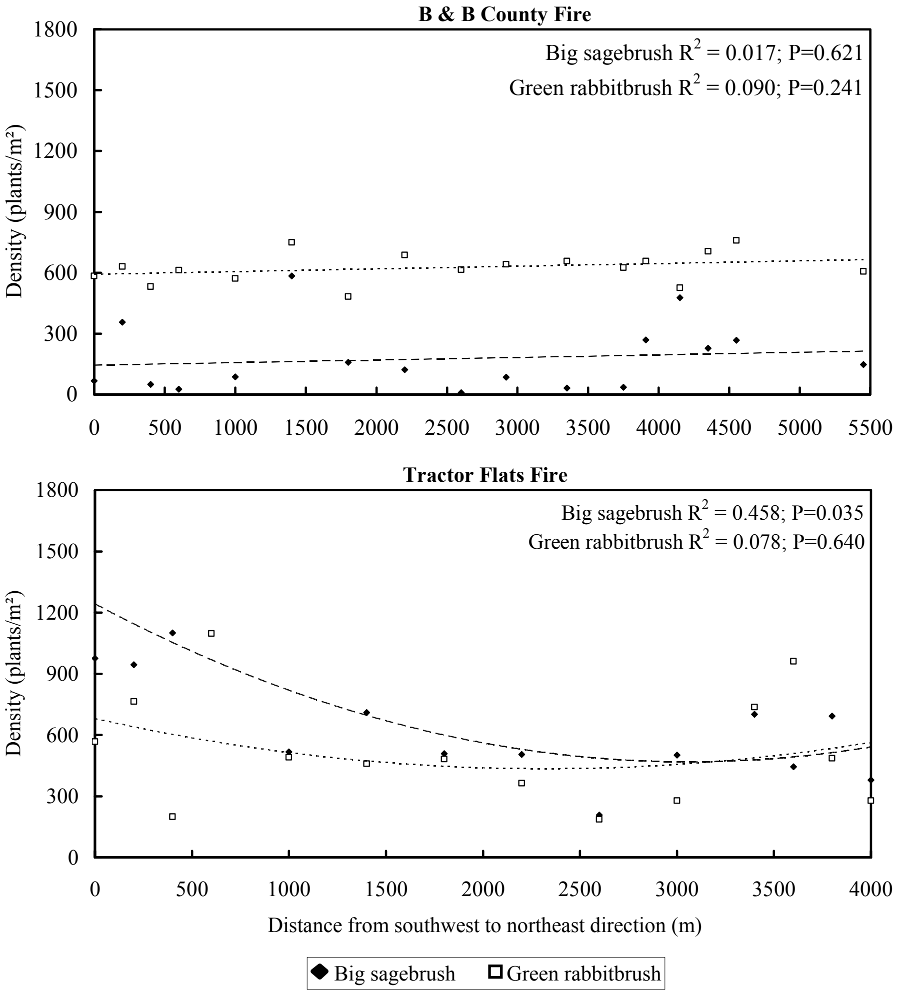

Figure 3.15. Figure showing density of big sagebrush and green rabbitbrush at plots within the B & B County Fire (N=17) and Tractor Flats Fire (N=14). Wide and narrow dashed lines depict trendlines for big sagebrush and green rabbitbrush, respectively.

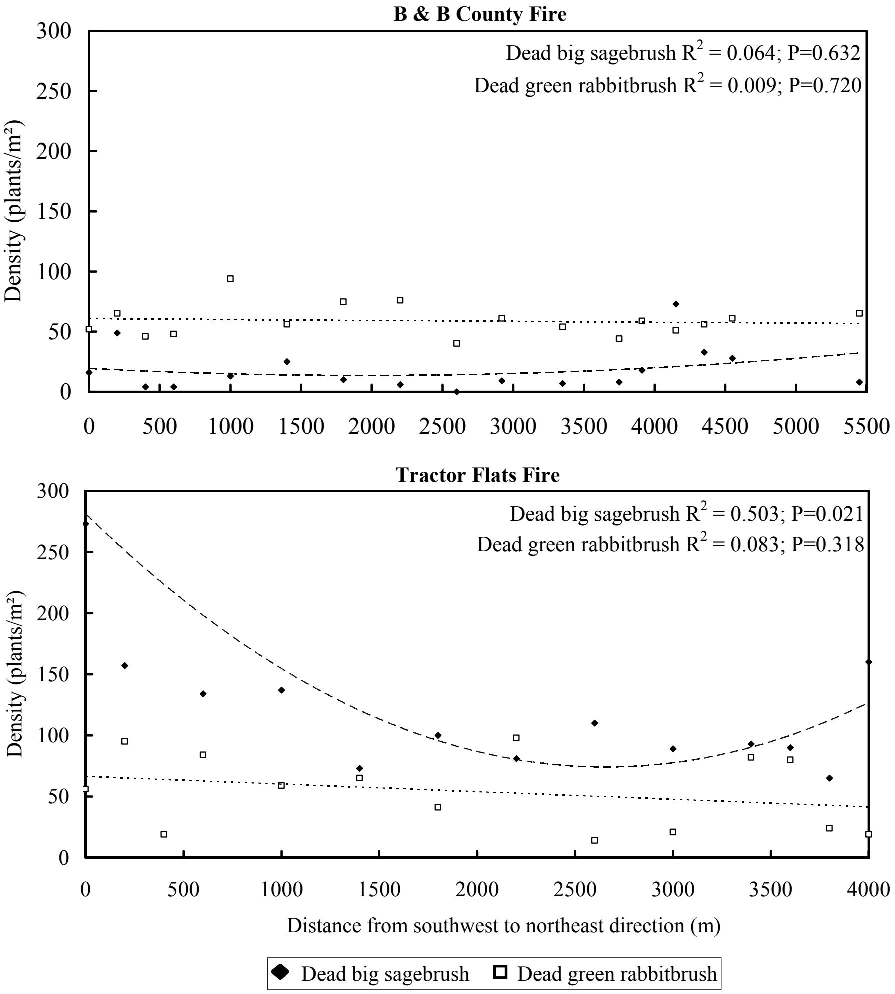

Figure 3.16. Figure showing density of dead big sagebrush and dead green rabbitbrush at plots within the B & B County Fire (N=17) and Tractor Flats Fire (N=14). Wide and narrow dashed lines depict trendlines for dead big sagebrush and dead green rabbitbrush, respectively.

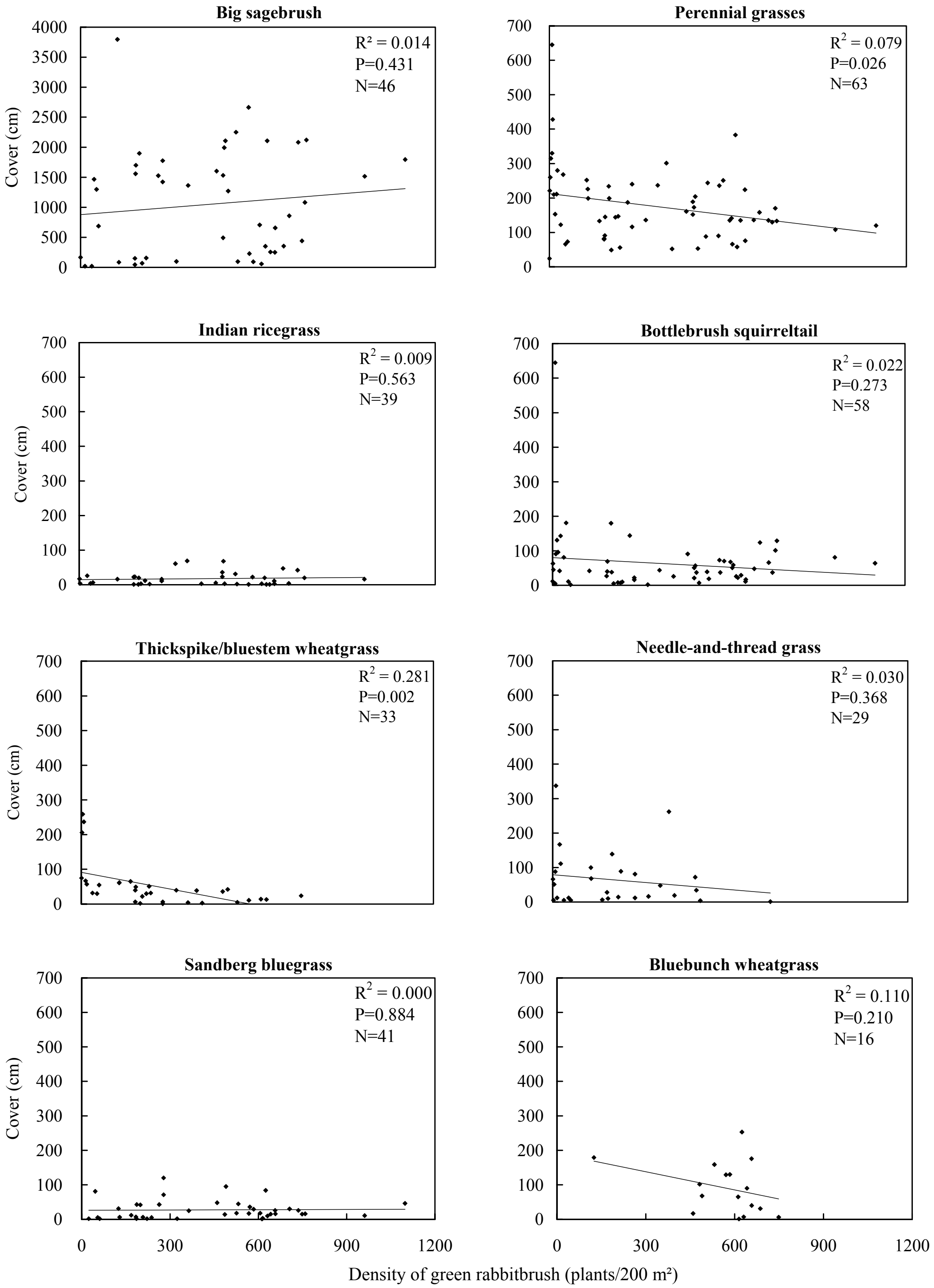

Figure 3.17. Relationship between density of green rabbitbrush and cover of perennial grasses and selected perennial grass and shrub species. Plots with cover values greater than zero are plotted against density of green rabbitbrush. Cover is based on line-interception data.

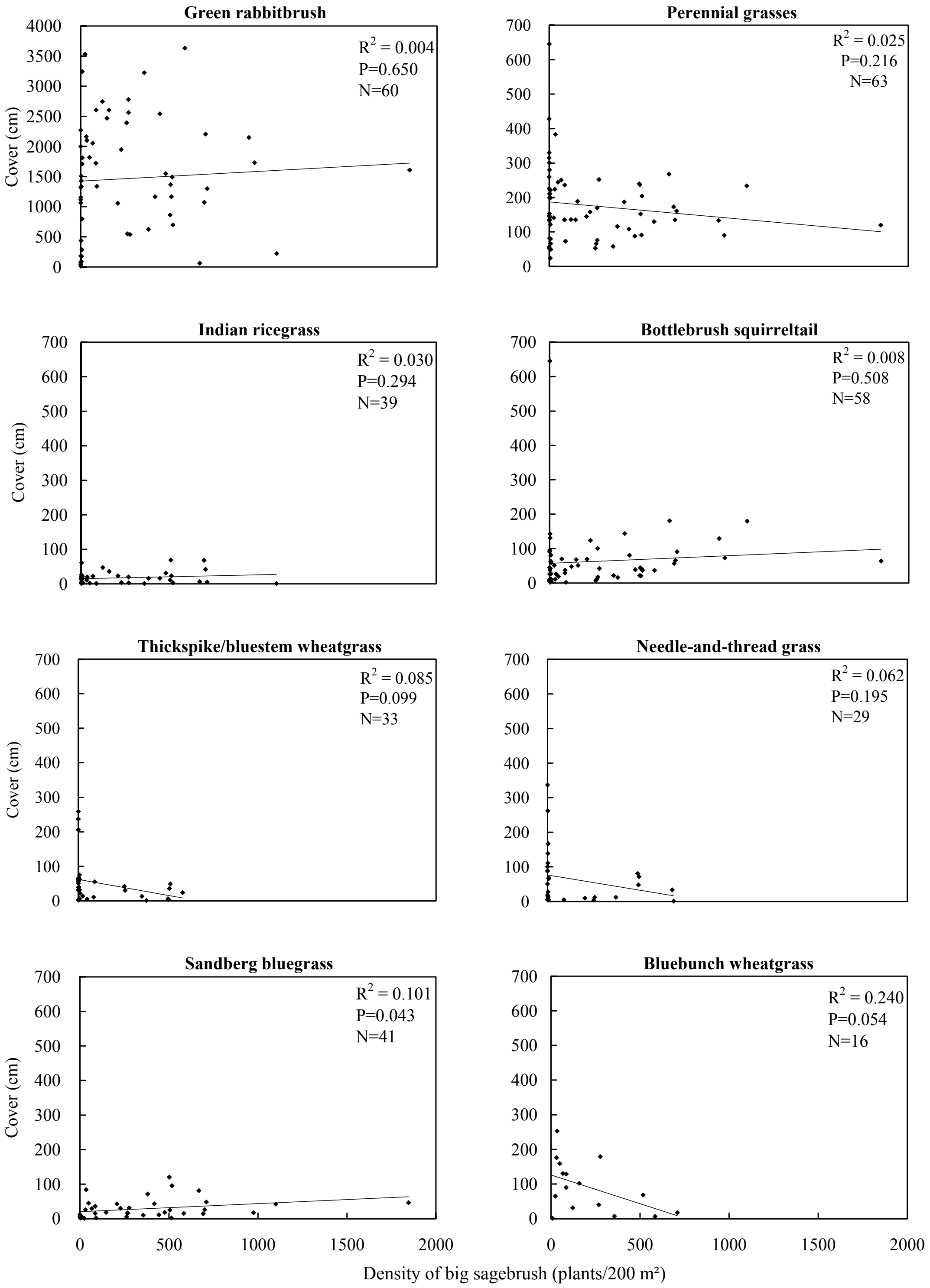

Figure 3.18. Relationship between density of big sagebrush and cover of perennial grasses and common perennial species. Plots with cover values greater than zero are plotted against density of big sagebrush. Cover is based on line-interception data.

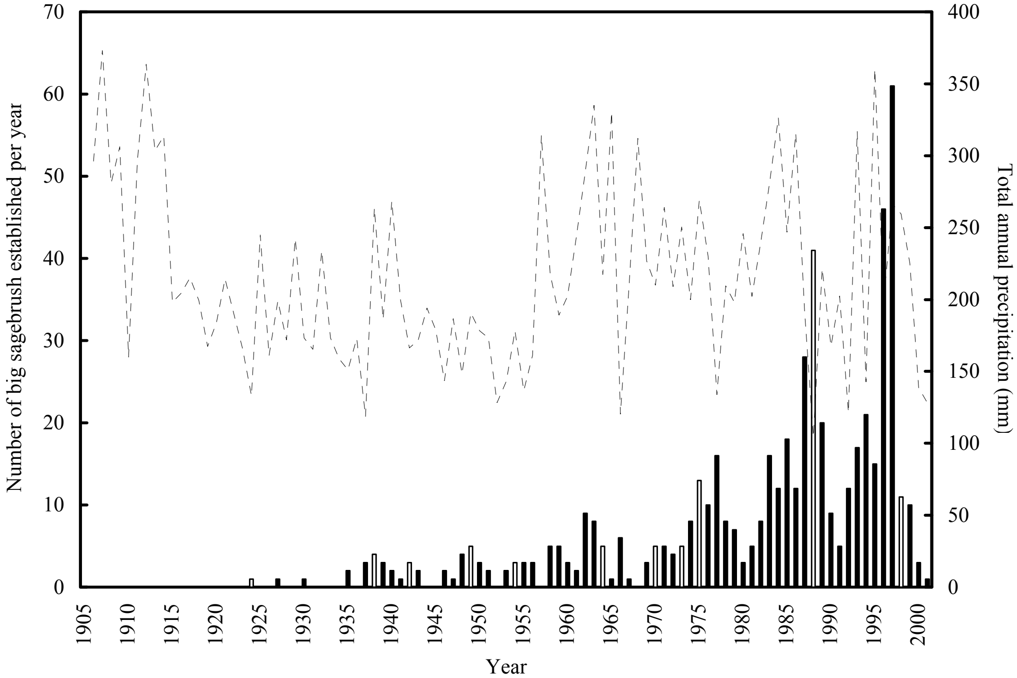

Figure 4.1. Figure showing total number of big sagebrush establishing each year (N=554) in relation to years with La Niña events (depicted by white columns). Dashed line represents total annual water-year precipitation (October-September) for 1905-2001.

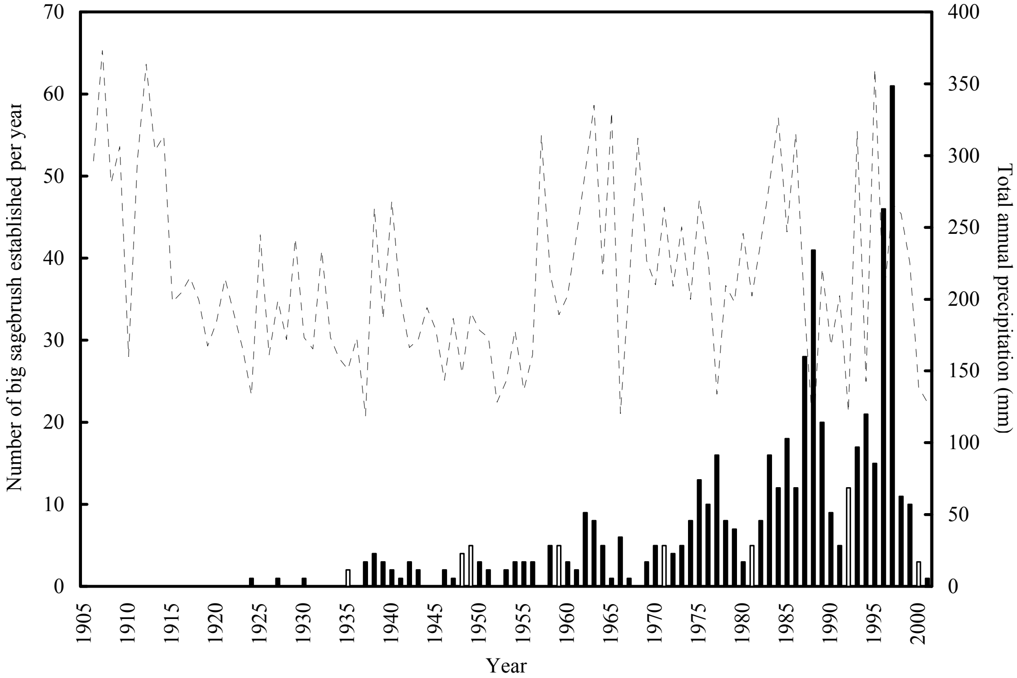

Figure 4.2. Figure showing total number of big sagebrush establishing each year (N=554) in relation to years with peak densities of black-tailed jackrabbit populations (depicted by white columns). Dashed line represents total annual water-year precipitation (October-September) for 1905-2001.

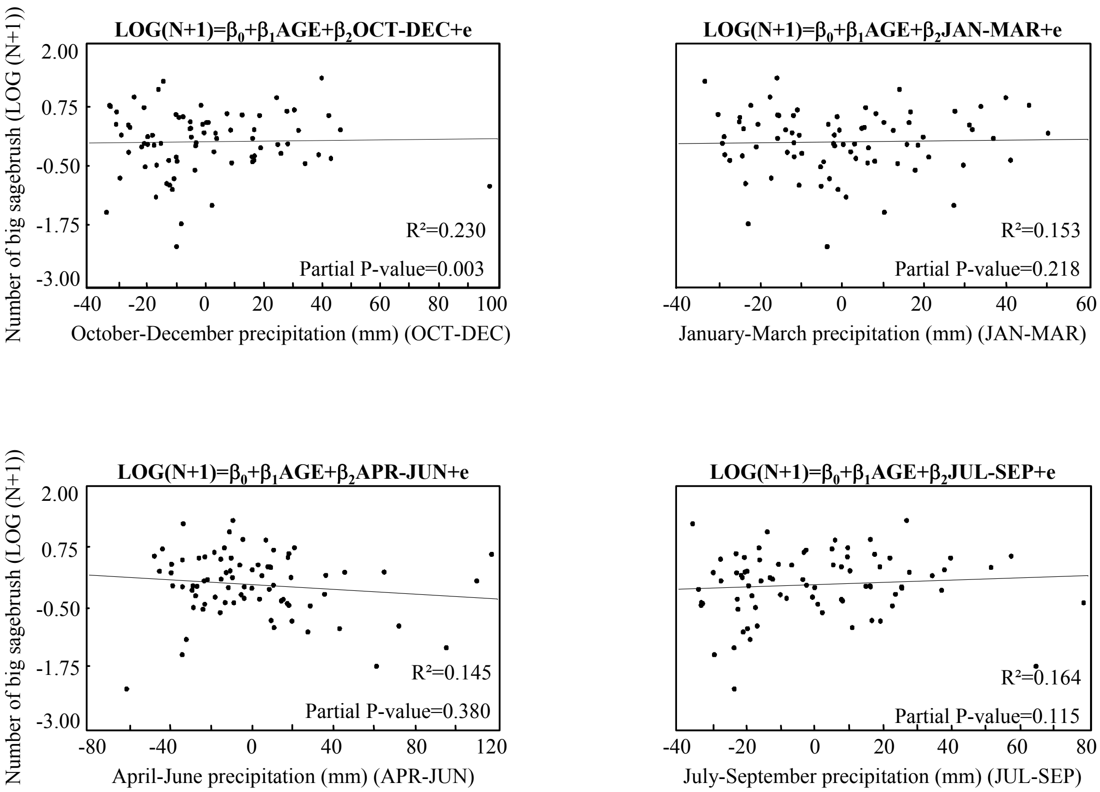

Figure 4.3. Figure showing partial regression plots showing the correlation between the number of big sagebrush establishing each year and total seasonal precipitation (October-December, January-March, April-June, July-September). Total seasonal precipitation is based on water-year precipitation (October-September) during the year of big sagebrush establishment. The regression equation used to develop each model is shown above each graph.

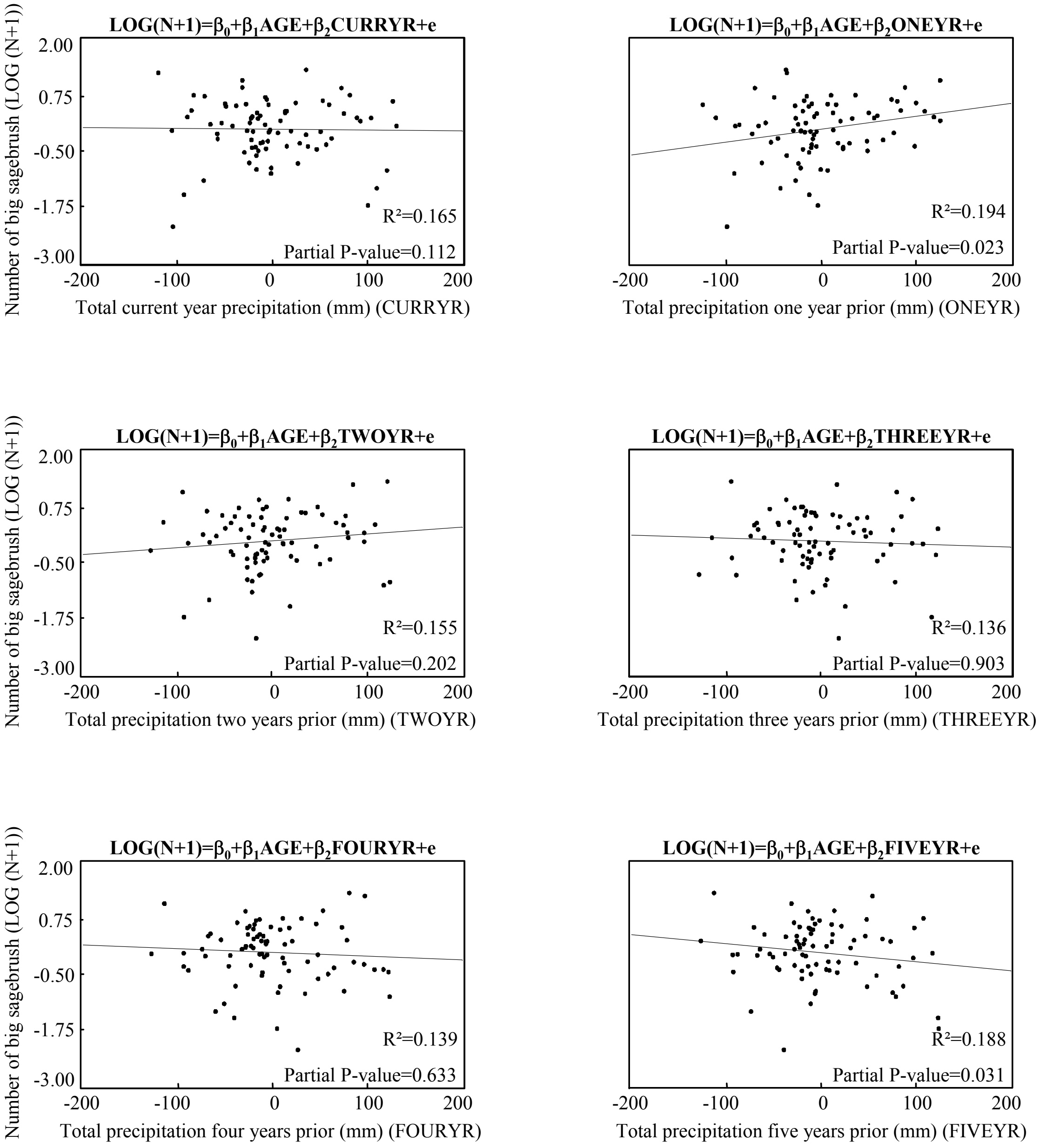

Figure 4.4. Figure showing partial regression plots showing the correlation between the number of big sagebrush establishing each year and total annual current year precipitation, and total annual precipitation for one, two, three, four, and five years prior to germination. Total annual precipitation is based on water-year precipitation (October-September) during the year of big sagebrush establishment. The regression equation used to develop each model is shown above each graph.

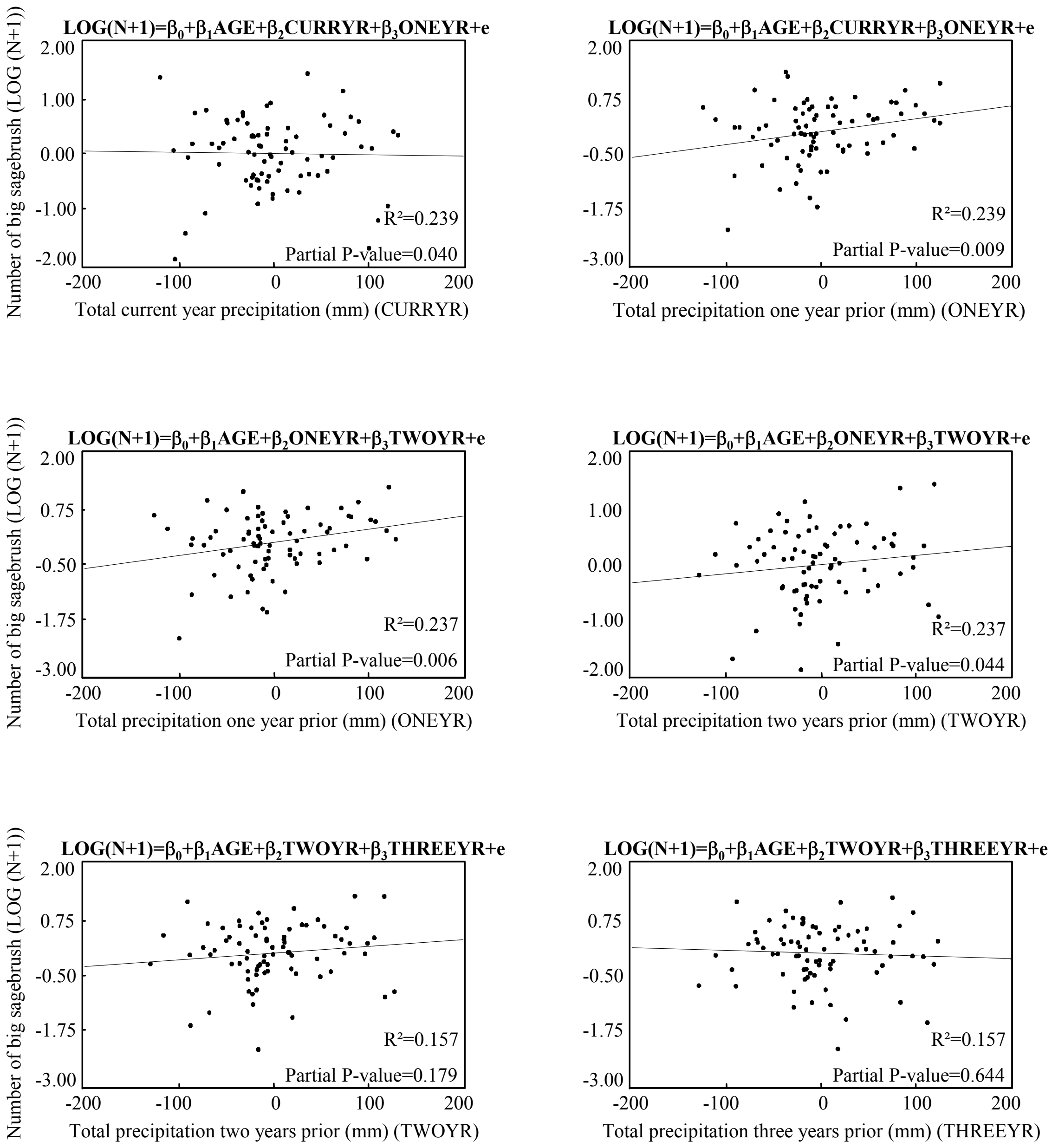

Figure 4.5. Figure showing partial regression plots showing the correlation between the number of big sagebrush establishing each year and total annual precipitation for two consecutive years (current year and one year prior to establishment [top], one and two years prior to establishment [center], and two and three years prior to establishment [bottom]). Total annual precipitation is based on water-year precipitation (October-September) during the year of big sagebrush establishment. The regression equation used to develop each model is shown above each graph.

APPENDIX I

UTM Coordinates of Vegetation Plots

Datum: NAD27 CONUS

Name	Plot	Date	Time	Easting	Northing
Long-term vegetation study	1	24-Jul-01	22:15	335440	4804395
Long-term vegetation study	2	24-Jul-01	21:01	335653	4804707
Long-term vegetation study	3	24-Jul-01	19:11	335877	4805074
Long-term vegetation study	4	24-Jul-01	17:35	336089	4805416
Long-term vegetation study	5	25-Jul-01	15:13	336303	4805757
Long-term vegetation study	6	24-Jul-01	0:59	336483	4806067
Long-term vegetation study	7	23-Jul-01	22:04	336695	4806440
Long-term vegetation study	8	23-Jul-01	19:04	336940	4806794
Long-term vegetation study	9	23-Jul-01	17:02	337149	4807083
Long-term vegetation study	10	23-Jul-01	14:16	337376	4807433
Long-term vegetation study	11	19-Jul-01	19:00	337627	4807770
Long-term vegetation study	13	19-Jul-01	16:29	338080	4808428
Long-term vegetation study	14	19-Jul-01	13:10	338304	4808762
Long-term vegetation study	15	18-Jul-01	18:20	338535	4809098
Long-term vegetation study	17	18-Jul-01	16:00	338987	4809753
Long-term vegetation study	18	18-Jul-01	13:34	339986	4811207
Long-term vegetation study	19	27-Jun-01	18:32	340900	4812529
Long-term vegetation study	20	27-Jun-01	15:17	341809	4813858
Long-term vegetation study	21	27-Jun-01	12:44	342721	4815179
Long-term vegetation study	22	22-Jun-01	12:49	343632	4816506
Long-term vegetation study	23	21-Jun-01	17:42	344547	4817832
Long-term vegetation study	24	21-Jun-01	15:29	345460	4819157
Long-term vegetation study	25	18-Jun-01	16:26	345936	4819860
Long-term vegetation study	26	18-Jun-01	14:03	346768	4821149
Long-term vegetation study	27	9-Jul-01	16:14	347603	4822522
Long-term vegetation study	28	9-Jul-01	13:38	348426	4823908
Long-term vegetation study	29	7-Jun-01	23:43	349274	4825272
Long-term vegetation study	30	7-Jun-01	19:25	350107	4826650
Long-term vegetation study	31	15-Jun-01	13:04	350938	4828025
Long-term vegetation study	32	12-Jun-01	22:19	351791	4829423
Long-term vegetation study	33	12-Jun-01	22:11	352625	4830799
Long-term vegetation study	34	12-Jun-01	22:05	353459	4832177

UTM Coordinates of Vegetation Plots (continued)

Name	Plot	Date	Time	Easting	Northing
Long-term vegetation study	35	12-Jun-01	21:56	354291	4833552
Long-term vegetation study	36	12-Jun-01	17:15	355138	4834926
Long-term vegetation study	37	12-Jun-01	13:21	355958	4836306
Long-term vegetation study	38	5-Jun-02	20:59	356795	4837683
Long-term vegetation study	39	25-Jun-01	13:21	357629	4839060
Long-term vegetation study	40	25-Jun-01	16:33	358463	4840433
Long-term vegetation study	41	26-Jun-01	13:10	359296	4841807
Long-term vegetation study	42	26-Jun-01	15:32	360129	4843193
Long-term vegetation study	43	26-Jun-01	17:32	360962	4844563
Long-term vegetation study	44	28-Jun-01	12:59	361775	4845913
Long-term vegetation study	45	28-Jun-01	15:27	362611	4847289
Long-term vegetation study	46	28-Jun-01	17:44	363449	4848670
Long-term vegetation study	47	29-Jun-01	13:01	364281	4850043
Long-term vegetation study	48	29-Jun-01	20:08	365111	4851422
Long-term vegetation study	49	29-Jun-01	17:48	365948	4852794
Long-term vegetation study	50	13-Jul-01	12:52	366777	4854170
Long-term vegetation study	51	13-Jul-01	16:24	367613	4855555
Long-term vegetation study	52	16-Jul-01	12:54	368453	4856919
Long-term vegetation study	53	16-Jul-01	14:56	369281	4858304
Long-term vegetation study	54	16-Jul-01	17:12	370113	4859675
Long-term vegetation study	55	20-Jul-01	14:00	370946	4861059
Long-term vegetation study	56	20-Jul-01	15:52	371780	4862429
Long-term vegetation study	57	20-Jul-01	17:48	372612	4863804
Long-term vegetation study	66	12-Jul-01	16:24	367221	4816279
Long-term vegetation study	67	12-Jul-01	14:52	366653	4816618
Long-term vegetation study	68	12-Jul-01	13:11	366508	4816714
Long-term vegetation study	69	11-Jul-01	18:40	365929	4817048
Long-term vegetation study	70	11-Jul-01	15:05	365251	4817470
Long-term vegetation study	71	10-Jul-01	19:04	364898	4817685
Long-term vegetation study	72	10-Jul-01	15:29	364554	4817892
Long-term vegetation study	73	3-Jul-01	20:14	364221	4818095
Long-term vegetation study	74	3-Jul-01	17:32	363860	4818319
Long-term vegetation study	75	3-Jul-01	14:37	363549	4818503
Long-term vegetation study	76	3-Jul-01	12:04	363204	4818705
Long-term vegetation study	77	2-Jul-01	16:29	362490	4819141
Long-term vegetation study	78	2-Jul-01	12:42	361109	4819975
Long-term vegetation study	79	21-Jun-01	12:42	359734	4820812

UTM Coordinates of Vegetation Plots (continued)

Name	Plot	Date	Time	Easting	Northing
Long-term vegetation study	80	20-Jun-01	19:43	358365	4821642
Long-term vegetation study	81	20-Jun-01	16:05	356988	4822477
Long-term vegetation study	82	13-Jun-01	15:13	355607	4823314
Long-term vegetation study	83	20-Jun-01	12:58	354234	4824145
Long-term vegetation study	84	9-Jun-01	18:55	352859	4824981
Long-term vegetation study	85	9-Jun-01	13:13	351481	4825813
Long-term vegetation study	86	15-Jun-01	13:27	348731	4827483
Long-term vegetation study	87	8-Jun-01	16:28	347380	4828299
Long-term vegetation study	88	8-Jun-01	12:49	346005	4829132
Long-term vegetation study	89	14-Jun-01	16:01	344629	4829969
Long-term vegetation study	90	14-Jun-01	16:19	343257	4830800
Long-term vegetation study	91	14-Jun-01	19:31	341857	4831649
Long-term vegetation study	92	19-Jun-01	19:26	340481	4832486
Long-term vegetation study	93	19-Jun-01	16:22	339106	4833320
Long-term vegetation study	94	19-Jun-01	12:56	337729	4834149
Long-term vegetation study	95	9-Jul-01	19:03	336354	4834984
Long-term vegetation study	96	17-Jul-01	12:35	334977	4835819
Long-term vegetation study	97	17-Jul-01	12:44	334634	4836028
Long-term vegetation study	98	17-Jul-01	17:01	334294	4836232
Long-term vegetation study	99	17-Jul-01	17:12	333955	4836437
Seven-Mile Road Fire	7M-01	18-Jun-02	18:03	354535	4829624
Seven-Mile Road Fire	7M-02	29-May-02	3:41	355175	4830158
Seven-Mile Road Fire	7M-03	29-May-02	3:43	355184	4830220
Seven-Mile Road Fire	7M-04	29-May-02	3:47	355269	4830205
Seven-Mile Road Fire	7M-05	31-May-02	17:00	355302	4830252
Seven-Mile Road Fire	7M-06	3-Jun-02	18:47	355329	4830285
Seven-Mile Road Fire	7M-07	3-Jun-02	19:45	355372	4830381
Seven-Mile Road Fire	7M-08	5-Jun-02	17:48	355467	4830388
Seven-Mile Road Fire	7M-09	20-Jun-02	17:22	355433	4830605
Seven-Mile Road Fire	7M-10	20-Jun-02	15:19	355546	4830635
Seven-Mile Road Fire	7M-11	20-Jun-02	13:35	355591	4830728
Seven-Mile Road Fire	7M-12	17-Jun-02	19:05	355718	4830674
Seven-Mile Road Fire	7M-13	17-Jun-02	22:27	355801	4830875
Seven-Mile Road Fire	7M-14	17-Jun-02	14:34	355792	4830889
Seven-Mile Road Fire	7M-18	6-Jun-02	19:04	355879	4830954
Seven-Mile Road Fire	7M-19	19-Jun-02	14:10	356283	4831212

UTM Coordinates of Vegetation Plots (continued)

Name	Plot	Date	Time	Easting	Northing
B & B County Fire	BB-01	19-Jul-02	0:44	356462	4820566
B & B County Fire	BB-02	19-Jul-02	1:05	356930	4821179
B & B County Fire	BB-03	19-Jul-02	1:10	357106	4821285
B & B County Fire	BB-04	22-Jul-02	3:17	357239	4821436
B & B County Fire	BB-05	22-Jul-02	3:26	357437	4821519
B & B County Fire	BB-06	22-Jul-02	3:29	357700	4821822
B & B County Fire	BB-07	16-Jul-02	22:33	358007	4822082
B & B County Fire	BB-08	16-Jul-02	22:38	358248	4822405
B & B County Fire	BB-09	16-Jul-02	22:41	358483	4822734
B & B County Fire	BB-10	16-Jul-02	22:46	358801	4822981
B & B County Fire	BB-11	18-Jul-02	12:50	359044	4823189
B & B County Fire	BB-12	18-Jul-02	1:40	359385	4823461
B & B County Fire	BB-13	23-Jul-02	2:27	359611	4823800
B & B County Fire	BB-14	29-Jul-02	4:54	359715	4823924
B & B County Fire	BB-15	29-Jul-02	4:48	359866	4824107
B & B County Fire	BB-16	24-Jul-02	23:10	360037	4824217
B & B County Fire	BB-17	24-Jul-02	12:38	360165	4824371
B & B County Fire	BB-19	23-Jul-02	3:01	360886	4825067
B & B County Fire	BB-20	24-Jul-02	2:27	360868	4825562
Fire Station Fire	FS-01	11-Jun-02	14:38	344086	4828626
Fire Station Fire	FS-02	7-Jun-02	2:54	344237	4828896
Fire Station Fire	FS-03	7-Jun-02	2:32	344253	4828951
Fire Station Fire	FS-04	7-Jun-02	2:33	344378	4828899
Fire Station Fire	FS-05	7-Jun-02	2:34	344348	4828999
Fire Station Fire	FS-06	7-Jun-02	2:35	344411	4829007
Fire Station Fire	FS-07	11-Jun-02	21:28	344398	4829095
Fire Station Fire	FS-09	7-Jun-02	2:38	344530	4829149
Fire Station Fire	FS-10	2-Aug-02	15:40	344566	4829183
Fire Station Fire	FS-11	2-Aug-02	15:39	344628	4829192
Fire Station Fire	FS-13	2-Aug-02	15:37	344667	4829294
Fire Station Fire	FS-14	2-Aug-02	15:37	344848	4829544
Tractor Flats Fire	TF-01	16-Jul-02	2:06	376326	4833382
Tractor Flats Fire	TF-02	26-Jun-02	21:19	373550	4832163
Tractor Flats Fire	TF-03	26-Jun-02	21:16	373729	4832266

UTM Coordinates of Vegetation Plots (continued)

Name	Plot	Date	Time	Easting	Northing
Tractor Flats Fire	TF-04	26-Jun-02	21:10	373828	4832452
Tractor Flats Fire	TF-05	26-Jun-02	21:06	374031	4832534
Tractor Flats Fire	TF-06	2-Aug-02	13:22	374267	4832863
Tractor Flats Fire	TF-07	2-Aug-02	13:23	374601	4833097
Tractor Flats Fire	TF-08	21-Jun-02	1:01	374862	4833401
Tractor Flats Fire	TF-09	26-Jun-02	20:52	375189	4833670
Tractor Flats Fire	TF-10	21-Jun-02	13:16	375471	4833961
Tractor Flats Fire	TF-11	21-Jun-02	0:33	375768	4834226
Tractor Flats Fire	TF-12	2-Aug-02	13:26	375971	4834591
Tractor Flats Fire	TF-13	21-Jun-02	0:35	376187	4834655
Tractor Flats Fire	TF-14	21-Jun-02	0:36	376310	4834820
Tractor Flats Fire	TF-15	21-Jun-02	0:37	376407	4835005
Tractor Flats Fire	TF-16	21-Jun-02	0:38	376751	4835250

APPENDIX II

INEEL Taxon List (adapted from list provided by Dr. Jay E. Anderson)

species code	7 letter species code
species name	30 letter species name
gf	1 digit growth form code
orig	1 digit historical origin code
fam	3 digit family code
taxon	3 digit taxon code

	growth form	taxon #'s
trees	1	100 - 199
shrubs	2	200 - 299
graminoids	3	300 - 399
forbs	4	400 - 549
succulents	5	550 - 599
annuals	6	600 - 699
biennials	7	700 - 799
unknowns	8	800 - 899
ferns	9	900 - 999

	origin codes
native	1
exotic	2

Family codes

Scientific family name	Common family name	Family code
Aceraceae	Maple	101
Aizoaceae	Carpetweed	148
Anacardiaceae	Sumac	102
Asclepiadaceae	Milkweed	103
Betulaceae	Birch	104
Boraginaceae	Borage	105
Capparidaceae	Caper	106
Caprifoliaceae	Honeysuckle	107
Caryophyllaceae	Pink	108
Cactaceae	Cactus	109
Chenopodiaceae	Goosefoot	110
Compositae	Composite	111
Convolvulaceae	Morning Glory	112
Cornaceae	Dogwood	113
Crassulaceae	Stonecrop	114
Cruciferae	Mustard	115
Cupressaceae	Cypress	116
Cyperaceae	Sedge	117
Euphorbiaceae	Spurge	118
Gentianaceae	Gentian	119
Gramineae	Grass	120
Grosulariaceae	Currant or Gooseberry	121
Hydrophyllaceae	Waterleaf	122
Labiatae	Mint	123
Leguminosae	Pea Fmaily	124
Liliaceae	Lily	125
Loasaceae	Blazing-star	126
Malvaceae	Mallow	127
Onagraceae	Evening-primrose	128
Orobanchaceae	Broomrape	129
Pinaceae	Pine	130
Plantaginaceae	Plantain	131
Polemoniaceae	Phlox	132
Polygonaceae	Buckwheat	133
Polypodiaceae	Common Fern	134
Ranunculaceae	Buttercup	135

Family codes (continued)

Scientific family name	Common family name	Family code
Rhamnaceae	Buckthorn	136
Rosaceae	Rose	137
Rubiaceae	Madder	138
Salicaceae	Willow	139
Santalaceae	Sandlewood	140
Saxifragaceae	Saxifrage	141
Scrophuylaraceae	Figwort	142
Solanaceae	Nightshade	143
Umbelliferae	Parsley	144
Urticaceae	Nettle	145
Verbenaceae	Verbena	146
Violaceae	Violet	147

Plant species list

Species code	Species name	gf	orig	fam	taxon
BARGROU	Bare ground	0	0	0	10
ROCKS	Rocks	0	0	0	20
DEASHRU	Dead shrub	0	0	0	30
LITTER	Litter	0	0	0	40
MOSS	Moss	0	0	0	50
LICHEN	Lichen	0	0	0	55
GRAVEL	Gravel	0	0	0	60
ANT MND	Ant mound	0	0	0	70
BAREANT	Bare area around ant hill	0	0	0	75
BNSHRUB	Burnt shrub	0	0	0	80
UNK 1	Unknown #1 on this plot	0	0	0	90
UNK 2	Unknown #2 on this plot	0	0	0	91
UNK 3	Unknown #3 on this plot	0	0	0	92
UNK 4	Unknown #4 on this plot	0	0	0	93
UNKTREE	Unknown tree	1	0	0	100
ACEGLAB	*Acer glabrum*	1	1	101	101
JUNOSTE	*Juniperus osteosperma*	1	1	116	102
JUNSP	*Juniperus* species	1	1	116	103
PINCONT	*Pinus contorta*	1	1	130	104
PINFLEX	*Pinus flexilis*	1	1	130	105
POPTREM	*Populus tremuloides*	1	1	139	106
PSEMENZ	*Pseudotsuga menziesii*	1	1	130	107
UNKSHRB	Unknown shrub	2	0	0	200
ARTSP1	*Artemisia* species (shrub)	2	1	111	201
ARTARBU	*Artemisia arbuscula*	2	1	111	202
ARTCANA	*Artemisia cana*	2	1	111	203
ARTNOVA	*Artemisia nova*	2	1	111	204
ARTSPIN	*Artemisia spinescens*	2	1	111	230
ARTTRID	*Artemisia tridentata*	2	1	111	205
ARTTRIT	*Artemisia tridentata tridentata*	2	1	111	228
ARTTRIW	*Artemisia tridentata wyomingensis*	2	1	111	229
ARTTRIP	*Artemisia tripartita*	2	1	111	206
ATRCANE	*Atriplex canescens*	2	1	110	207
ATRCONF	*Atriplex confertifolia*	2	1	110	208
ATRNUTT	*Atriplex nuttallii*	2	1	110	209

Plant species list (continued)

Species code	Species name	gf	orig	fam	taxon
ATRSPIN	*Atriplex spinosa*	2	1	110	210
CEAVELU	*Ceanothus velutinus*	2	1	136	211
CERLANA	*Ceratoides lanata*	2	1	110	212
CHRNAUS	*Chrysothamnus nauseosus*	2	1	111	213
CHRSP	*Chrysothamnus* species	2	1	111	214
CHRVICI	*Chrysothamnus vicsidiflorus*	2	1	111	215
ERIMICR	*Eriogonum microthecum*	2	1	133	216
GUTSARO	*Gutierrezia sarothrae*	2	1	111	217
HAPNANU	*Haplopappus nanus*	2	1	111	218
HAPRESI	*Haplopappus resinosus*	2	1	111	219
HOLDUMO	*Holodiscus dumosus*	2	1	137	220
LEPPUNG	*Leptodactylon pungens*	2	1	132	221
PURTRID	*Purshia tridentata*	2	1	137	222
RIBCERE	*Ribes cereum*	2	1	121	223
SARVERM	*Sarcobatus vermiculatus*	2	1	110	224
SYMOREO	*Symphoricarpos oreophilus*	2	1	107	225
TETCANE	*Tetradymia canescens*	2	1	111	226
TETGLAB	*Tetradymia glabrata*	2	1	111	227
TETSPIN	*Tetradymia spinosa*	2	1	111	227A
UNKGRSS	Unknown grass	3	0	120	300
AGRCRIS	*Agropyron cristatum*	3	2	120	301
AGRDASY	*Agropyron dasytachyum*	3	1	120	302
AGRSMIT	*Agropyron smithii*	3	1	120	303
AGRSP	*Agropyron* species	3	1	120	304
AGRSPIC	*Agropyron spicatum*	3	1	120	305
ARIFEND	*Aristida fendleriana*	3	1	120	306
ARILONG	*Aristida longiseta*	3	1	120	307
CARDOUG	*Carex douglasii*	3	1	117	308
CARSP	*Carex* species	3	1	117	309
DANSP	*Danthonia* species	3	1	120	310
ELYAMBI	*Elymus ambiguus*	3	1	120	311
ELYCINE	*Elymus cinereus*	3	1	120	312
ELYSP	*Elymus* species	3	1	120	313
ELYTRIT	*Elymus triticoides*	3	1	120	314
FESIDAH	*Festuca idahoensis*	3	1	120	315
FESSP	*Festuca* species	3	1	120	316

Plant species list (continued)

Species code	Species name	gf	orig	fam	taxon
HESKING	*Hesperochloa kingii*	3	1	120	317
KOECRIS	*Koeleria cristata*	3	1	120	318
ORYHYME	*Oryzopsis hymenoides*	3	1	120	319
POACOMP	*Poa compressa*	3	1	120	320
POAFEND	*Poa fendleriana*	3	1	120	321
POAJUNC	*Poa juncifolia*	3	1	120	322
POANEVA	*Poa nevadensis*	3	1	120	323
POAPRAT	*Poa pratensis*	3	1	120	324
POASAND	*Poa sandbergii*	3	1	120	325
POASP	*Poa* species	3	1	120	326
POABULB	*Poa bulbosa*	3	2	120	334
POASECU	*Poa secunda*	3	1	120	335
SITHYST	*Sitanion hystrix*	3	1	120	327
SPOCRYP	*Sporobolus cryptandrus*	3	1	120	328
STICOMA	*Stipa comata*	3	1	120	329
STILETT	*Stipa lettermanii*	3	1	120	330
STIOCCI	*Stipa occidentalis*	3	1	120	331
STISP	*Stipa* species	3	1	120	332
STITHUR	*Stipa thurberiana*	3	1	120	333
UNKFORB	Unknown forb	4	0	0	400
ACHMILL	*Achillea millefolium*	4	1	111	401
AGACUSE	*Agastache cusickii*	4	1	123	402
AGAURTI	*Agastache urticifolia*	4	1	123	403
AGOGLAU	*Agoseris glauca*	4	1	111	404
ALLACUM	*Allium acuminatum*	4	1	125	405
ALLSP	*Allium* species	4	1	125	406
ALLTEXT	*Allium textile*	4	1	125	407
ANTDIMO	*Antennaria dimorpha*	4	1	111	408
ANTMICR	*Antennaria microphylla*	4	1	111	409
ANTSP	*Antennaria* species	4	1	111	410
ANTROSE	*Antennaria rosea*	4	1	111	410A
ARAHOLB	*Arabis holboellii*	4	1	115	411
ARALIGN	*Arabis lignifera*	4	1	115	412
ARASP	*Arabis* species	4	1	115	413
AREACUL	*Arenaria aculeata*	4	1	108	414B
ARECONG	*Arenaria congesta*	4	1	108	414

Plant species list (continued)

Species code	Species name	gf	orig	fam	taxon
AREFRAN	*Arenaria franklinii*	4	1	108	415
ARENUTT	*Arenaria nuttallii*	4	1	108	416
ARESP	*Arenaria* species	4	1	108	417
ASCSPEC	*Asclepias speciosa*	4	1	103	418
ASTCERA	*Astragalus ceramicus*	4	1	124	419
ASTCALY	*Astragalus calycosus*	4	1	124	420
ASTCONV	*Astragalus convallarius*	4	1	124	421
ASTCURV	*Astragalus curvicarpus*	4	1	124	422
ASTCUSI	*Astragalus cusickii*	4	1	124	423
ASTFILI	*Astragalus filipes*	4	1	124	424
ASTGEYE	*Astragalus geyeri*	4	1	124	425
ASTLENT	*Astragalus lentiginosus*	4	1	124	426
ASTMISE	*Astragalus miser*	4	1	124	427
ASTPURS	*Astragalus purshii*	4	1	124	428
ASTSP1	*Astragalus* species #1 on this plot	4	1	124	429
ASTSP2	*Astragalus* species #2 on this plot	4	1	124	430
BALSAGI	*Balsamorhiza sagittata*	4	1	111	431
CALBRUN	*Calochortus bruneaunis*	4	1	125	432B
CALELEG	*Calochortus elegans*	4	1	125	432
CALNUTT	*Calochortus nuttallii*	4	1	125	433
CALSP	*Calochortus* species	4	1	125	434
CASANGU	*Castilleja angustifolia*	4	1	142	435
CASCHRO	*Castilleja chromosa*	4	1	142	436
CASLONG	*Castilleja longispica*	4	1	142	437
CASSP	*Castilleja* species	4	1	142	438
COMUMBE	*Comandra umbellata*	4	1	140	439
CREACUM	*Crepis acuminata*	4	1	111	440
CRERUNC	*Crepis runcinata*	4	1	111	441
CRESP	*Crepis* species	4	1	111	442
CRYINTE	*Cryptantha interrupta*	4	1	105	443
CYMTERE	*Cymopterus terebinthinus*	4	1	144	444
DELANDE	*Delphinium andersonii*	4	1	135	445
DELNUTT	*Delphinium nuttalianum*	4	1	135	446
EPIALPI	*Epilobium alpinum*	4	1	128	447
ERIFILI	*Erigeron filifolius*	4	1	111	448
ERIFOLI	*Erigeron foliosus*	4	1	111	449

Plant species list (continued)

Species code	Species name	gf	orig	fam	taxon
ERIHUMI	*Erigeron humilis*	4	1	111	450
ERIPUMI	*Erigeron pumilus*	4	1	111	451
ERIGSP	*Erigeron* species	4	1	111	452
ERICAES	*Eriogonum caespitosum*	4	1	133	453
ERIHERA	*Eriogonum heracleoides*	4	1	133	454
ERIOVAL	*Eriogonum ovalifolium*	4	1	133	455
ERIUMBE	*Eriogonum umbellatum*	4	1	133	456
ERIMANC	*Eriogonum mancum*	4	1	133	456A
FRIPUDE	*Fritillaria pudica*	4	1	125	457
GALASPE	*Galium asperrimum*	4	1	138	458
GALMULT	*Galium multiflorum*	4	1	138	459
GILCONG	*Gilia congesta*	4	1	132	460
HAPACAU	*Haplopappus acaulis*	4	1	111	461
HAPSTE	*Haplopappus stenophyllus*	4	1	111	461B
HYDSP	*Hydrophyllum species*	4	1	122	462
IVAAXIL	*Iva axillaris*	4	1	111	463
KOCAMER	*Kochia americana*	4	1	110	464
KOCSCOP	*Kochia scoparia*	4	1	110	464A
LIGSP	*Ligusticum* species	4	1	144	465
LITSP	*Lithophragma* species	4	1	141	466
LOMDISS	*Lomatium dissectum*	4	1	144	467
LOMFOEN	*Lomatium foeniculaceum*	4	1	144	468
LOMMACR	*Lomatium macrocarpum*	4	1	144	469
LOMSP	*Lomatium* species	4	1	144	470
LOMTRIT	*Lomatium triternatum*	4	1	144	471
LUPARGE	*Lupinus argenteus*	4	1	124	472
LUPSERI	*Lupinus sericeus*	4	1	124	473
LUPSP	*Lupinus* species	4	1	124	474
LYGGRAN	*Lygodesmia grandiflora*	4	1	111	475
LYGJUNC	*Lygodesmia juncea*	4	1	111	476
LYGSP	*Lygodesmia* species	4	1	111	477
LYGSPIN	*Lygodesmia spinosa*	4	1	111	478
MENLAEV	*Mentzelia laevicaulis*	4	1	126	479
MITSP	*Mitella* species	4	1	141	480
OENCAES	*Oenothera caespitosa*	4	1	128	481
OENPALL	*Oenothera pallida*	4	1	128	482

Plant species list (continued)

Species code	Species name	gf	orig	fam	taxon
OROFASC	*Orobanche fasciculata*	4	1	129	483
PENSP	*Penstemon* species	4	1	142	484
PENHUMI	*Penstemon humilis*	4	1	142	484B
PHAHAST	*Phacelia hastata*	4	1	122	485
PHLHOOD	*Phlox hoodii*	4	1	132	486
PHLLONG	*Phlox longifolia*	4	1	132	487
PHLSP1	*Phlox* species #1 on this plot	4	1	132	488
PHLSP2	*Phlox* species #2 on this plot	4	1	132	489
PLAHISP	*Plagiobothrys hispidus*	4	1	105	490
POTGLAN	*Potentilla glandulosa*	4	1	137	491
PSOLANC	*Psoralea lanceolata*	4	1	124	492
PSOSP	*Psoralea* species	4	1	124	493
RIBSP	*Ribes* species	4	1	121	494
SAXSP	*Saxifraga* species	4	1	141	495
SCHLINI	*Schoenocrambe linifolia*	4	1	115	496
SILDOUG	*Silene douglasii*	4	1	108	497
SILSCOU	*Silene scouleri*	4	1	108	498
SILSP	*Silene* species	4	1	108	499
SMISP	*Smilacina* species	4	1	125	500
SPHMUNR	*Sphaeralcea munroana*	4	1	127	501
SPHSP	*Sphaeralcea* species	4	1	127	502
STAVIRI	*Stanleya viridiflora*	4	1	115	503
STETENU	*Stephanomeria tenuifolia*	4	1	111	504
TAROFFI	*Taraxacum officinale*	4	1	111	505
TOWFLOR	*Townsendia florifer*	4	1	111	506
TOWSP	*Townsendia* species	4	1	111	507
VILPURP	*Viola purpurea*	4	1	147	508
ZIGPANI	*Zigadenus paniculatus*	4	1	125	509
ZIGVENE	*Zigadenus venenosus*	4	1	125	510
LOMGRAY	*Lomatium grayi*	4	1	144	511
AREACUL	*Arenaria aculeata*	4	1	108	512
ARASPAR	*Arabis sparsiflora*	4	1	115	513
ASTCERI	*Astragalus cibarius*	4	1	124	514
CALBRUN	*Calochortus bruneaunis*	4	1	125	515
CHOWATS	*Chorizanthe watsonii*	4	1	133	516
CONCANA	*Conyza canadensis*	4	1	111	517

Plant species list (continued)

Species code	Species name	gf	orig	fam	taxon
CYMACAU	*Cymopterus acaulis*	4	1	144	518
ERIMACU	*Eriogonum maculatum*	4	1	133	519
EUPGLYP	*Euphorbia glyptosperma*	4	1	118	521
EUPLAT	*Euphorbia lathyris*	4	1	118	537
HEUPARV	*Heuchera parvifolia*	4	1	140	522
LITRUDE	*Lithospermum ruderale*	4	1	144	523
LUPPUSI	*Lupinus pusillus*	4	1	124	524
OENMINO	*Oenothera minor*	4	1	128	525
OROCORY	*Orobanche corymbosa*	4	1	129	526
PENHUMI	*Penstemon humilis*	4	1	142	527
PENPUMI	*Penstemon pumilus*	4	1	142	528
PHAGLAN	*Phacelia glandulifera*	4	1	122	529
STESPUR	*Stephanomeria spinosa*	4	1	111	530
CASINVE	*Castilleja inverta*	4	1	142	531
CIRSCAR	*Cirsium scariosum*	4	1	111	532
HEDBORE	*Hedysarum boreale*	4	1	124	533
LUPBURK	*Lupinus burkei*	4	1	124	534
MEDSATI	*Medicago sativa*	4	2	124	535
ASTSCOP	*Aster scopulorum*	4	1	111	536
UNKSUCC	Unknown succulent	0	0	0	550
OPUPOLY	*Opuntia polyacantha*	5	1	109	551
SEDSTEN	*Sedum stenopetalum*	5	1	114	552
UNKANNU	Unknown annual	6	0	0	600
ALYDESE	*Alyssum desertorum*	6	2	115	601
AMBACAN	*Ambrosia acanthicarpa*	6	1	111	602
AMSMENZ	*Amsinckia menziesii*	6	1	105	603
BROJAPO	*Bromus japonicus*	6	2	120	604
BROTECT	*Bromus tectorum*	6	2	120	605
CHEALBU	*Chenopodium album*	6	1	110	606
CHELEPT	*Chenopodium leptophyllum*	6	1	110	607
CHEFREM	*Chenopodium fremontii*	6	1	110	608
CHESP	*Chenopodium* species	6	1	110	609
CHOBREV	*Chorizanthe brevicornu*	6	1	133	610
COLNUTT	*Coldenia nuttallii*	6	1	105	611
COLSP	*Collinsia* species	6	1	142	612
COLTENE	*Collomia tenella*	6	1	132	613

Plant species list (continued)

Species code	Species name	gf	orig	fam	taxon
CORRAMO	*Cordylanthus ramosus*	6	1	142	614
CORSP	*Cordylanthus* species	6	1	142	615
CRYAFFI	*Cryptantha affinis*	6	1	105	616
CRYCIRC	*Cryptantha circumscissa*	6	1	105	617
CRYFEND	*Cryptantha fendleri*	6	1	105	618
CRYSCOP	*Cryptantha scoparia*	6	1	105	619
DESPINN	*Descurainia pinnata*	6	1	115	620
DESSOPH	*Descurainia sophia*	6	2	115	621
DESSP	*Descurainia* species	6	0	115	622
DRAVERN	*Draba verna*	6	1	115	623
EPIMINU	*Epilobium minutum*	6	1	128	624
EPIPANI	*Epilobium paniculatum*	6	1	128	625
ERISPAR	*Eriastrum sparsiflorum*	6	1	132	626
ERICERN	*Eriogonum cernuum*	6	1	133	627
FESOCTO	*Festuca octoflora*	6	1	120	628
GALAPAR	*Galium aparine*	6	1	138	629
GALBIFO	*Galium bifolium*	6	1	138	630
GAYDIFF	*Gayophytum diffusum*	6	1	128	631
GAYRACE	*Gayophytum racemosum*	6	1	128	632
GAYRAMO	*Gayophytum ramosissimum*	6	1	128	633
GAYSP	*Gayophytum* species	6	1	128	634
GILLEPT	*Gilia leptomeria*	6	1	132	635
GILMINU	*Gilia minutiflora*	6	1	132	636
GILSINU	*Gilia sinuata*	6	1	132	637
HALGLOM	*Halogeton glomeratus*	6	2	110	638
LACSERR	*Lactuca serriola*	6	2	111	639
LAPREDO	*Lappula redowski*	6	1	105	640
LAPSP	*Lappula* species	6	0	105	641
LANSETO	*Langloisia setosissima*	6	1	132	642
LEPPERF	*Lepidium perfoliatum*	6	2	115	643
LINSEPT	*Linanthus septentrionalis*	6	1	132	644
MENALBI	*Mentzelia albicaulis*	6	1	126	645
MEROBLO	*Mertensia oblongifolia*	4	1	105	670
MICGRAC	*Microsteris gracilis*	6	1	132	646
MOLVERT	*Mollugo verticillata*	6	1	148	647
NAMDENS	*Nama densum*	6	1	122	648

Plant species list (continued)

Species code	Species name	gf	orig	fam	taxon
NAVSP	*Navarretia* species	6	1	132	649
OENBOOT	*Oenothera boothii*	6	1	128	650
OENCONT	*Oenothera contorta*	6	1	128	651
OENSCAP	*Oenothera scapoidea*	6	1	128	652
ORTSP	*Orthocarpus* species	6	1	142	653
PHAHUMI	*Phacelia humilis*	6	1	122	654
PLAPATA	*Plantago patagonica*	6	1	131	655
POLDOUG	*Polygonum douglasii*	6	1	133	656
RANTEST	*Ranunculus testiculatus*	6	1	135	657
SALKALI	*Salsola kali*	6	2	110	658
SISALTI	*Sisymbrium altissimum*	6	2	115	659
ERIWILC	*Eriastrum wilcoxii*	6	1	132	660
COLPARV	*Collinsia parviflora*	6	1	142	661
CRYWATS	*Cryptantha watsonii*	6	1	105	662
GYMNUDI	*Gymnosteris nudicaulis*	6	1	132	663
MALAFRI	*Malcolmia africana*	6	2	115	664
STEEXIG	*Stephanomeria exigua*	4	1	111	666
OXYDEND	*Oxytheca dendroides*	6	1	133	667
GILTENE	*Gilia tenerrima*	6	1	132	668
AMBART	*Ambrosia artemisiifolia*	6	1	111	669
UNKBIEN	Unknown biennial	7	0	0	700
CHADOUG	*Chaenactis douglasii*	7	1	111	701
CIRUTAH	*Cirsium utahense*	7	1	111	702
CIRVULG	*Cirsium vulgare*	7	2	111	703
ERIDIVE	*Erigeron divergens*	7	1	111	704
MACCANE	*Machaeranthera canescens*	7	1	111	705
PHAHETE	*Phacelia heterophylla*	7	1	122	706
THELACI	*Thelypodium laciniatum*	7	1	115	707
TRADUBI	*Tragopogon dubius*	7	2	111	708
OENANDI	*Oenothera andina*	7	1	128	709
AGOSP	*Agoseris* species	8	0	111	800
ATRSP	*Atriplex* species	8	0	0	801
ARTSP2	*Artemisia* species (forb)	8	1	111	802
ATRSP	*Atriplex* species (unk growth form)	8	0	110	803
ASTSP	*Asteraceae* species	8	0	111	829
CHASP	*Chaenactis* species	8	1	111	804

Plant species list (continued)

Species code	Species name	gf	orig	fam	taxon
CIRSP	*Cirsium* species	8	0	111	805
CRUSP	Unknown Cruciferae	8	0	115	806
CRYSP	*Cryptantha* species	8	1	105	807
EPISP	*Epilobium* species	8	0	128	808
ERIOSP	*Eriogonum* species	8	1	133	809
GILSP	*Gilia* species	8	1	132	810
HAPSP	*Haplopappus* species	8	1	111	811
LACSP	*Lactuca* species	8	0	111	812
LATSP	*Lathyrus* species	8	0	124	813
MENSP	*Mentzelia* species	8	1	126	814
OENSP	*Oenothera* species	8	1	128	815
PHASP	*Phacelia* species	8	1	122	816
POLSP	*Polygonum* species	8	1	133	818
RUMSP	*Rumex* species	8	0	133	819
SAPSP	Unknown saprophyte	8	0	0	820
STESP	*Stephanomeria* species	8	0	111	821
SUNSP	Undecipherable record	8	0	0	822
TRISP	*Trifolium* species	8	0	124	823
UNKCOMP	Unknown composite	8	0	111	824
UNKLEGU	Unknown legume	8	0	124	825
VICSP	*Vicia* species	8	0	124	826
VIOSP	*Viola* species	8	0	147	827
UNKFERN	Unknown fern	9	0	0	900
CYSSP	*Cystopteris* species	9	1	134	901
WOODREG	*Woodsia oregana*	9	1	134	902

Lichen species list (1995 survey)

Species code	Species Number	Species name
ASFR	901	*Aspicilia fruticulosa*
ASHI	902	*Aspicilia hispida*
ASPH	903	*Aspicilia ??*
CACO	904	*Catapyrenium congestum*
CAJU	905	
CALO	906	
CASQ	907	*Catapyrenium squammosum*
CATE	908	
CATO	909	*Calaplaca toninia*
FUDE	910	*Fulgensia ??*
GUNI	911	
PSCE	912	*Psora cerebriformis*
PSTU	913	*Psora tuckermani*