IMPLICATIONS OF CLIMATE VARIABILITY ON LARGE FIRES ACROSS SPATIOTEMPORAL SCALES IN SAGEBRUSH STEPPE

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ABSTRACT

The occurrence of large fires in the western United States raises questions about the effect of climate change on fire regimes in the past and future. Sagebrush steppe has long been exposed to agriculture, excessive grazing, and invasive species. This endangered ecosystem is facing a new threat of increasingly large wildfires and climate change. The objectives of this study were to reconstruct the fire history for sagebrush steppe ecosystems across three spatial scales of sagebrush-dominated steppe: a. Idaho National Laboratory (INL), b. Snake River Plain (SRP) to include the INL and c. portions of the Northern Basin and Range to include the SRP and the INL. This study used geographic information systems (GIS) to correlate size and occurrence of fires over 5,000 ha with topography, vegetation, and climatic variables. Large fires increased between 1960 – 2003 both in size and number, and increasingly formed a greater proportion of all wildfires over the time period studied. The influence of climate, topography, and vegetation on fire occurrence and size can vary depending on the spatial and temporal scales over which information is collected and analyzed. At the broadest spatial scale, the size of large fires was positively correlated with average annual maximum temperature during the year of the fire event (r^2 = 0.114, P = 0.038. Fire occurrence and average yearly precipitation one year previous to the large fire event were also correlates (r^2 = 0.054, P = 0.031). There was also some correlation with topographical aspect for occurrence of large fires (r^2 = 0.039, P = 0.009 and for total fires (r^2 = 0.041, P = 0.001). From 1960 to 2003 the area was subject to an increase in maximum temperature and a decrease in precipitation. Increases in large fire occurrence and size are attributed to increase in air temperature and exotic grasses. My results and the projected trend toward warmer, drier growing seasons and summers suggest that sagebrush steppe systems are likely to continue to experience an increase in large fires in the future.

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Big sagebrush (*Artemisia tridentata* Nutt.) is the most widespread sagebrush subspecies in the Intermountain region. Almost 99% of the basin big sagebrush [*Artemisia tridentata* Nutt. ssp. *tridentata* (Beetle & Young) Welsh] grassland communities in the Snake River Plain have been converted to agriculture because of their deep and well-drained soils (Noss, LaRoe & Scott 1995). Over 50% of the native sagebrush steppe within the Great Basin has been converted to annual grasslands through disturbances of both natural and anthropogenic origins, such as the invasion and dominance of cheatgrass (*Bromus tectorum* L.) (West 2000).

Fire played an important role in the evolution of many plant species that comprise sagebrush steppe communities. There are competing ideas about how often fire historically burned in sagebrush steppe ecosystems, from a 10-40 year fire interval (Winward 1991) to more than 100 years (Whisenant 1990; Welch and Criddle 2003), and there is little understanding of the pattern produced by natural fires in sagebrush landscapes. The introduction and increasing dominance of cheatgrass has changed the seasonal occurrence and increased the frequency and size of wildfires in these ecosystems, altering successional patterns (Billings 1994; Peters and Bunting 1994; Whisenant 1990).

Wildfires are a serious threat over much of the U.S. They threaten life and property, particularly when they move from rangeland into developed areas, and they may be exacerbated by climate change (Westerling *et al*. 2006). Large fires that lead to extensive loss of sagebrush cover may have negative effects on sagebrush obligate species such as sage sparrow (*Amphispiza belli)*, sage thrasher (*Oreoscoptes montanus)*, sage grouse (*Centrocercus urophasianus),* ground squirrel (*Spermophilus townsendii*), and pygmy rabbit (*Brachylagus idahoensis*) (Knick *et al.* 2003), and more specialized species, and may delay sagebrush recovery after fire (Longland and Bateman 2002). Large, frequent fires can cause extinction of sage grouse populations (Pedersen *et al*. 2003), and small mammals often cannot survive drastic changes in habitat conditions resulting from extensive stand-replacing fires (McGee 1982). Increasing temperature and decreasing precipitation have intensified these ecological changes in arid and semi-arid regions, resulting in the sagebrush habitat being recognized as one of North America's most "at-risk ecosystems" (Noss 1995).

Encroachment of native conifers such as juniper (*Juniperus* spp. L.) on more mesic sagebrush steppe lands may alter fire regimes in these regions. Changes in wildfire size and severity and in the type and patterns of precipitation in burned areas may have significant effects on the ecological and hydrological response of large areas of the landscape (Omi 2005).

Studies of fire history and ecology are vital to understanding and forecasting the impacts of climate change on sagebrush steppe ecosystems. An improved understanding and the ability to forecast future impacts can serve as the scientific foundation upon which fire and land management decisions can be based. Information on fire regimes and the impacts of altered fire regimes on sagebrush steppe can be used to develop and support effective prescribed fire programs, and can be used to increase public understanding of the role and importance of fire in these ecosystems.

The challenge is to separate the natural climatic variability, especially extended droughts that have always existed in sagebrush steppe, from anthropogenic climate change, to adapt management strategies to adjust to the changing environment. This is particularly relevant as climate regimes that extend fire seasons lead to larger fires, more frequent fire and more area burned (Hessl *et al.* 2004). Extreme weather events or anomalies such as extended drought may be of even greater consequence than average weather conditions (Moritz 2004).

There is increasing evidence that the Earth is currently warming (Mann *et al.* 1999). The Intergovernmental Panel on Climate Change (IPCC 2007) deduced that there is a strong probability of climate change on global and regional scales, and considerable warming of the earth's surface will accompany increasing concentrations of greenhouse gases in the atmosphere. An increase in temperature could result in earlier and longer fire seasons (Flannigan *et al*. 2000; Gitay *et al*. 2001; Westerling *et al.* 2006). The occurrence of large fires in the western United States raises questions about the effect of climate change on fire regimes in the past and future. Many sagebrush steppe communities are currently fragmented by cheatgrass, a highly competitive introduced annual and occasionally, biennial grass native to Eurasia and the Mediterranean (Knick and Rotenberry 1997). Prior to the introduction of annuals, insufficient fuels may have limited fire spread in big sagebrush communities except under more severe weather conditions. Annuals have increased fuel loads and continuity allowing fires to spread easily. The spread of fire is typically limited by patchiness in the cover type and is primarily a function of initiation occurrences. A relatively flat, homogenous terrain may allow for propagation of fire spread across the entire landscape with a single ignition if the probability of spread is sufficiently high (Turner and Romme 1994), thereby reducing the cover of perennial species.

With the increase of invasive annuals and anthropogenic disturbances, and climate change, the frequencies of wildfires in sagebrush-steppe regions have been on the rise (Whisenant 1990; Peters and Bunting 1994). Big sagebrush cannot successfully establish and reproduce in a rangeland dominated by annuals with a fire frequency as high as five years, and as a result, recurring fires have eliminated big sagebrush from extensive areas in the Great Basin. Increasing fire frequency hinders the re-establishment of sagebrush post-fire, allowing for the competitive cheatgrass species to become established. These fire-prone, cheatgrass-dominated lands form greater connectivity (Whisenant 1990), in which fires tend to be larger than those on sagebrush-dominated lands. The finer fuels of sagebrush steppe are unable to retain moisture as long as larger fuels due to the differences in particle size (Schmoldt 2000), which increase fire-consumption potential and annual grasses become dormant more rapidly than perennial grasses, resulting in a longer period of cured grass fuel each summer. Disturbances may accelerate cheatgrass invasion but it is not required for cheatgrass establishment. Cheatgrass initially established in the Intermountain area of the western U.S. with the introduction of agriculture and contaminated grain seed, which dramatically changed the ratio of herbaceous understory to woody overstory species with the removal of native bunchgrasses. Cheatgrass may dominate on big sagebrush sites after removal of the shrub overstory as long as there is a seed source and a seed bank (Young and Evans 1973). Native perennial plants are typically incapable of preventing cheatgrass dominance on big sagebrush sites (Young and Evans 1973).

Climate models project the globally averaged surface temperature to increase by 1.1 to 5.4ºC during the next century. Precipitation is likely to be altered, although the magnitude and direction of this change are uncertain (Romme and Turner 1991). An amplification of extreme climatic events is projected to increase. Phase transitions of the Pacific Decadal Oscillation (PDO) and Atlantic Multidecadal Oscillation (AMO) or of teleconnected climate cycles (McCabe *et al.* 2004; Hessburg *et al.* 2005) may have an important influence on fire occurrence at the regional scale. Increasing CO_2 levels may stimulate cheatgrass biomass (Ziska *et al*. 2005) and cheatgrass is expected to become more coarse (e.g., lignin content will increase) in the future, reducing the time that it is palatable to livestock and wildlife and causing fuel loads to accumulate as much as six-fold due to reduced decomposition rates (McIver *et al*. 2004).

The response of natural systems and human systems to climate change and extreme events is of great ecological, social, and economic concern. Consideration of potential effects of climate change must be incorporated into restoration strategies as management actions applied in the present will have to be applicable in the future to meet resource and social needs. For example, re-establishment of sagebrush in areas burned by wildfires is a high restoration priority. Sagebrush is very sensitive to local climatic conditions, thus it is important that appropriate seed sources be selected for areas selected for seeding projects to maximize the potential that the sagebrush will adapt to survive in an altered climate in the future.

Interannual climate variability is expected to continue to be an important part of future climate. The impact of climate variability and extreme climatic events on fire occurrence and size can vary depending on the spatial and temporal scales over which information is collected and analyzed. Climate effects may be of greater importance at finer spatial scales, or shorter temporal scales, than at longer temporal scales and larger spatial scales. Cross-scale ecological interactions necessitate analyses across multiple scales. Climate, topography, and vegetation vary in importance at different scales, and fire size distributions vary across scales (Simard 1991; Hessl *et al.* 2004). At relatively fine spatial scales, fire ignition and spread are dominated by fuel type, moisture, and continuity of fuels, microclimate, and microtopography (Rothermel 1983). Fine-scale analyses tend to emphasize spatially explicit factors such as characteristics of wildfire as they relate to topography, local weather, and site-specific fuels.

At mid-scales, vegetation, macrotopography, seasonal weather and synoptic climate influence fire occurrence and behavior (Nash and Johnson 1996; Westerling and Swetnam 2003; Schoennagel *et al.* 2004; Mermoz *et al.* 2005; Crimmins 2006). Analyses at mid-scale may demonstrate that the conditions in a specific area do not fit generalized broad-scale patterns (Barbour *et al.* 2005). Decadal to millennial variation in climate and vegetation types become important at the broadest scales (Lynch *et al.* 2003; Hessburg *et al.* 2005). Analysis at the broad scale may provide a regional understanding of fire regimes, fuel patterns, climate variability or change, and anthropogenic disturbance that can be used for broad-scale prioritization. Failure to look across multiple scales also complicates local policy implementation developed from broad-scale analyses of climate and fire, and vice versa, because perception is relative and depends on context. Ecological processes examined at a particular scale provide information and understanding that may not be discernable at coarser or finer scales (Allen and Hoekstra 1992). Policymakers, managers, and the public can benefit from multiscale analyses to improve their understanding of the long-term dynamics of climate on disturbances such as wildfire.

The climate portion of the research focuses on understanding the relationship between seasonal climate patterns and fire potential so that climate analogs can help in forecasting large fire events (i.e., a wet winter that increases biomass followed by a dry spring and summer results in higher potential for fire). Corresponding historic climate data with size, extent, and other characteristics of historic fires allows for an assessment of the relationship between climate and fire. This provides fine-scale information necessary to assess the relative contribution of climate variability and change for strategic planning based on climate forecasts.

The specific objectives of this research were: 1) reconstruct the fire history for sagebrush steppe ecosystems across three spatial scales of sagebrush-dominated steppe: a. Idaho National Laboratory (INL), b. Snake River Plain (SRP) to include the INL, and c. portions of the Northern Basin and Range to include the SRP and INL; 2) examine the links between climate variability and large fire events in sagebrush-steppe vegetation; and 3) examine climate change scenarios to assess how climate change may affect fire frequency and size in sagebrush steppe in the future.

STUDY AREAS

Portions of the ecoregions of the Snake River Plain and Northern Basin and Range (Omernik 1987) were chosen for analysis based on their geography, sagebrush steppe vegetation, and availability of data. Classification of ecoregions was based on the identification of a hierarchy of ecological regions through the analysis of the patterns and the composition of geology, physiography, vegetation, climate, soils, land use, wildlife, and hydrology which affect or reflect differences in ecosystem quality and integrity. An ecoregion is defined as a region of relative homogeneity in ecological systems or in relationships between organisms and their environment (Crowley 1967; Bailey 1976; Omernik 1987).

As an example of fine scale analyses, I selected the Idaho National Laboratory, situated in the Snake River Plain in Idaho **(Figure 1.1)**. The Snake River Plain ecoregion (Omernik 1987) was selected to represent mid-scale analyses. The Snake River Plain and Northern Basin and Range ecoregions were combined to present the broadest-scale analysis. The study areas were selected for analysis based on the presence of sagebrush steppe (West 1983a).

Average annual precipitation ranges from 3.7 to 44 cm (Glenn's Ferry and Ft. Bidwell climate stations, respectively; source: WRCC). Mean temperatures for the same period ranged from 5.2 to 20.1°C (Idaho Falls and Glenn's Ferry climate stations, respectively; source: WRCC).

Idaho National Laboratory

The Idaho National Laboratory is situated along the western edge of the Upper Snake River Plain in southeast Idaho (43° N, 112° W), occupying approximately 2,225 km^2 of the Snake River Plain. The average elevation is approximately 1,500 m (range: 1,460 – 1,620 m) throughout much of the INL, with little overall elevation change. The topography is flat to gently rolling, with lava outcrops.

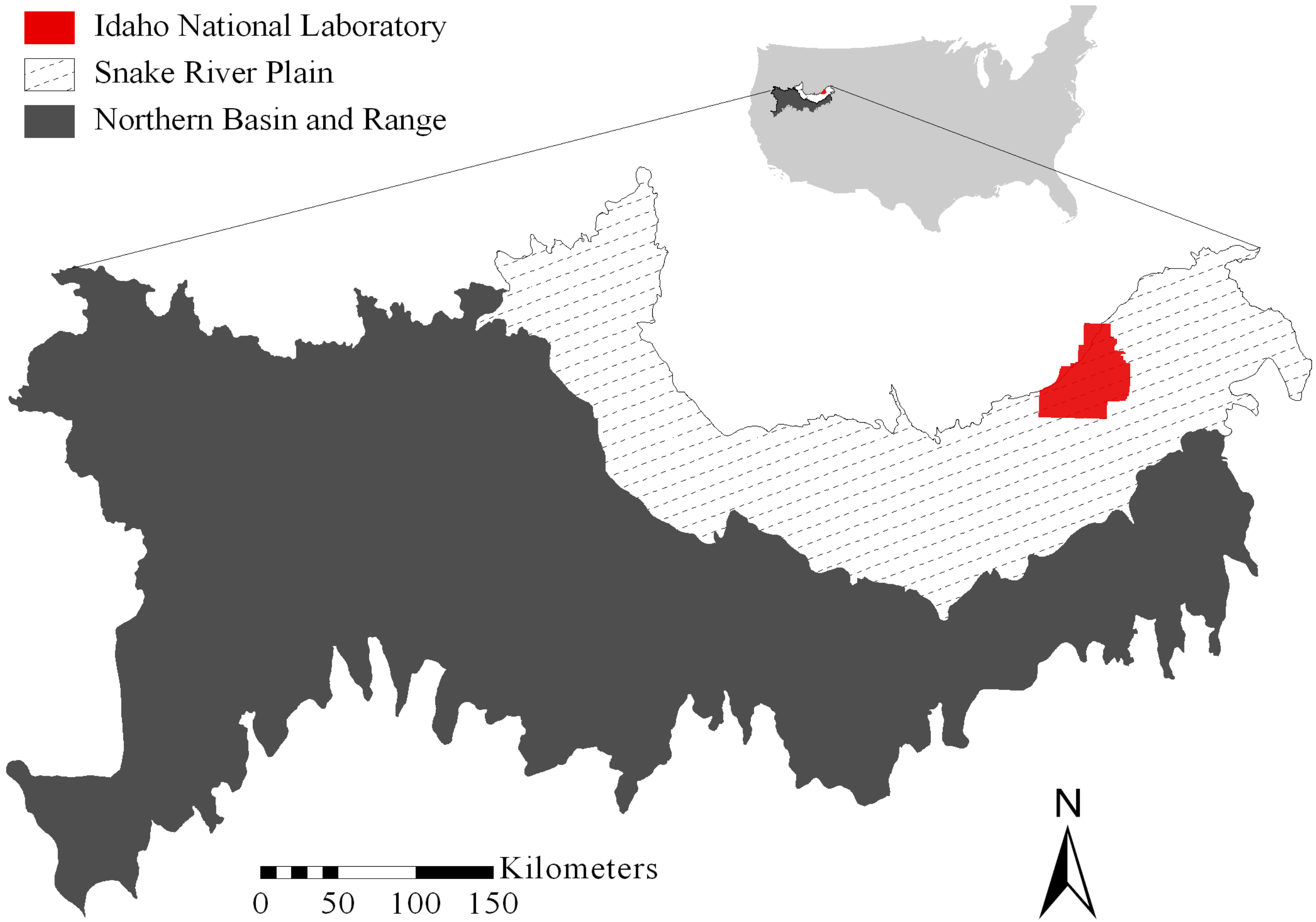

Figure 1.1. The 3 semi-arid regions selected for analysis. The Idaho National Laboratory is nested within the Snake River Plain ecoregion.

Soils and Vegetation

An overstory of Basin and Wyoming big sagebrush **(***Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young) forms the dominant vegetation (Shumar and Anderson 1986). Basin big sagebrush usually grows in deep, well-drained, fertile soils between elevations 610-2,140 m (McArthur and Plummer 1978; McArthur *et al*. 1979; Morris *et al*. 1976; Winward 1980; Winward and Tisdale 1977). Wyoming big sagebrush is found in drier, shallow, poorer rocky soils on between elevations of 1,520-2,150 m (McArthur and Plummber 1978; McArthur *et al*. 1979; Morris *et al*. 1976; Winward and Tisdale 1977). Other dominant shrub species include: rabbitbrush, granite prickly phlox (*Linanthus pungens* (Torr.) J.M. Porter & L.A. Johnson), and broom snakeweed (*Gutierrezia sarothrae* (Pursh) Britt. & Rusby). Perennial grasses in the shrub understory commonly include: bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve), crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.), Indian ricegrass (*Achnatherum hymenoides* (Roemer & J.A. Schultes) Barkworth), needle-and-thread grass (*Hesperostipa comata* (Trin. & Rupr.) Barkworth), Sandberg bluegrass (*Poa secunda* J. Presl), and western wheatgrass (*Pascopyrum smithii* (Rydb.) A. Löve). Perennial forbs comprise a smaller component of the shrub understory.

Most of the soils are aeolian sandy loams and loess, derived from older silicic volcanic flows and Paleozoic sedimentary rocks from the surrounding mountains (McBride *et al*. 1978). The area is underlain by 1,500 m of basalt, the chief 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 INL are dominated by Aridosols 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).

The early grazing history of the INL prior to 1950 is not well documented, but widespread grazing on the Snake River Plain by sheep and cattle since the late 1800s coupled with low perennial grass cover has led to the inference that the INL lands were extensively overgrazed prior to 1950 (Harniss and West 1973; Anderson and Holte 1981; Anderson and Inouye 2001). Grazing impacts were probably compounded by lengthy droughts during the 1930s and 1940s.

<u>Climate</u>

The INL typically experiences hot summers with temperatures exceeding 38°C with high evaporation rates, and cold winters with persistent snow cover from December through March. The mean average temperature and mean annual precipitation are 5.6°C and 22 cm respectively, and is typified by large diurnal and seasonal fluctuations in temperature (West 1983b). A peak of precipitation occurs in the early growing season in the months of May and June. The dominant plant species have similar water-use patterns, as available water is typically depleted by midsummer (Anderson *et al*. 1987). Winds prevail from the southwest (Harniss 1968; Harniss and West 1973).

Snake River Plain

The Snake River Plain ecoregion forms a broad arch covering across the southern part of Idaho extending 600 km eastward from the Oregon border to the Yellowstone Plateau (43° N, 111° W). This portion of the xeric intermontane basin and range area of the western United States is considerably lower and more gently sloping than the surrounding ecoregions. The topography of the area is relatively flat with the exception of the Snake River Canyon and isolated buttes.

<u>Soils and Vegetation</u>

Sagebrush steppe vegetation communities in this region are highly fragmented (Knick and Rotenberry 1995) and predominantly consist of big sagebrush communities in the northeast, grading into winterfat (*Krascheninnikovia lanata* (Pursh) A.D.J. Meeuse & Smit), and shadscale saltbrush (*Atriplex confertifolia* (Torr. & Frém.) S. Wats.) communities in the southwest (Yensen and Smith 1984). Dominant native grasses are bunchgrasses, including bottlebrush squirreltail (*Elymus elymoides* (Raf.) Swezey) and Sandberg bluegrass.

This ecoregion is predominantly sagebrush steppe, with saltbush and greasewood (*Sarcobatus vermiculatus* (Hook.) Torr.) found in alkaline soils (Omernik 1987, Bailey 1995). A large percent of the alluvial valleys bordering the Snake River is used for intensive agriculture. Cattle feedlots and dairy operations are also common in the river plain (Kochert and Pellant 1986). Grazing is restricted to spring and autumn in the northern portion (U.S. Dep. Inter. 1995). Prior to the 1930s, the area was grazed heavily without restriction (Yensen 1982). Except for the scattered barren lava fields, the remainder of the plains and low hills in the ecoregion are now used for cattle grazing. Several fires in the 1980s converted nearly one-half of the native shrub communities to grasslands dominated by cheatgrass, Russian thistle (*Salsola kali* L.), and annual mustards (*Descurainia* spp Webb & Berth., *Sisymbrium* L. spp.) (U.S.D.I. 1995). Fire return intervals have decreased in the Snake River Plain from more than 75 years to 5 to

10 years in some areas resulting from a combination of fire and invasion by exotic annuals (Whisenant 1990) The prevalence of grassland vegetation and exotic annuals allow for widespread fire events (Whisenant 1990). The fire season has also been extended in cheatgrass-invaded regions, and fire events are larger because cheatgrass senesces earlier than native perennials, providing an earlier continuous fuel source (Peters and Bunting 1994).

Climate

The climate of the region is semi-arid with precipitation ranging from 26 cm in the west to 51 cm in the east (Bailey 1995).

Northern Basin and Range

The Northern Basin and Range is a high semi-desert which encompasses a large portion of Nevada, western Utah, and smaller portions of Idaho, California, and Oregon (39° N, 119° W). Elevations range from a low of about 1,200 m to the highest point of the Steens Mountains at roughly 2,930 m.

Soils and Vegetation

The Northern Basin sagebrush-steppe occurs between salt deserts and occupies valley floors and woodlands, conifer forests, and alpine meadows. The climatic regime of the region promotes growth and development of deep-rooted shrubs and C3 herbaceous species (Comstock and Ehleringer 1992). Predominant sagebrush is Wyoming big sagebrush, mountain big sagebrush (*Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle), and little sagebrush (*Artemisia arbuscula* Nutt.). Water availability is limited and a forb and perennial bunchgrass form the typical understory.

Greasewood and saltbush are found in environments of higher salinity. Ranges are generally covered in a mix of species and vegetation types at lower and mid-elevations; As a result of altered fire regimes, juniper encroachment has displaced grasses and sagebrush, especially in the northern portion of the ecoregion.

Throughout the ecoregion, soils are typically rocky and thin, low in organic matter and high in mineral content. Overall, the ecoregion is higher and cooler than the Snake River Plain, and irrigated agriculture in the eastern portion is less prevalent than in the Snake River Plain ecoregion, mostly due to the rugged topography.

Dramatically north-south trending fault-block escarpments, typically comprised of basalt and rhyolite, overlook low-lying lake basins. Higher elevation vegetation consists of upland forests of ponderosa pine (*Pinus ponderosa* P.& C. Lawson) and lodgepole pine (*Pinus contorta* Dougl. ex Loud.), quaking aspen (*Populus tremuloides* Michx.) and curl-leaf mountain-mahogany (*Cercocarpus ledifolius* Nutt.), whereas the intermediate elevations are drier, with western juniper trees (*Juniperus occidentalis* Hook.), big sagebrush, and grasses including Idaho fescue (*Festuca idahoensis* Elmer) and basin wildrye (*Leymus cinereus* (Scribn. & Merr.) A. Löve) most prominent. The low elevations are arid and covered by xeric species such as sagebrush, greasewood, and saltbush. Wetlands systems fed by small streams support tule (*Schoenoplectus acutus* (Muhl. ex Bigelow) A. Löve & D. Löve) and cattail (*Typha latifolia* L.) marshes and their associated biota. Sand dunes often border the marsh systems. Dense coverings of Indian ricegrass (*Achnatherum hymenoides* (Roem. & Schult.) Barkworth, wild rye, and needle–and-thread grass grow in these areas when moisture and temperature regimes are favorable.

Climate

Large differences in topography and low moisture produce relatively large differences in daily and seasonal temperatures. The general aridity of the lower elevations is caused by the rain shadow effect of the Cascade Mountains, which form the western border of the basin. The climate over the region is highly variable, with cold winters and hot summers typical of a continental climate. Most of the precipitation in Northern Basin occurs during the winter and early spring. Interannual precipitation is highly variable and is unevenly distributed across the basin due to elevational gradients and localized rain shadow features (Thompson 1990). Annual precipitation is generally between 180 and 300 mm, although records of 530 mm (1993) and as low as 140 mm (1994) have been recorded.

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CHAPTER 2 – FIRE HISTORIES

INTRODUCTION

Big sagebrush, a native, late-seral shrub species, occupies great expanses of arid lands in the western United States. Big sagebrush often forms the dominant community structure in these areas, commonly described as sagebrush steppe. Maturity varies among subspecies, and occurs from 2 to 15 years of age in big sagebrush, at which time reproduction by seed occurs, maturing in late October and early November. The vast majority of big sagebrush seed produced during fall is absent by spring and very few seeds persist, though seed of some big sagebrush subspecies may persist in the seed bank (Meyer 1994). Seeds can remain in the seed bank for up to 48 months (Tisdale and Hironaka 1981). On-site seed sources are more important as sagebrush seed is not disseminated over large distances, but instead occur generally within 30 m of the parent plant (Colket 2003, Goodwin 1956). Approximately 10% of big sagebrush seed is dispersed more than 9 m of the parent shrub (Goodrich *et al*. 1985). Seedlings appear in early spring soon after snowmelt (Meyer *et al.* 1990). Differences among subspecies in root development, seed germination and dispersal capabilities, the ability to conduct photosynthesis at low temperatures and the production of allelopathic substances (Blaisdell *et al*. 1982; Kelsey 1986a; Kelsey 1986b; Meyer and Monsen 1992) influence the distribution and persistence of big sagebrush. Climatic patterns, elevation gradients, soil characteristics and response to fire are among the factors regulating the distribution of sagebrush subspecies. Each subspecies is considered a topographic late seral dominant (Monsen and Shaw 1994). Among the major subspecies of big sagebrush, basin big sagebrush is considered intermediate in flammability. Mountain big sagebrush

(*Artemisia tridentata* Nutt. ssp. *vaseyana* (Rydb.) Beetle) is most flammable of the big sagebrush species, and Wyoming big sagebrush (*Artemisia tridentata* Nutt. ssp. *wyomingensis* Beetle & Young) is least flammable (Britton and Clark 1985).

Basin big sagebrush

Basin big sagebrush (*Artemisia tridentata* Nutt. ssp. *tridentata*) is the tallest of the big sagebrush species (Tisdale and Hironika 1981) and grows at elevations of 1220 to 2410 m. It is distributed from Montana west to Washington and south to California (Winward 1970; Barker and McKell 1983). Big sagebrush normally occurs on deep, productive, well-drained, and gravelly to fine sandy loams and deep alluvial soils (Welsh *et al*. 1987). Precipitation on basin big sagebrush sites ranges from 250-460 mm annually. The root systems of all subspecies of big sagebrush are well adapted to extract moisture from both shallow and deep portions of the soil profile due to a well developed system of lateral roots near the surface and a deep root system. This makes them highly competitive with associated grasses and forbs (Britton *et al*. 1981). Basin big sagebrush exhibits greater plant height, crown cover, production, and annual leader growth than Wyoming big sagebrush. Basin big sagebrush is the least palatable of the major subspecies of big sagebrush (Owens and Norton 1990), and both mountain big sagebrush and Wyoming big sagebrush are preferred by grazers (Weiss and Verts 1984). Basin big sagebrush grows rapidly and spreads readily from seed. It commonly reaches 40 to 50 years of age, and some plants may exceed 100 years. Slow-growing individuals on unfavorable sites attain the greatest age.

Because of the time needed to produce seed, basin big sagebrush is eliminated as a subspecies by frequent fires (Bunting *et al*. 1987). Basin big sagebrush reinvades a site primarily by off-site seed or seed from plants that survive in unburned patches. The rate of stand recovery depends on the season of burn because season affects the availability of seed, post-fire precipitation patterns, and the prevalence of other regenerating plant species that may cause interference (Britton and Clark 1985; Daubenmire 1975; Zschaechner 1985). Establishment may be delayed until favorable moisture conditions occur (Humphrey 1984), as seedling survival is often dependent on precipitation. Heat exposure may decrease basin big sagebrush seed emergence (Chapin and Winward 1982). Big sagebrush does not resprout after fire (Sheehy and Winward 1981). Basin big sagebrush is a more copious seed producer than Wyoming big sagebrush.

Wyoming big sagebrush

Wyoming big sagebrush grows at elevations of 1525 to 1980 m, and may be found at elevations up to 2,700 m in sagebrush, rabbitbrush, salt desert shrub, juniper and bitterbrush (*Purshia tridentata* DC. ex Poir.) communities (Cronquist 1994; Welsh *et al*. 1987). Wyoming big sagebrush occurs in Idaho, Montana, Wyoming, and Colorado (Barker and McKell 1983), and is the most drought tolerant of the three major big sagebrush subspecies found in the Great Basin (Meyer and Monsen 1992). Wyoming big sagebrush typically grows on shallow, gravelly soil on sites receiving 180 to 310 mm of annual precipitation (Cronquist 1994; Goodrich *et al*. 1999; Monsen and McArthur 1984), in areas which are wetter and cooler than those occupied by Basin big sagebrush (Britton *et al*. 1981). Sites receiving less precipitation are typically dominated by shadscale (*Atriplex confertifolia*) and/or winterfat (*Ceratoides lanata*) (Hironaka *et al.*

1991). Where the ranges of Wyoming and mountain big sagebrush overlap, Wyoming big sagebrush generally occurs where precipitation is less than 300 mm (Bunting *et al*.1987), and tends to grow on the hottest soils relative to the other two subspecies. Wyoming big sagebrush is generally the most palatable of the big sagebrush subspecies. It is a long-lived species, whose community diversity is lower than in other big sagebrush types. A few perennial forb species are usually present in low numbers within Wyoming big sagebrush communities (Bunting *et al*. 1987).

Wyoming big sagebrush reproduces from seed, and establishment occurs mostly from the seed bank (Hironaka *et al.* 1983). Wyoming big sagebrush is a mid- to late-seral species (Sturges *et al.* 1994). Establishment after a stand-replacing fire is variable with site, but typically about 10 years (Sturges *et al.* 1994). Prior to re-establishment of Wyoming big sagebrush, disturbed Wyoming big sagebrush communities are mostly populated with associated grasses. On burns, Wyoming big sagebrush that escapes fire is an important seed source. If the seed bank is destroyed over a large area by repeated fires, Wyoming big sagebrush may take decades to reseed from nearby areas (Tisdale *et al.* 1969). Fire does not stimulate nor does it inhibit germination of Wyoming big sagebrush seed stored in the soil (Chapin and Winward 1982). Drought conditions favor establishment of Wyoming big sagebrush over perennial bunchgrasses (Beetle 1960), though seedling growth is slow compared to growth of other subspecies even with available water and nutrients (Blank *et al.* 1994; Booth *et al.* 1990).

Cheatgrass

Cheatgrass is an exotic invasive monocarpic C_3 plant that out-competes sagebrush and other species for resources, including moisture, in the post-fire environment by quickly recolonizing the post-burn environment, preventing shrub growth. (Billings 1990; D'Antonio & Vitousek 1992; Melgoza *et al*. 1990; Stewart and Hull 1949; Young and Evans 1978). On fragmented areas formerly occupied by shrub and bunchgrasses, the accumulation of cheatgrass tillers during summer creates a dense vegetative cover and higher fuel load, increasing fire susceptibility (Whisenant 1990). In some areas formerly inhabited by sagebrush steppe, fire frequency is decreased from historical levels to every three to six years (Young and Evans 1978; Whisenant 1990; Peters and Bunting 1994).

Excessive grazing and frequent fires can damage biological soil crusts and decrease soil moisture, eradicating many perennial plants, which then encourages cheatgrass establishment, survival, persistence, and dominance (Billings 1994; D'Antonio and Vitousek 1992; Peters and Bunting 1994; Whisenant 1990; Young and Evans 1972). A classic successional pattern after disturbance in the Northern Basin and Range and Snake River Plains commences with Russian-thistle and/or mustard species, followed by cheatgrass dominance within five years (Daubenmire 1970; Evans and Young 1978; Piemeisel *et al*. 1951; Young and Evans 1972). Wyoming and basin big sagebrush are the most xeric of the big sagebrush community types and most susceptible to invasion by cheatgrass, increasing the likelihood of conversion to annual grasslands with increased fire frequency as a result (Bunting *et al*. 1987; Miller and Eddlemen 2000). In one study, cheatgrass root systems showed no reduction in root elongation when in competition with bluebunch wheatgrass seedlings. The shortest cheatgrass roots were found where bluebunch wheatgrass was found in the lowest density, suggesting that intra-specific competition was more heavily felt by cheatgrass than inter-specific competition (Harris 1967).

Cheatgrass reproduces before summer, providing highly flammable fine fuels sufficient to promote fires that can spread rapidly (Bradford and Lauenroth 2006). As quickly as one to five years following the initial fire, the quantity and continuity of fuels in a cheatgrass monoculture is generally adequate to carry another fire (DiTomaso 2000). Under favorable conditions, cheatgrass is a prolific seed producer and the seed usually highly viable, germinating within days. Cheatgrass is dependent on the timing and amount of moisture received for growth and tillering (Harris 1967; Klemmedson and Smith 1964), as moisture availability can affect cheatgrass productivity and thus affect fuel loads on a site. Drought years may reduce the dominance of cheatgrass in both recently burned and unburned areas, decreasing fuel loads and the chance of fire (Knapp 1998). It is worth noting that cheatgrass can also survive periodic drought because viable seeds survive in the soil for up to 5 years (Young *et al.* 1969).

Cheatgrass has several unique adaptations, including fructan metabolism which gives it the ability to metabolize carbohydrates in such a manner that permits growth at fairly low temperatures (Chatterton 1994), which assist in early, rapid growth, enabling it to retain dominance on many sites. Germination potential is present in temperatures varying from 0 to 40°C. In more mesic areas of sagebrush-grass ecosystems, cheatgrass germination occurs in both spring and fall, while in more arid portions, germination occurs in the early spring because it is usually too cold for germination by the time effective moisture is received in the fall. Cheatgrass usually matures by July, and wildfires often occur earlier in the season where cheatgrass dominates, when the immature native perennials are more susceptible to injury and mortality by burning (Young and Evans 1989). Because cheatgrass can grow and deplete soil moisture before native plants surpass dormancy, it gains a competitive advantage in cold, semiarid environments (Harris 1967).

Optimal growing temperatures for cheatgrass range from 10 to 20° C (McCarlie *et al.* 2003), and areas receiving 300 to 560 mm of precipitation that peaks in late winter or early spring tend to promote maximum invasiveness (Pyke and Novak 1994). The success of cheatgrass has been attributed to its phenotypic plasticity, rapid growth of an extensive root system, ability to germinate and establish over a wide range of temperature and moisture conditions (Nasri and Doescher 1995; Rasmuson and Anderson 2002), and ability to adapt to different environments (Young and Evans 1975). The seed bank can contain up to three times as many viable cheatgrass seeds as there are established plants in the community once cheatgrass is established on a site (Young and Clements 2000).

Fire regimes

Fire regimes in sagebrush ecosystems have changed dramatically during the past century (Knick *et al*. 2005). Sagebrush and grassland communities existed in multiple successional stages in the early 1800s as a result of fires of varying frequency and severity (Young *et al*. 1979). Historic heavy grazing allowed for the invasion and establishment of introduced species, in particular, cheatgrass and increased fire frequency. Homogenous annual grasslands have resulted from these practices

(Whisenant 1990; Billings 1994; Peters and Bunting 1994). The invading grass species inhabit openings between native plants in sparsely vegetated systems, creating an uninterrupted path of fine fuel that promotes the spread of fire. Grass populations tend to recover quickly following fire events as compared with the slower recovery of native perennials. Resulting higher fire frequencies allow for a reduction in the density of widely spaced perennials, convert shrublands to annual grasslands. In southern California, chaparral and coastal sage scrub are at risk of being replaced by grasslands, in part due to an increase in fire frequencies, combined with competition from non-native annuals and other anthropogenic forces (Minnich & Dezzani 1998).

Historic fire regimes are variable in big sagebrush steppe/bunchgrass ecosystems, with fire return intervals estimated to have ranged from a decade to more than a century (Arno and Gruell 1983; Burkhardt and Tisdale 1973; Houston 1973; Miller and Rose 1999; Sapsis 1990; Vincent 1992; Young and Evans 1981; Wright *et al*. 1979; Wright and Bailey 1982). Wright *et al*. (1979) inferred that the interval between fires was adequate for non-resprouting big sagebrush to recover, or the extensive areas dominated by sagebrush would likely have been dominated by root-sprouting shrubs such as horsebrush or rabbitbrush. The life history of a species in sagebrush communities is defined by its ability to persist through a fire or to establish seedlings after a fire, and by its longevity or competitive ability after a fire (Noble and Slatyer 1980). Some plant species, respond vigorously to post-fire conditions (Wright *et al.* 1979; Ratzlaff and Anderson 1995), and rabbitbru sh (*Chrysothamnus viscidiflorus Hook*.), horsebrush (*Tetradymia glabrata, T. canescens*) and snakeweed (*Gutierrezia sarothrae* (Pursh) Britt. & Rusby) often dominate these vegetation communities following fire (Paysen *et al.*

2000; Wright and Bailey 1982; Bunting *et al.* 1987; Wambolt and Sherwood1999). Many sagebrush steppe communities evolved with recurring fire due to cold, wet winters and springs and hot, dry summers.

Historically, fires in most sagebrush steppe landscapes were likely more uniform, leaving fewer unburned areas in a mosaic, than is observed in modern prescribed fires (Wrobleski and Kauffman 2003; Whisenant 1990) due to higher fine-fuel amounts, greater fuel continuity, and drier burning conditions. Wyoming big sagebrush has a mean stand age of 32 years, and basin big sagebrush has a mean stand age of 17 years with a more frequent recruitment interval due to higher productivity and higher fuel loads than Wyoming big sagebrush communities (Perryman *et al.* 2001). Wyoming big sagebrush steppe communities historically were characterized by 10 to 70 year interval, patchy fires that produced a mosaic of burned and unburned lands (Britton *et al*. 1981; Bunting *et al.* 1987; Frandsen 1983; Harniss *et al.* 1973; Peek *et al*. 1979; Vincent 1992; West and Hassan 1985; Young and Evans 1981). Fire return intervals in basin big sagebrush are intermediate between mountain big sagebrush (5 to 15 years) and Wyoming big sagebrush (10 to 70 years) (Sapsis 1990). Historically, fire seasons in sagebrush steppe occurred between July and September (Acker 1992; Antos *et al*. 1983; Knapp 1995; Young and Evans 1978), with the most favorable fire conditions occurring in August.

In big sagebrush communities dominated by cheatgrass, fire regimes have been drastically changed (Bunting *et al*. 1987; Hironaka 1991; McArthur *et al*. 1998; Tisdale *et al.* 1969). Wildfire frequency in sagebrush steppe has been increasing from historical levels due to an increase of invasive annuals combined with anthropogenic disturbances and climate change (Knick and Rotenberry 1997; Knick 1999), and are larger, more intense, and faster spreading, extending the fire season by as much as 2 months (Hull and Pechanec 1947). Successive fires become common, with each fire further reducing the surviving shrub cover and native seed bank, thereby inhibiting processes essential to secondary succession (Peters and Bunting 1994; Pickford 1932; Whisenant 1990). Many sagebrush subspecies require a fire-return interval of no less than 20 to 50 years for establishment and persistence (Peters and Bunting 1994). Expansion of cheatgrass has altered fire return intervals on the more xeric or dry interior rangelands from more than 50 years to less than 10 years (Miller and Tausch 2000). The average fire-return interval for cheatgrass-dominated stands has been reduced to 3-5 years from 30-100 years on the Snake River Plain (Whisenant 1990; Peters and Bunting 1994). As burned areas increase as a result of larger fire size and increasing connectivity, the chance for successful re-establishment of sagebrush steppe remnants become ever more unlikely (Knick and Rotenberry 1997) as new recruitment spaces become available (Owens and Norton 1992; Perryman *et al*. 2001; Wambolt and Hoffman 2001). Larger fires decrease chances that seed from sagebrush adjacent to burned areas can immigrate across the larger distance to the burned patches. The increased connectivity of fine fuels in the landscape is therefore more important in the fire regime than the amount of aboveground biomass from fine fuels (Whisenant 1990).

Wildfires transition along a continuous spectrum from small fires of short duration to extremely large, complex fires of long duration. Although not common, separate fires have linked to form one “megafire”. Mega-fires are uniquely complex and resistant to suppression efforts. With few exceptions, they usually occur on drier sites, where the buildup of debris and accumulation of live biomass can fuel high-intensity events. Mega-fires often burn under extreme fire weather conditions and exhibit extreme fire behavior characteristics. Typically, breaks in weather or fuel conditions are the mitigators of these “ultra-catastrophic” fires. They have not been defined in absolute terms such as in size, but rather they are the “headliners” that gain public attention during times of contention when media, overstressed operations, and politics collide. It is only recently that catastrophic mega-fires have begun to increase again following all-out suppression of fires. In the U.S. alone, since the late 1990s, there have been no less than ten extremely large, complex, very destructive wildfires that warrant the label “mega-fire” (Bartlett *et al.* 2007), including the 264,000 hectare 2007 Murphy Complex Fire which began as a combination of six lightning-caused wildfires in south-central Idaho and north-central Nevada. Increases in the occurrences of large fires in sagebrush-steppe may ultimately lead to an increase in the occurrence of mega-fires, especially when climate change is factored in.

Effects of climate change

Climate change may modify the fire regime of the regions, negatively affecting ecosystem responses to climate change, on both a species level (Weber and Flannigan 1997) and on the ecosystem processes level (Ryan 1991, Keane 1995). Potential future changes in climate could alter precipitation patterns, which have a high potential to alter the structure, function and productivity of sagebrush steppe ecosystems (Neilson *et al.* 1989; Brown *et al*. 1997; Ehleringer *et al*. 2001; Bates 2006). Nonlinear changes in succession of vegetation will likely occur in response to climate change as the pattern, severity, and fire season are altered (Flannigan and Van Wagner 1991; Agee 1993; Crutzen and Goldammer 1993; DeBano *et al*. 1998).

Broad-scale climate patterns provide natural mechanisms that could synchronize fire occurrence over regions. The effect that regional and continental climatic gradients have on productivity and accumulation of fuel biomass can serve as a dominant synchronizing agent (Heyerdahl *et al.* 2002; Hallett *et al.* 2003; Weisberg and Swanson 2003). The more synchronous fires are across a larger spatiotemporal scale, the more indicative of a stronger climate signal (Swetnam and Betancourt 1998). Swetnam and Betancourt (1998) suggests that climatic variability can intensify or alternately, mute anthropogenic effects on fire regimes. Consecutive anomalously wet years (which increases the fuel loading) followed by drought may increase the likelihood of fire. Interannual variability in annual area burned by large fires in the western U.S. has been distinctly higher over the last 2-3 decades than at any time during the last century for which data is available (Brown *et al.* 2004). There is an urgent need for a better understanding of the influence of climate on fire regimes to enhance management efforts for fuels, fire risk, and impacts on ecosystem structure and function.

Specific research questions at the initiation of this research project were: How does fire occurrence and fire size class change with the spatial scale of observation and how will climate change affect fire frequency and size characteristics within sagebrush steppe ecosystems? My hypotheses were: 1) Across size classes, fires are clustered clustered or non randomly distributed across the landscape and spatial scales; 2) Climate change will likely influence the occurrence of large fire events by increasing disturbance regime and the abundance of invasive species. I predicted that a) Fires occur in areas with high biomass of annuals and fuel continuity; b) Fire occurrence would be more frequent with increased drought stress and evapotranspiration due to an increase in summer temperatures; c) An increase in winter or spring precipitation will increase fire occurrence and size the following year due to an increase in available fuels; and d) Large fire (5 000+ ha) occurrence will increase as non-native annuals dominate previous sagebrush-dominated lands, increasing fuel loads.

METHODS

Fire history investigations were conducted using fire records from 1960 to 2003 for the Idaho National Laboratory, the Snake River Plain ecoregion and the Northern Basin and Range ecoregion. These records were synthesized to improve our knowledge on fire regimes, with a specific focus on the variability of fire size and frequency of occurrence. This spatial reconstruction of fire regimes were mapped in a GIS and compared with base vegetation classes for better understanding of how fire history is represented in each particular sagebrush steppe vegetation type across the landscape, and to allow the spatial reconstruction of fire regimes across the study areas. Comparison of temporal fire records to independent climate indices were done to determine if they are significantly correlated.

Fire

Fire records (1960 - 2003) for the Idaho National Laboratory, Snake River Plain ecoregion and Northern Basin and Range ecoregion were obtained from the U.S. Geological Survey's Snake River Field Station SAGEMAP database (The Great Western

Fire Map). The Great Western Fire Map attribute data was gathered from the Internet, public databases, and personal contact with agency personnel throughout the Western U.S. Each data set was attributed with fields for record number, source file, and year of fire, and the record number and source file fields were concatenated. Because various resource management entities map fires somewhat independently (especially in more recent years), there existed overlap of some data in the coverage. Duplicates were removed prior to analyses.

Vegetation

The first vegetation dataset I obtained was a current vegetation coverage in ArcInfo GRID format from the U.S. Geological Survey's SAGESTITCH map (Comer *et al.* 2002) downloaded from the SAGEMAP website (http://sagemap.wr.usgs.gov/). A regional dataset was produced by the USGS using decision tree classifier and other techniques to model landcover. Multi-season satellite imagery (Landsat ETM+, 1999-2003) and digital elevation model (DEM) derived datasets (i.e., elevation, landform, aspect) were utilized in the development of the coverage to derive rule sets for the various landcover classes. The mapping area models were subsequently combined with a Regional Gap Landcover Dataset to create the final seamless eight-state regional landcover map. Mapping area models were then mosaicked to create separate mapping zones for the Northern Basin and Range and Snake River Plain. The final map contains 126 Landcover classes (103 NatureServe Ecological Systems, 7 NLCD and 16 non-native vegetation classes) and has a minimum mapping unit (MMU) of approximately 2.5 ha.

The second vegetation dataset I obtained was of vegetation types in ArcInfo GRID format that uses biophysical settings (BpS) vegetation models developed for the

LANDFIRE project (Rollins and Frame 2006). The LANDFIRE project is an effort to map vegetation, fuels, and fire regime data across the United States. LANDFIRE BpS models were formerly referred to as Potential Natural Vegetation Groups (PNVG; Küchler 1964) and are derived from NatureServe (a non-profit conservation organization that provides scientific information and tools) Ecological System Classification models (Comer *et al*. 2003). LANDFIRE BpS classifications describe physiographic and ecological landscape characteristics, integrating both environmental site potentials (elevation, soil types, and climate) and disturbance regimes to model historical vegetation distributions. The BpS represents the vegetation that may have dominated the landscape prior to Euro-American settlement and is based on both the current biophysical environment and an approximation of the historical disturbance regime. The advantages of using biophysical and current vegetation spatial data layers are that many land managers are familiar with BpS vegetation classifications, and this methodology is applicable to multiple spatial scales of analysis (Schmidt *et al.* 2002). Fires in big sagebrush are typically stand-replacing; the BpS vegetation coverage provides a reference for vegetation types that existed before modern day variations. I grouped vegetation species into general cover types for the analyses. This procedure reduced the original 248 BpS vegetation types to 7 and reduced the current 49 vegetation types to 10.

For each spatial scale, the percentage of large fires (size class “D” or >5 000 hectares) within each vegetation cover type and the percentage of the total number of large fires within each vegetation cover-type were compared with the land-area occupied by the vegetation cover type within the region. I calculated the percent of size class “D” fires by current and BpS vegetation cover type for all regions. I also calculated percentage of total land area the predominant current and BpS cover types comprised for each region. Cover types which had less than 1% of the vegetation in an ecoregion and less than 1% of the area burned by large fires were omitted from the analysis.

Topography

A 1:24,000-scale, 7.5 minute DEM was obtained from the U.S.G.S. National Elevation Dataset (NED). The 2 arc-second dataset approximates a 60-m resolution. Topographic variables for size class "D" fires were obtained using the Zonal Statistics function in Spatial Analyst and selecting unique fire ID's from the attribute table.

I assessed local variation in fire occurrence by correlating fire frequency with topographic variables averaged across each fire, and examining the spatial patterns of fire size with respect to topography. The large fires were correlated with the elevation, aspect, and slope of that location using a nonparametric test (Spearman rank correlation).

Fire Patterns

Fire data was separated into four selected groupings, used as proxies for very small, small, medium, and large fires based on size class: (1) A-class fires that burned areas up to 50 ha); (2) B-class fires that burned areas in the range of 51 and 500 ha; (3) C-class fires that burned in the range of 501 and 5 000 ha; and (4) D-class fires that burned areas greater than 5 000 ha.

RESULTS AND DISCUSSION

Broad Scale

Topography

Topographical variables such as slope and elevation were not significant drivers of large fire size or occurrence (**Table 2.1)**. Aspect was positively correlated with fire occurrence for both large fires (r^2 = 0.031, P = 0.009) and for total fires (r^2 = 0.031, P = 0.001), though the relationship was weak. South-facing slopes receive more direct sunlight and more snow is lost to evaporation, leading to less soil moisture than locations with north and east aspects. South-facing slopes may also have mostly sagebrush, few perennial grasses and forbs and larger amounts of cheatgrass. Much of the precipitation in the study region typically occurs as snow during the winter and spring months, and snowfall is often redistributed by strong winds from the southwest or west. Winds may blow large amounts of snow off south and west facing slopes on to adjacent north and east facing slopes. Elevation ranged from 183 – 1006 meters in the study region (**Figure 2.1)**. The majority of area burned by large fires in the study area occurred at an elevation range of 450 and 760 m (74.73%). The majority of area burned by large fires (97.92%) occurred on slopes of 0 – 25%, with the remainder occurring on slopes between 25 and 50% (**Figures 2.2 – 2.3)**.

Vegetation

Current vegetation was grouped into 11 cover types present in the study area **(Figures 2.4– 2.6**). Ten predominant current vegetation types were recognized across the study region (**Table 2.2)**, with five types most prevalent in areas burned by size class D fires (sagebrush steppe species combined). Sagebrush steppe was the dominant cover

Table 2.1. Spearman rank correlations between large fires (>5 000 ha) and all fires, and slope, elevation and aspect in the 3 selected study areas.

SCALE: **BROAD**

	Large fires		All fires	
NBR/SRP/INL	r^2	**P**	r^2	**P**
Slope	0.010	0.407	0.010	0.127
Elevation	- 0.014	0.245	- 0.014	0.207
Aspect	**0.031**	**0.009***	**0.031**	**0.001***

* Significant at alpha = 0.05

Table 2.2. Predominant vegetation cover types (> 1%) in the study area to include the Idaho National Laboratory and the Snake River Plain and Northern Basin and Range Ecoregions.

SCALE: **BROAD**

Biophysical Setting (BpS)	Total Land Area (%)
Desert Scrub	18.5
Sagebrush Steppe	8.1
Sparsely Vegetated Systems	3.5
Woodlands	4.9

Current Vegetation	Total Land Area (%)
Sagebrush Steppe	59.3
Agriculture	12.8
Bunchgrass	6.7
Exotic	2.8
Forests/ Woodlands	3.9
Barren/ Rock/ Lava	1.8
Salt Desert Scrub	1.5
Bitterbrush	1.4
Black Greasewood	1.3
Mountain Shrub	1.2

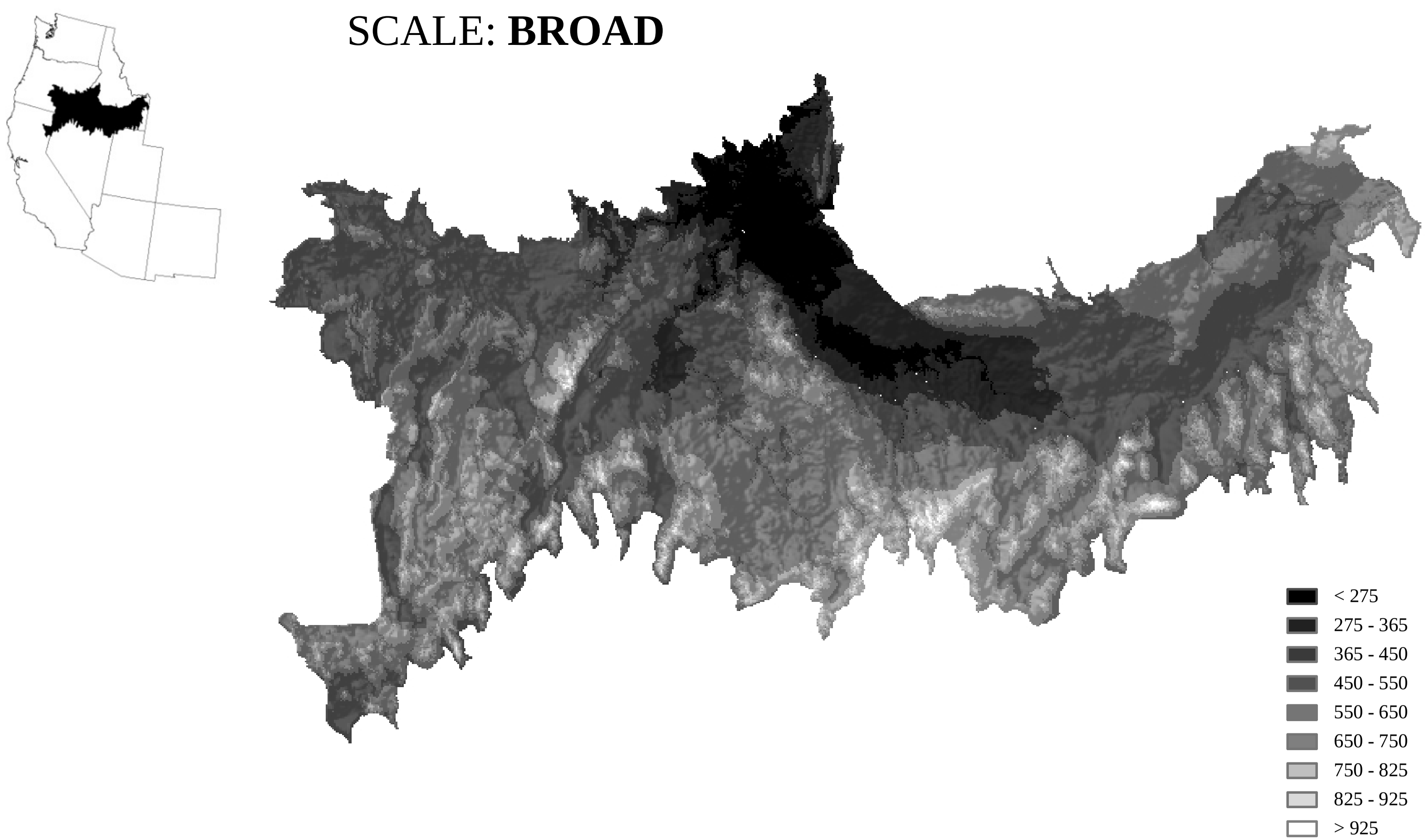

Figure 2.1. Elevation (meters) and hillshade in the selected study area to include the Snake River Plain and Basin and Range Ecoregions, and the Idaho National Laboratory.

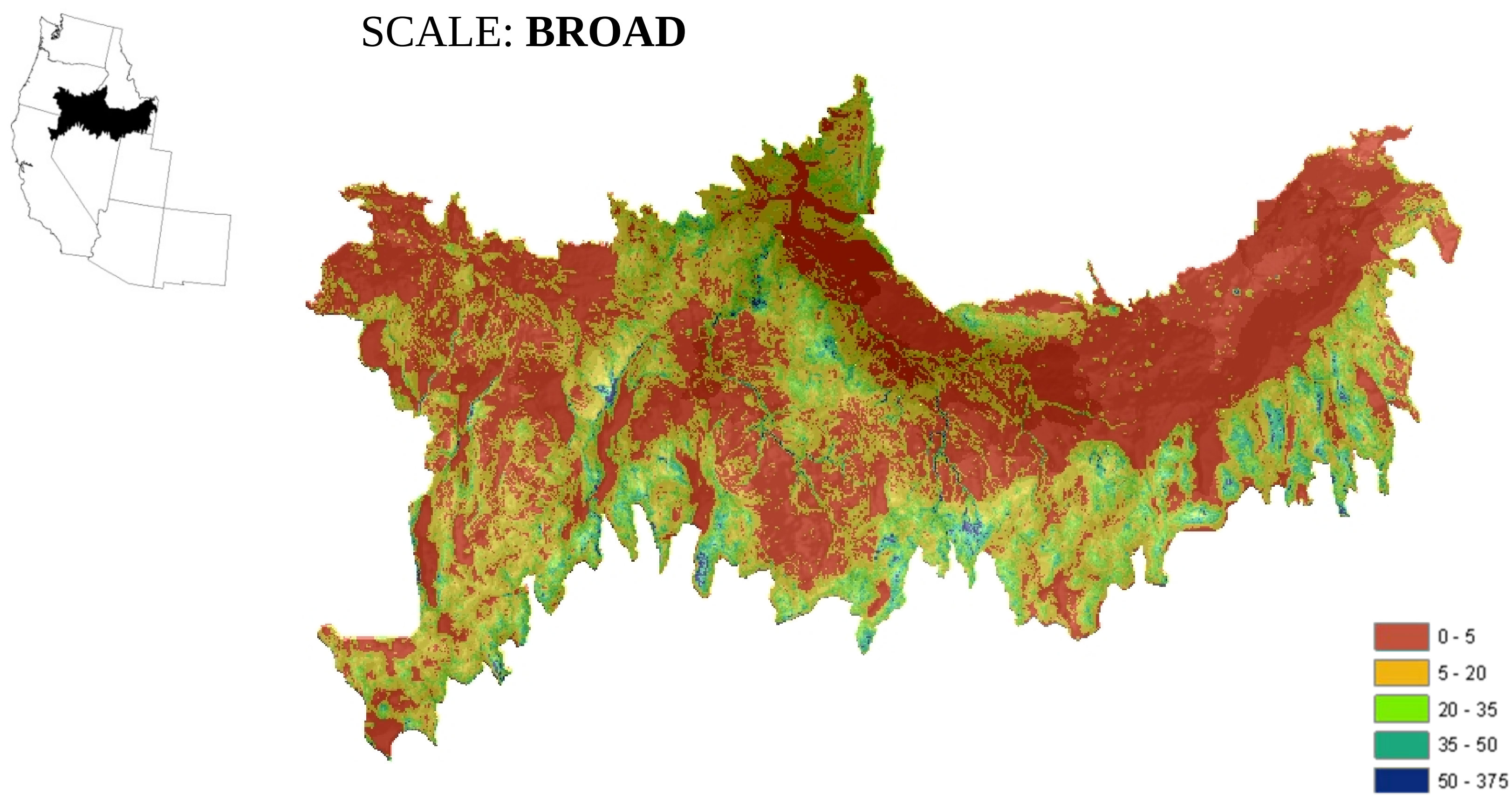

Figure 2.2. Slope (degrees) overlaid onto elevation in the selected study area to include the Snake River Plain and Basin and Range Ecoregions, and the Idaho National Laboratory.

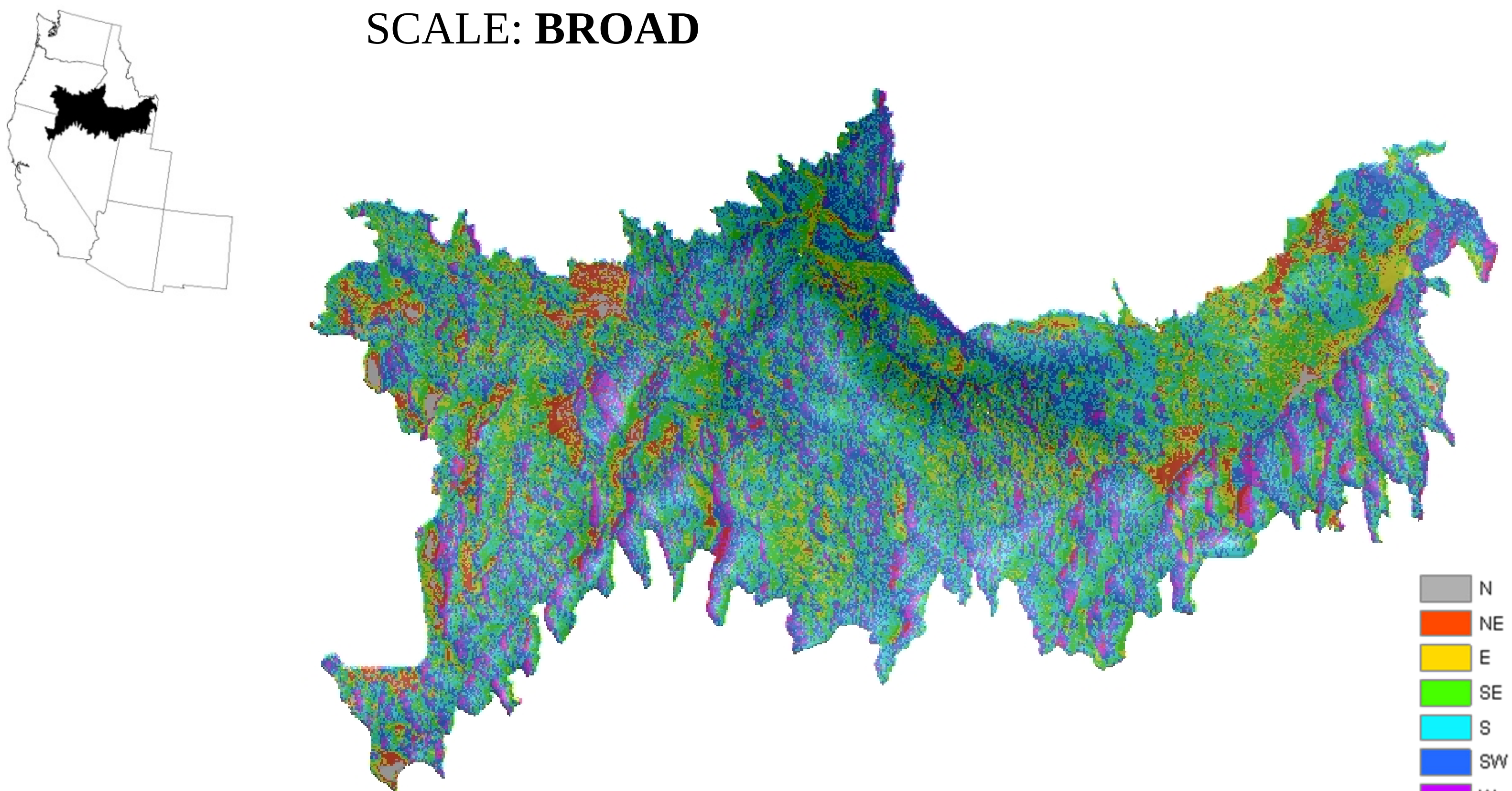

Figure 2.3. Aspect overlaid onto elevation in the selected study area to include the Snake River Plain and Basin and Range Ecoregions, and the Idaho National Laboratory.

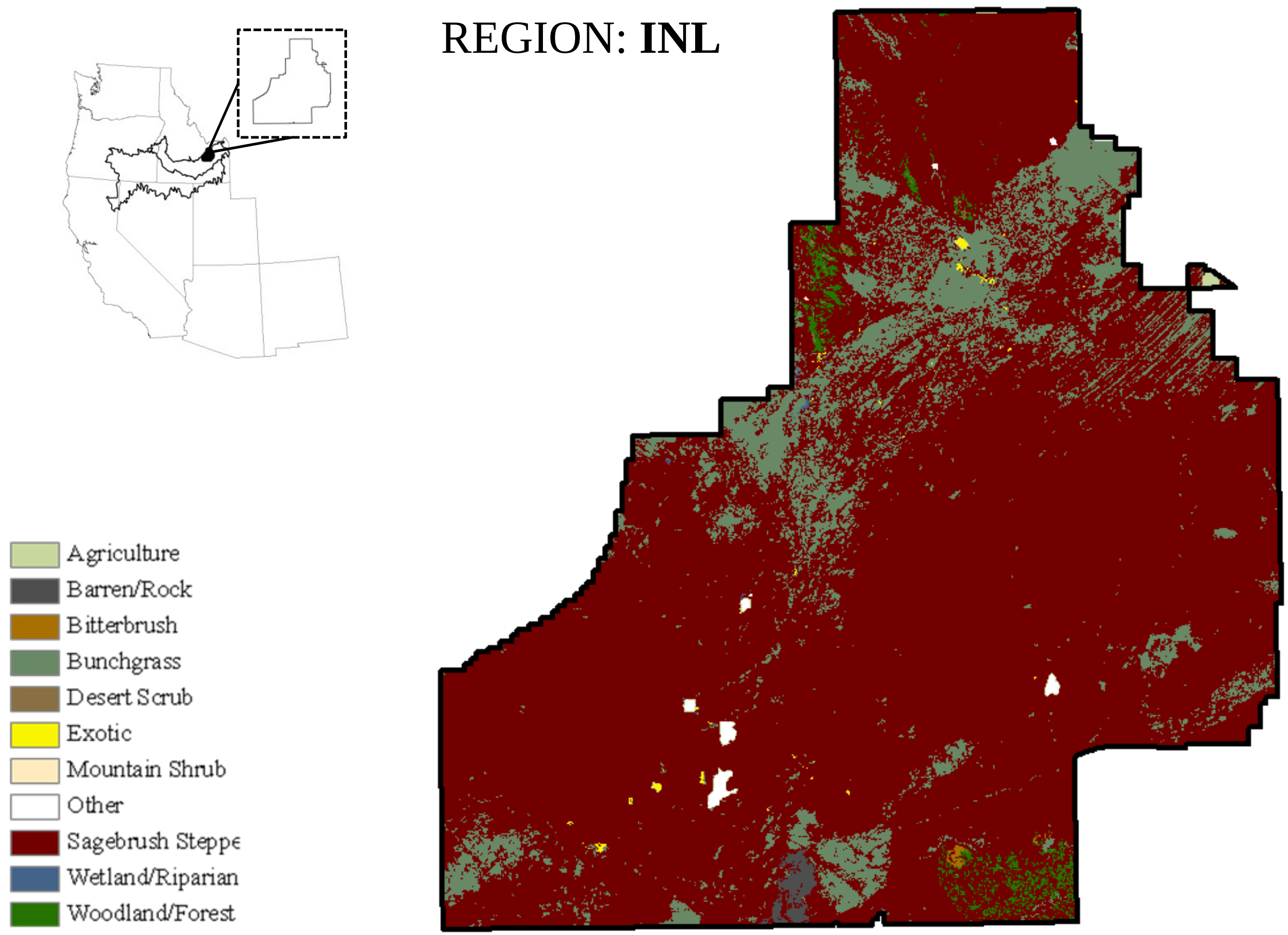

Figure 2.4. Map showing current vegetation derived from Sagesteppe in the Idaho National Laboratory.

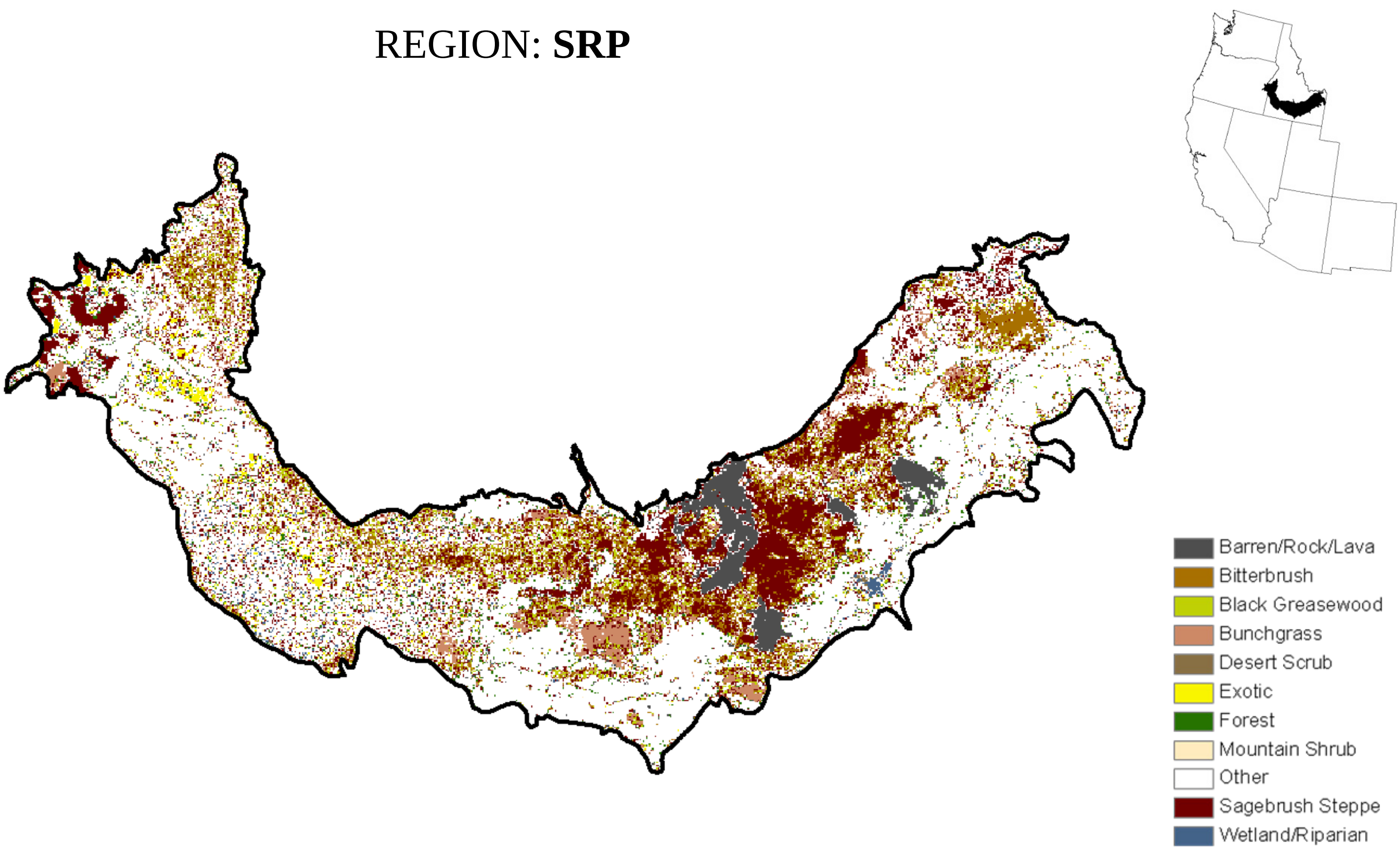

Figure 2.5. Map showing current vegetation derived from SAGESTEPPE in the Snake River Plain Ecoregion.

REGION: **NBR**

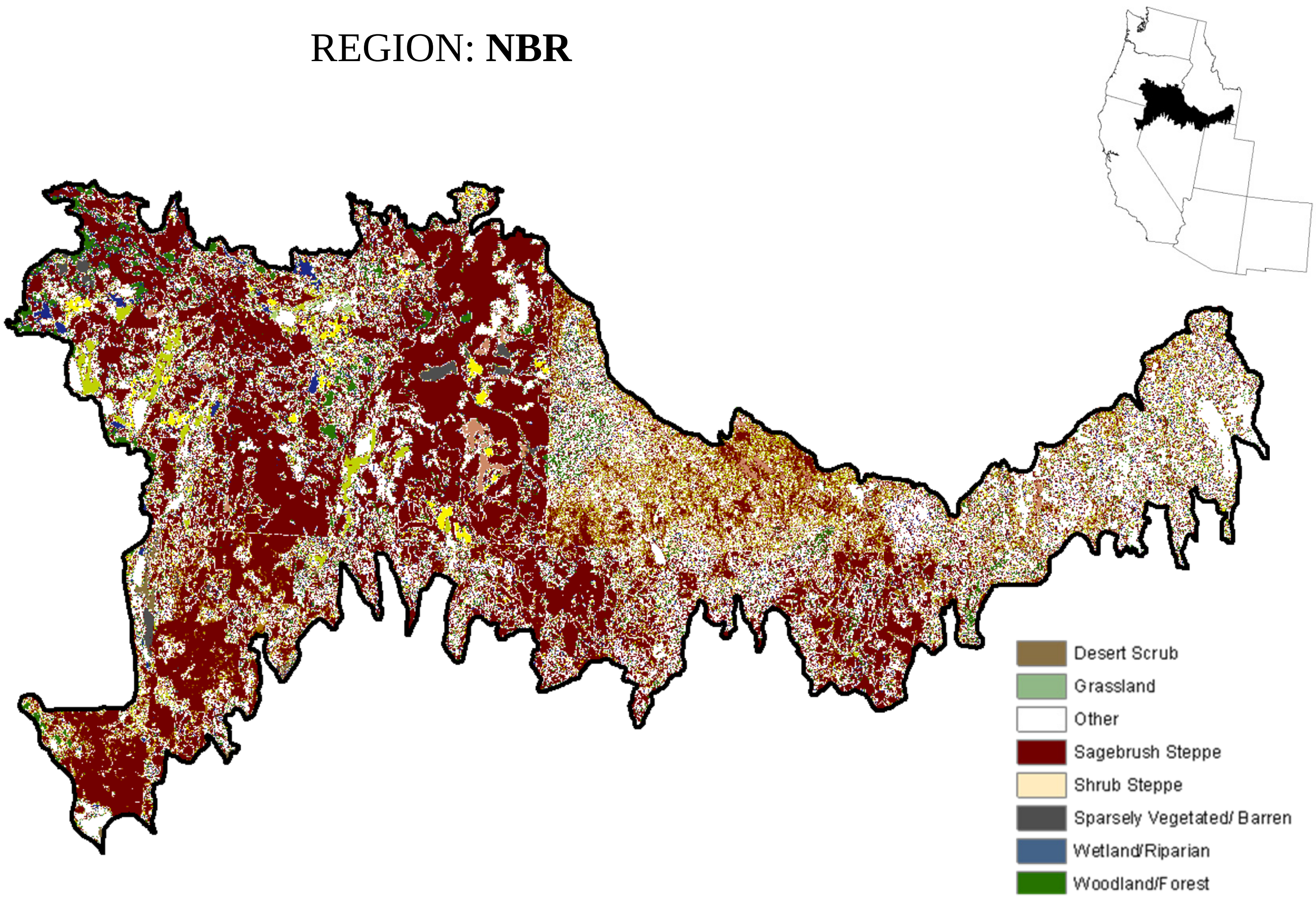

Figure 2.6. Map showing current vegetation derived from USGS in the Northern Basin and Range Ecoregion.

type in the region, occupying 59.3% of total land area **(Table 2.2)** and comprising 69.4% of the area burned by large fires **(Table 2.3)**. Large fires burned 10.2% of the sagebrush steppe (**Table 2.4**), and 5.8% of exotic vegetation in the study area. Bunchgrass occupied a 6.7% of the total land area, but formed a considerable (11.8%) portion of the area burned by large fires. Although exotics occupied only 2.8% of total land area, they comprised 5.6% of area burned by large fires. Area burned was assessed for area burned by fire at any time over the study and not for fire frequency.

Biophysical vegetation types were grouped together into eight broad cover types (**Figures 2.7 – 2.9**). Four BpS vegetation types were predominant across the study area (not including vegetation that comprised <1% of study area; **Table 2.2**). Six BpS vegetation types made up the majority of area burned by large fires at the broadest scale (**Table 2.3**). Desert-scrub biophysical setting (BpS) comprised the most abundant vegetation type in the study region (18.5%, **Table 2.2**), and also included the most abundant type within the burns from large fires (49.8%, **Table 2.3**). In desert scrub (BpS) vegetation, 25.5% of had class “D” fires present from 1960 to 2003 (**Table 2.4**). Sagebrush steppe (BpS) comprised 8.1% of total land area but formed 12.2% of area burned by large fires (**Tables 2.2 – 2.3**). Large fire was present in 13.7% of the sagebrush steppe (BpS) in the region over the study period (**Table 2.4**). If extreme events are a factor, such as wind and precipitation conditions, the mosaic of vegetation or topography may not be as important a factor in fire events. This is demonstrated in the fires of 1988 in Yellowstone National Park (Turner and Romme 1994).

There was a significant difference in the total number of fires between 1960 – 1982 and 1983 – 2003 (**Table 2.5)**, and for fire size (ha) between the first and second half of the study. This also held true for large fires occurrences, which were significantly different between the two time periods. At the broadest scale, the total number of large fires was higher during the 1982-2003 time period than in 1960-1982 (**Table 2.6**). The number of large fires increased linearly on a decadal scale. When all fires were included, the average number of fires increased across the decades, with the period 2000 – 2003 showing a decrease in average number of fires. For the purposes of my analyses, the time period for 2000 – 2003 is referenced as a decade when discussing decadal events. It is important to realize that this period is 6 years less than the other "decades" as referred to in the discussion. When viewed in this context, the average number of total fires for this 4-yr period was higher than that for 1960 – 1980, and size class "D" fires were higher in this time period than for any previous decade included in the analyses. The Northern Basin and Range Ecoregion experienced the majority of large fires, although the Snake River Plain Ecoregion had more fires from all size classes than did the NBR (**Table 2.7)**.

At the broadest spatial scale, average fire size across all size classes was higher in 1983 – 2003 than in 1960 – 1982 (**Table 2.8)**. Large fires experienced a linear increase on a decadal scale for both average fire size and total area burned, which peaked in 2000 – 2003 (**Table 2.9**). This same phenomenon occurred when all (size classes A – D) fires were averaged with respect to decade. Average total area burned for all fires was dramatically higher for the second half of the study (**Table 2.10**). This was also repeated

Table 2.3. Percentage of large fires (>5000 ha) by vegetation type in the INL and the Snake River Plain and Basin and Range Ecoregions.

SCALE: **BROAD**

Biophysical Setting (BpS)	**Class D Fires (%)**
Desert Scrub	49.8
Warm Desert Sparsely Vegetated Systems	24.8
Sagebrush Steppe	12.2
Semi-Desert Shrub-Steppe	2.9
Wetland/ Riparian	1.0
Other (< 0.01)	9.3

Current Cover Type	**Class D Fires (%)**
Sagebrush Steppe	* 69.4
Bunchgrass	11.8
Exotics	5.6
Salt Desert Scrub	4.3
Agriculture	2.3
Other (<0.01)	4.0

• Includes *Artemisia tridentata* ssp. *arbuscula* , ssp. *tridentata,* ssp. *vaseyana, and* ssp. *wyomingensis*

Table 2.4. Percentage of total land cover type (ha) that had class D fires present (>1%)

SCALE: **BROAD**

Biophysical Setting (BpS)	Class D Fires (%)
Desert Scrub	25.5
Sagebrush Steppe	3.7
Wetland/Riparian	5.5
Sparsely Vegetated Systems	2.5

Current Cover Type	Class D Fires (%)
Wyoming Big Sagebrush	* 8.2
Exotic	5.8
Salt Desert Scrub	3.7
Low Sagebrush	* 2.8
Bunchgrass	2.6
Mountain Big Sagebrush	* 2.5
Low Sagebrush/ Wyoming Big Sagebrush	* 1.7
Bitterbrush	1.7
Wyoming Big Sagebrush/ Basin Big Sagebrush	* 1.2
Basin Big Sagebrush	* 1.1
Agriculture	1.3

• Total sagebrush cover types: 17.5%

Table 2.5. Paired student *t*-test for fire occurrence and fire size for two time periods: 1960-1982 and 1983-2003.

SCALE: **BROAD**

		Large fires			All fires		
NBR/SRP/INL		**mean**	***t***	**P**	**mean**	***t***	**P**
Size (ha)	1960 - 82	11,819			472		
	1983 - 03	9,894	1.19	0.24	809	-4.915	**0.00 ***
Occurrence	1960 - 82	2.9			196		
	1983 - 03	14.7	-2.314	**0.036 ***	703.4	- 3.922	**0.002 ***

* Significant at alpha = 0.05

Table 2.6. The number of fires by size class, over time, in the study region to include the Snake River Plain and Northern Basin and Range Ecoregions, and the Idaho National Laboratory, 1960 - 2003.

SCALE: **BROAD**

Number of Fires

	Size-Class					
	A	**B**	**C**	**D**	**Total**	**Average**
Time period	**0 – 50**	**51 – 500**	**501 – 5000**	**5000+**		
1960 – 1969	89	169	72	9	**339**	**85**
1970 - 1979	664	473	141	17	**1 295**	**49**
1980 – 1989	786	877	500	74	**2 237**	**559**
1990 - 1999	2 116	1 332	1 332	73	**4 030**	**1 008**
2000 – 2003	1 458	878	2 116	92	**2 894**	**724**
Average (decade)	***5 113***	***7 458***	***8 322***	***53***	**2159**	
1960 – 1982	**1 029**	**908**	**369**	**45**	**2 351**	**588**
1983 - 2003	**4 088**	**2 821**	**1 319**	**220**	**8 444**	**2 111**
Difference	***3 059***	***1 913***	***950***	***175***	***6 093***	

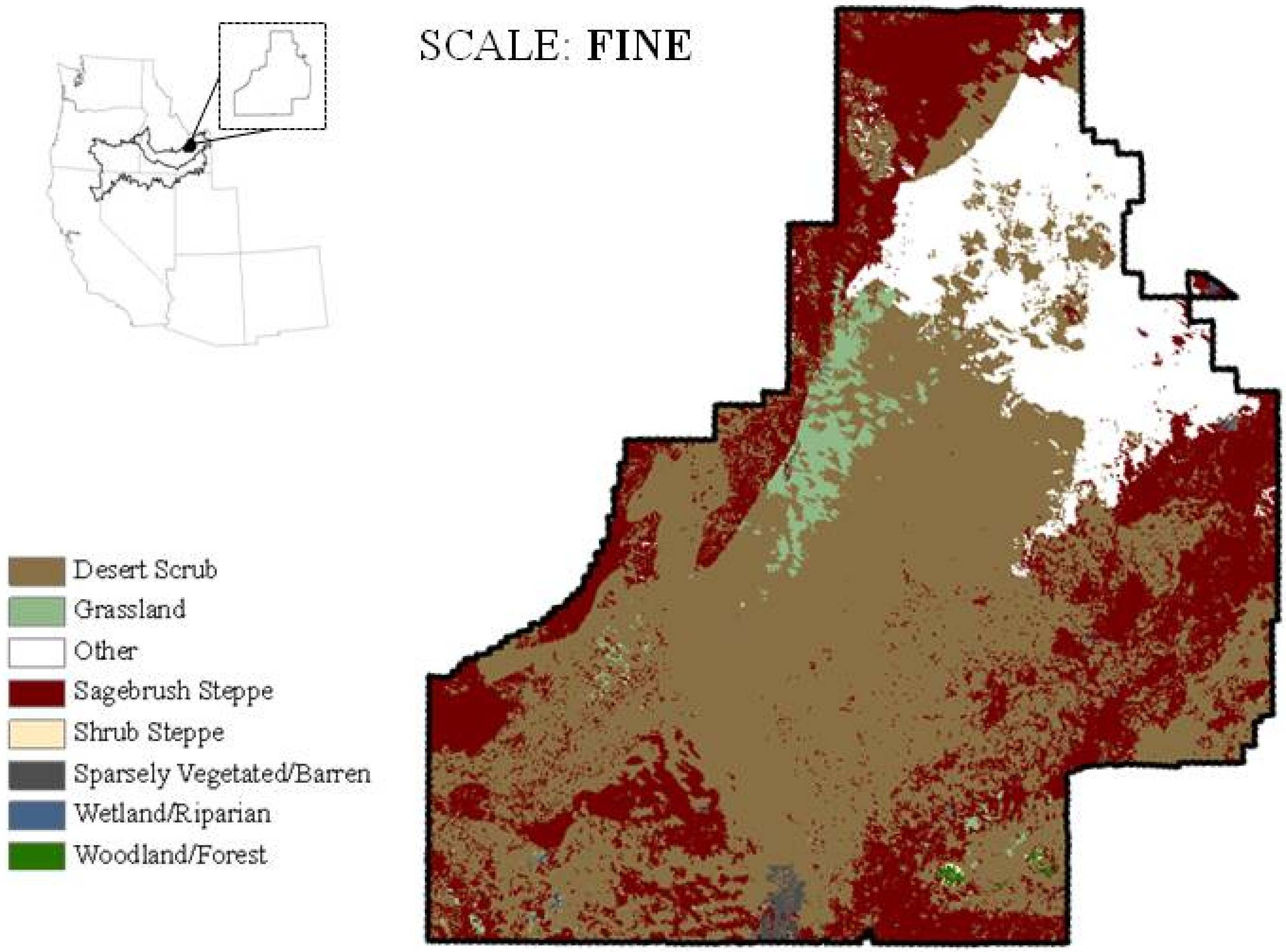

Figure 2.7. Map showing Biophysical Setting (BpS) of the Idaho National Laboratory derived from the Landfire Model.

REGION: **SRP**

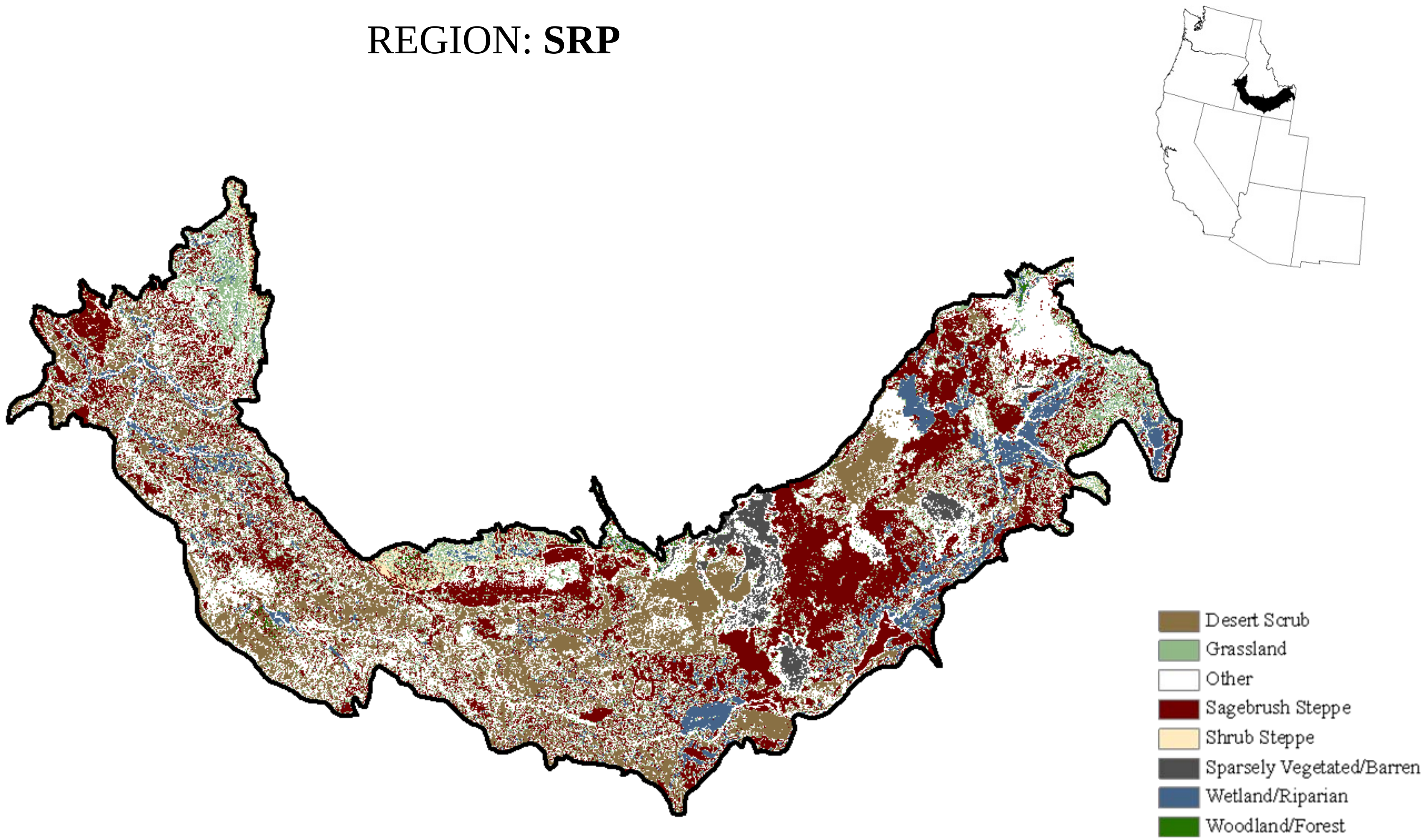

Figure 2.8. Map showing biophysical setting vegetation of the Snake River Plain Ecoregion derived from the Landfire Model.

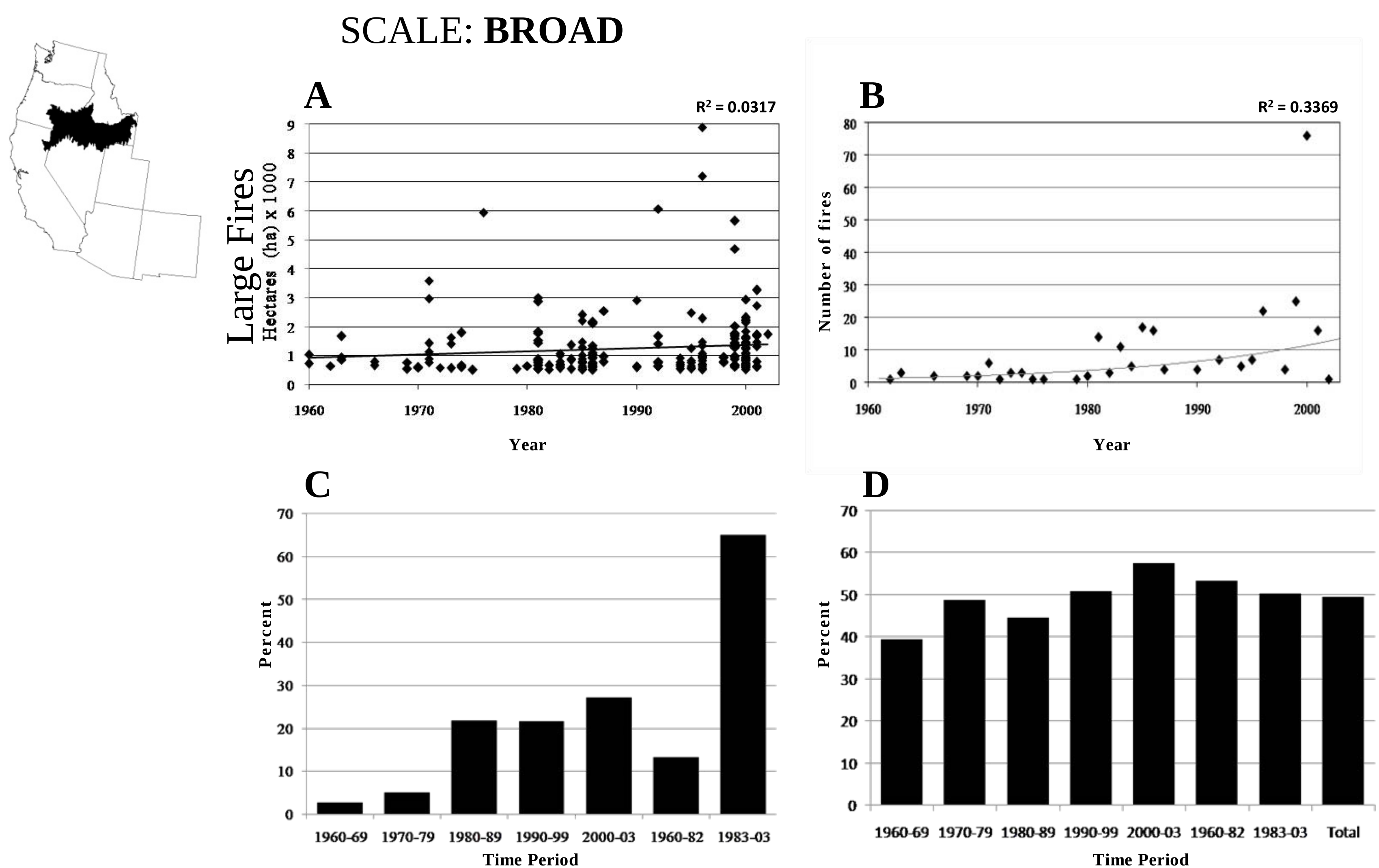

Figure 2.9. Large fires >5000 hectares by size, annually (Figure A), number of occurrences (Figure B) with exponential trend lines and R^2 values, and number of large fires as a percentage of all fires (size classes A-D) and the percentage of land area burned by decade (Figures C and D) in the selected study area to include the Snake River Plain and Basin and Range Ecoregions, and the Idaho National Laboratory, 1960- 2003.

Table 2.7. The number of fires by size class, over time, in the Snake River Plain Ecoregion, 1960- 2003.

REGIONS: **SRP**

Number of Fires

	Size-Class				
	A	**B**	**C**	**D**	**Total**
Time period	**0 – 50**	**51 – 500**	**501 – 5000**	**5000+**	
1960 – 1969	257	109	53	6	**425**
1970 - 1979	560	416	139	18	**1133**
1980 – 1989	625	734	401	54	**1814**
1990 - 1999	1526	734	279	49	**2588**
2000 – 2003	74	11	0	0	**85**
1960 – 1982	**997**	**734**	**291**	**32**	**2 054**
1983 - 2003	**2 045**	**1 270**	**581**	**95**	**3 991**
Difference	***1 048***	***536***	***290***	***63***	***1 937***

Table 2.8. The number of fires by size class, over time, in the Northern Basin and Range Ecoregion, 1960- 2003.

REGIONS: **NBR**

Number of Fires

Size-Class					
	A	**B**	**C**	**D**	**Total**
Time period	**0 – 50**	**51 – 500**	**501 – 5000**	**5000+**	
1960 – 1969	152	61	18	4	**235**
1970 - 1979	108	106	4	0	**170**
1980 – 1989	184	168	110	22	**485**
1990 – 1999	412	461	206	26	**1 073**
2000 – 2003	1380	864	471	93	**2 808**
1960 - 1982	**374**	**231**	**85**	**17**	**707**
1983 - 2003	**1 862**	**1 429**	**724**	**128**	**4 143**
Difference	***1 488***	***1 198***	***639***	***111***	***3 436***

Table 2.9. The average fire size in hectares by size class in the Northern Basin and Range Ecoregions, 1960- 2003.

REGIONS: **NBR**

Average Fire Size (ha)

	Size-Class				
	A	**B**	**C**	**D**	**Average**
Year	**0 – 50**	**51 – 500**	**501 – 5000**	**5000+**	
1960 – 1969	9	192	1 350	8 558	**2 527**
1970 - 1979	12	175	1 275	14 503	**4 928**
1980 – 1989	17	189	1 566	10 533	**3 076**
1990 - 1999	13	178	1 463	13 221	**3 719**
2000 – 2003	13	191	1 582	13 395	**3 818**
1960 – 1982	**13**	**152**	**1 407**	**11 135**	**3 177**
1983 - 2003	**44**	**186**	**1 515**	**12 583**	**3 582**
Difference	***31***	***34***	***108***	***1 448***	***405***

Table 2.10. The total area burned in hectares by size class over time, in the Idaho National Laboratory, Snake River Plain and Northern Basin and Range Ecoregions,1960- 2003.

SCALE: **BROAD**

Total Area Burned (ha)

	A	B	C	D	Total	Average
Year	**0 – 50**	**51 – 500**	**501 – 5000**	**5000+**		
1960 – 1969	3 846	32 255	95 872	85 579	**217 551**	**54 388**
1970 - 1979	8 191	82 931	183 867	261 052	**536 040**	**134 010**
1980 – 1989	12 938	165 657	783 108	768 902	**1 730 606**	**432 651**
1990 – 1999	3 526	207 599	702 011	965 111	**1 902 294**	**475 573**
2000 – 2003	9 411	167 774	737 887	1 245 048	**2 170 304**	**542 575**
Average (decade)	***7 582***	***75 074***	***500 549***	***665 138***		
1960 - 1982	**15 564**	**166 059**	**316 438**	**567 312**	**1 065 374**	**266 343**
1983 - 2003	**55 912**	**480 155**	**2 186 308**	**2 759 046**	**5 491 421**	**1 372 855**
Difference	***40 348***	***314 096***	***1 869 870***	***2 191 734***	***4 426 047***	***1 106 512***

for large fires (size class D). The average fire size for large fires peaked in 2000 – 2003. This pattern was also revealed for total fires (size class A-D).

Mid- Scale

The Snake River Plain experienced a considerable increase in average area burned in 1983-2003, inclusive of all fire size classes, than in 1960-2003 (**Table 2.11**). This pattern was consistent across size classes, including size class "D" fires. The average fire size for size class "D" fires was higher on average during the second half of the study period than during the first half (**Table 2.12**). Average fire size was higher during 1983-2003 across all fire size classes than during 1960-1982. The total number of large fires (size class "D") during 1983 - 2003 was more than 3 times that of the 1960 - 1982 time period. However, no large fires were recorded in the SRP during 2000 – 2003. The total number of fires inclusive of all size classes in the SRP was consistently higher across all size classes during the second half of the study.

In the SRP, precipitation during the summer months is relatively minimal despite being accompanied by lightning activity during frequent thunderstorms. When high winds are superimposed on these conditions, combined with highly flammable exotic annuals, large fire events are likely to occur with increasing probability. The INL was used as the finest spatial scale, and did not display any patterns in fire history due to the small sample size. This finding supports the idea that fire patterns may become apparent only when values are averaged across broader spatial scales, similar to climate variables.

In support of the first null hypothesis, large fires tended to occur in aggregated patterns (more abundant in some locations) across the study region. Large fires tended to move from east to west on a decadal scale, and occurred heavily in the southern region of the Northern Basin and Range from 2000 – 2003 (**Figure 2.10)**. Thus, the spatial distribution of large fires reflects more a non-random distribution, possibly in part due to topography, microclimate, vegetation variability, and human presence. This result challenges the assumption of homogeneous spatial distribution of fires across the region. The time period for which fire records were available indicate that a change has occurred in the fire regime in this region since 1960.

LIMITATIONS

Because this analysis substantially relied on the Great Western Fire Map dataset it is subject to all the limitations in accuracy and precision associated with this dataset. The fire data was compiled using records from a variety of sources and duplicates were present throughout the dataset. The dataset was not assessed for accuracy; however, I made every effort to remove obvious duplicates. Because misclassification of the fires by year and possible repetition of fire occurrences could significantly alter the results of the analyses, particularly on an annual or decadal basis, it is advisable that this limitation be considered for future analyses.

The Great Western Fire Map recorded only three large fires in the Idaho National Laboratory during the study period, totaling 18 075 ha. I used this dataset in my analyses to maintain consistency for the three spatial scales; however, it should be noted that another dataset provided by S.M. Stoller Corporation showed a total of 5 large fires as

Table 2.11. Total area burned in hectares by fire size category over time, in the Snake River Plain Ecoregion, 1960- 2003.

REGION: **SRP**

Total Area Burned (ha)

	A	B	C	D	Total	Average
	0 – 50	**51 – 500**	**501 – 5000**	**5000+**		
Time period			**Hectares**			
1960 – 1969	2 420	3 937	69 446	44 505	**120 308**	**30 077**
1970 - 1979	7 528	71 532	178 980	261 051	**519 091**	**129 773**
1980 – 1989	10 423	142 120	599 544	547 946	**1 300 033**	**325 008**
1990 – 1999	19 998	126 602	413 917	656 220	**1 216 737**	**304 184**
2000 – 2003	1 242	2 030	0	0	**3 272**	**1 636**
Average (decade)	***8 322***	***69 244***	***252 377***	***301 944***		
1960 - 1982	**12 286**	**116 540**	**391 279**	**363 118**	**883 223**	**220 806**
1983 - 2003	**29 325**	**229 681**	**870 608**	**1 146 604**	**2 276 218**	**569 055**
Difference						

Table 2.12. The average fire size in hectares by fire size category over time, in the Snake River Plain Ecoregion, 1960- 2003.

REGION: **SRP**

Average Fire Size (ha)

	Size-Class				
	A	**B**	**C**	**D**	**Average**
	0 – 50	**51 – 500**	**501 – 5000**	**5000+**	
Time period		**Hectares**			
1960 – 1969	9	189	1 310	7 418	**2 232**
1970 - 1979	13	172	1 288	15 059	**4 133**
1980 – 1989	16	195	1 478	8 928	**2 654**
1990 - 1999	13	172	1 484	13 392	**3 765**
2000 – 2003	16	184	0	0	**100**
Average (decade)	***13***	***182***	***1 390***	***11 199***	***3 196***
1960 – 1982	**12**	**186**	**1 347**	**9 891**	**2 859**
1983 - 2003	**47**	**548**	**2 996**	**12 027**	**3 904**
Difference	***35***	***362***	***1 649***	***2 136***	***1 045***

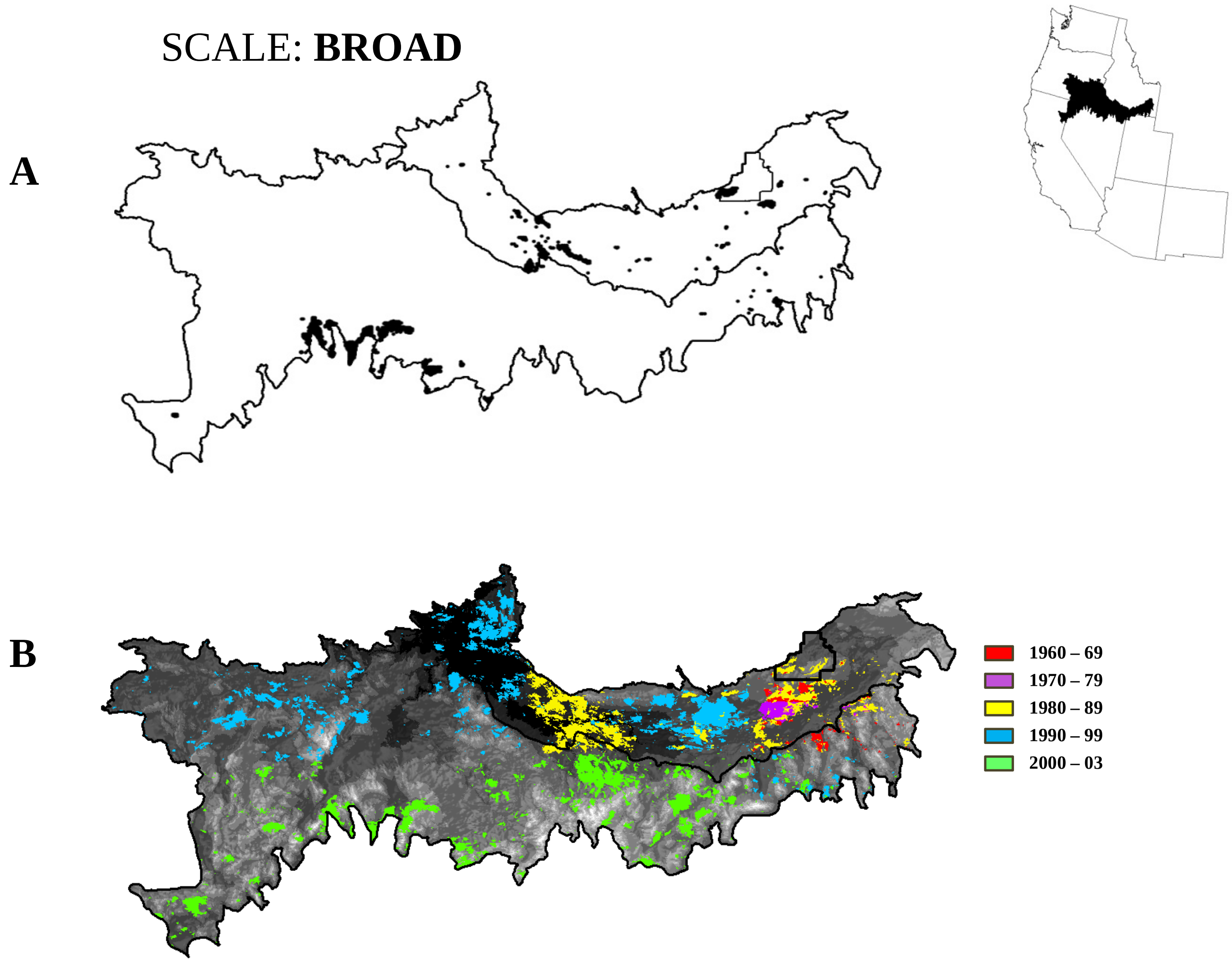

Figure 2.10. Point locations of large fires (>5000 ha) (Figure A) and location of all fires (size classes A – D) by decade (Figure B) in the selected study area 1960 – 2003.

having occurred since 1994. This dataset showed that 52 880 ha were burned by these large fires.

There is no standard methodology of reporting fires by all agencies from which fire data was compiled; it is not known whether instances of several small fires being reported as one larger fire occurred, and to what extent. Therefore, there may be instances when the accuracy of fire size classification was lower. If standardization of fire reporting is achieved across agencies, we may be able to perform more accurate comparisons in the future.

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CHAPTER 3 - CLIMATE AND FIRE

INTRODUCTION

Disturbance regime modification is one of the most important indirect effects of climatic change on ecosystems as ecosystem become more sensitive to changes in climate (Chapin *et al*. 2004). In sagebrush steppe, there are several distinct sources of disturbance that are expected to respond sensitively to climatic change, including changes in warming and precipitation patterns and inputs. Climate change has the potential to drastically alter the fire regime of sagebrush steppe. An increase in temperature could result in earlier and longer fire seasons (Flannigan *et al*. 2000; Gitay *et al*. 2001). The occurrence of large fires in the western United States raises questions about the effect of climate change on fire regimes in the past and future. Increased wildfires and conversion to cheatgrass dominance in shrublands in the Great Basin causes broad-scale conversion of rangeland carbon sinks to carbon sources (Bradley *et al*. 2006).

The Intergovernmental Panel on Climate Change (IPCC) has developed four categories of scenarios based on population growth, economies, and technologies, to represent possible changes in greenhouse gas and sulfate aerosol emissions for use in climate models. Warming is projected by most models to occur for all seasons, though the greatest warming effect is anticipated to occur during winter and spring months, which will likely give non-native grasses a competitive advantage over native vegetation, which remains in a dormant state longer. The factors that control the trajectory of climate are not fully known, and it is not currently possible to predict the state of the atmosphere, ocean, and other climate components in full detail. For this reason, a few factors can be considered, with the other factors interpreted as internal variation in the climate system, or rather, “noise” (Mearns *et al.* 2001). Therefore, it is important to look at many climate model forecasts to fully understand the range of possible future climate.

In this chapter I discuss the linkages between climate, CO_2 and potential direct and indirect impacts on large fire size and frequency in the sagebrush steppe ecosystems. Climate and short-term weather (i.e., wind, relative humidity, temperature, and precipitation) directly influence fire behavior and distribution through effects on fuel moisture, probability of ignition, and fire spread (Rothermel 1983). If precipitation changes from winter- to spring-dominated in the study region, native annuals may be eliminated, reducing live biomass and increasing bare ground available (Svejcar *et al.* 2003). These changes facilitate soil erosion and invasion by non-native exotic species. Aboveground biomass production is a function of site potential, length of time since last burn, and land-use history. Locations that have experienced a high fire return interval typically have fewer shrubs and more grasses and forbs. Site potential depends on interactions among annual and seasonal precipitation, redistribution of precipitation due to winds and topography, aspect, and the water holding capacity of the soil (a function of depth, texture, and rock content).

Climate Scenarios

The probability of future regional fire years in sagebrush steppe under two different climate scenarios is discussed. The IPCC assessment scenario representing the upper limit of projected greenhouse gas change, known as the A2 scenario, is vastly different from the B1 scenario, which represents the lower end of projected emissions, particularly beyond 2050. The climate forcing of all scenarios is similar until mid-century. Model projections using the A2 scenario forecast a relatively high rate of warming, an increase in severity of precipitation events, and involves a relatively strong increase in atmospheric carbon dioxide emissions over the 21st century. The B1 emissions scenario forecast a cooler, drier system than the results from the A2 scenario. Models using the B1 scenario forecast atmospheric CO_2 increasing slightly in future decades before decreasing to lower than current levels by 2100. However, this scenario is also expected to result in an increase in global temperatures.

Even under the lower B1 scenario, many ecological and water-related impacts may exceed acceptable thresholds. Estimates of future emissions are uncertain because they rely on estimates of future economic growth, changing energy use and policy. Many of the current climate models do not incorporate extreme events or seasonality. Other aspects of climate change may be very important as well but are not explored here: wind speed and direction; frequency of extreme cold, hot, wet, or dry events; and cloudiness. Therefore, the discussion presented here is not a prediction of what will happen but is based on scenarios of what might happen given continued, unrestrained growth in emissions of greenhouse gases.

Warming projected for the end of the 21st century ranges from about 1.5 to 6°C if both emissions scenarios are considered. Beyond mid-century, the projections of warming depend increasingly on emissions in the next few decades and on actions that would limit or increase emissions. Precipitation changes projected by most models are moderate, and will likely only be discernible from natural variability towards the latter part of the 21st century. Most global climate models predict increasing winter precipitation and decreasing summer precipitation. The average warming rate in the

Northwestern U.S. during the next several decades is projected to be in the range 0.5-3.2°C by the 2040s (Mote *et al.* 2003). If precipitation during the summer growing season increases, additional water may become available for shallow rooted grasses and forbs in the region. Alternately, if warming increases during the summer season, water may not be available for ecosystem processes as evapotranspiration increases.

El Niño Southern and Pacific Decadal Oscillations

The El Niño Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO) are atmospheric processes related to broad-scale patterns of variations in sea surface temperature. These patterns oscillate between warm and cool phases on yearly (2-7 years for ENSO) to decadal (20-30 years for PDO) temporal scales and may be an underlying driver of interannual climate variation (Biondi *et al*. 2001, Dettinger *et al*. 2001), though this is not clear for the Great Basin. In the warm phase of ENSO (El Niño) and PDO, dry conditions typically exist in the Great Basin. The effects of cool-phase ENSO, or La Niña conditions, and cool-phase PDO are opposite to those associated with the warm phases.

The PDO is a pattern of high and low pressure systems over the northern portions of the Pacific Ocean that operates on a decadal time scale and correlates with relatively wetter or drier periods in the western U.S. The predictability of the regional effects of ENSO is linked to the phase of the PDO (Gershunov and Barnett 1998). During the cool phase of the PDO, the effects of La Niña may be enhanced whereas the effects of El Niño on climate may be weakened. La Niña originates with anomalies in tropical sea-surface temperatures, but affects climate across western North America, especially winter conditions. In the Great Basin region, El Niño typically produces warmer drier winters (positive phase) and ENSO conditions tend to produce cooler wetter winters (negative phase) (McCabe and Dettinger 1999).

When the PDO is in a positive phase, as it has been since 1977, regional and large regional fire events have occurred more frequently. Shifts in the PDO regime have occurred in 1925, 1947, and 1977, and another shift may have begun around 1995 (Mantua *et al.* 1997). Regional fire years tend to occur during La Niña events or in years of drought (Hessl *et al*. 2004).

ENSO may have the ability to affect fire occurrence and fire spread in arid ecosystems by increasing herbaceous understory vegetation (fine fuels) during wet years prior to the fire season or through low fuel moisture conditions during the dry fire season (Hessl *et al*. 2004), or alternatively through its effect on the timing of snowmelt, thereby changing the likelihood of large fires (Heyerdahl *et al*. 2002).

METHODS

Climate and Fire

The objective of this chapter is to explore the relationships between fire and climate in the Idaho National Laboratory, and the Snake River Plain and Northern Basin and Range ecoregions by investigating a range of potentially important climatic variables (i.e., precipitation and temperature). Averaged monthly and annual maximum temperature and average precipitation data were obtained using the Parameter Elevation Regressions on Independent Slopes Model (PRISM). PRISM uses point data and a digital elevation model (DEM) to generate gridded estimates of climate parameters on a monthly and annual basis (Daly *et al*. 1994). PRISM accounts for spatial variations in climate due to elevation, terrain orientation and steepness, moisture regime, coastal proximity and inversion layer. Monthly and annual PRISM data were downloaded from the Prism Group web site (http://www.prism.oregonstate.edu) for the years 1960 to 2003, on a four-kilometer grid scale covering the study region. A total of 946 ASCII files were downloaded. Clipping of the data to the study area was completed as an initial step. These clipped data were then projected in a two-step projection process from their native projection of WGS72 to the transverse Mercator projection NAD 83.

Monthly precipitation and maximum temperature values in each ecoregion were averaged to obtain mean seasonal values for spring (January - April), summer (May – August), autumn (September - October), winter (November - December), and annual (January – December). Seasonal averages were calculated by performing arithmetic operations on monthly data in Spatial Analyst in ArcMap using the Raster Calculator. A dataset describing seasonal climate values for size class “D” fires was produced using fire polygons as zones to average the PRISM raster datasets within each fire perimeter. Summaries for individual fires were obtained using the Zonal Statistics function in Spatial Analyst and selecting unique fire ID’s from the attribute table.

The datasets were imported into SYSTAT 11 statistical software and correlations between seasonal/ annual precipitation and fire frequency/ size were examined using Pearson’s product moment correlations. Temporally, large fires were correlated with climate variables in the year preceding that of the fire, and for two years preceding the fire to determine if there was a lag response.

I obtained a raster dataset for big sagebrush depicting the probability of species occurrence from the Nature Conservancy (Gradient Analysis-*Artemisia tridentata*, 1998). The dataset was downloaded from the Interior Columbia Basin Ecosystem Management Project website (http://www.icbemp.gov/spatial/veg/). It shows the probability of species occurrence for present atmospheric conditions and predicted species distribution changes under a climate change scenario in which CO_2 is doubled. Fire occurrence is not accounted for in this prediction. The layers were generated using FORTRAN program and imported into ARC/INFO using GRID-ASCII. General linear modeling (GLM) was used to quantify the presence or absence of plots as functions of climate variables attributed to the plots. The resulting model equation was applied to 2-km cell-size climate data layers to predict probabilities of occurrence. Probabilities of occurrence ranging from 0 to 1 were mapped as one of four classes (0 to 0.25, 0.26 to 0.50, 0.51 to 0.75, and 0.76 to 1.00). The product is a map of probability statements about the distribution of species. I imported the data into a spreadsheet and calculated the percent of study area that had data available that would experience a change between the current atmospheric CO_2 and the probability of occurrence under a doubling of atmospheric CO_2.

RESULTS AND DISCUSSION

Precipitation

Monthly precipitation patterns from 1960 – 2003 showed relatively higher (>25mm per month) precipitation between the November and May, with precipitation dropping below

35mm per month between June and October (**Table 3.1**). Average precipitation for the second half of the study showed a decrease during the summer and autumn months (2.5 and 5.1 mm, respectively), a slight increase in winter (1.3 mm), and no change for annual precipitation. Average spring precipitation increased 5.1 mm for the 1983-2003 period.

A shift to a spring/summer dominated precipitation pattern may lead to the forb component being lost or severely reduced, with the potential to reduce ecosystem biodiversity (Bates *et al.* 2006). Winter precipitation can significantly influence the amount of surface soil moisture and percolation into deep soils in the sagebrush-steppe ecosystem (Schwinning *et al.* 2003). If precipitation during the winter decreases, spring drought may be enhanced. Bates (*et al.* 2006) suggested that an increase in winter precipitation combined with summer drought would be unlikely to cause major changes to vegetation composition or productivity of big sagebrush communities in the northern Great Basin. In one study on the Colorado Plateau, growth of a shrub/grass community was more sensitive to spring drought than summer drought (Schwinning *et al.* 2005). Kwon (*et al.* 2007) found the most active period of plant growth in mountain big sagebrush, based on net carbon uptake, occurred in early June. Some studies observed germinated cheatgrass mortality after a warm, dry autumn (Harris 1967; Piemeisel 1951), and mortality following low precipitation in late spring (Young *et al.* 1969). Fall and spring precipitation are influential to cheatgrass germination and growth. Cheatgrass relies on a shallow moisture supply during germination in the fall, and subsequent growth permits average annual production if small amounts of moisture occur in intervals during spring. Thus, cheatgrass germination and growth may have been impacted to some extent in the Northern Great Basin over the course of our study due to a decrease in fall precipitation, though this was probably negligible because spring precipitation increased.

Average annual precipitation on a decadal scale decreased between 1960 and 2003, with 2000 – 2003 experiencing the lowest average annual precipitation (**Table 3.2**). At the same time, average annual maximum temperature trended up, with the highest average temperatures for 2000 – 2003. Despite this decrease, response of NPP in arid ecosystem is relatively more dependent on variability of precipitation than on total annual precipitation (Knapp *et al.* 2002; Fay *et al.* 2003; Huxman *et al.* 2004; Bates *et al.* 2006). My analyses show that average annual precipitation (year of fire) and fire size or number of large fires are not correlated. This result supports previous studies in semi-arid regions of the western U.S. (D'Elia 1998; McKelvey and Busse 1996). Intra-seasonal variations in precipitation timing and amount affect moisture balance through abiotic (e.g., drought, soil water recharge) and biotic (e.g., stomata control) processes, which are closely coupled with carbon balance (Kwon *et al.* 2007). Water limitation in arid and semi-arid regions can affect vegetation dynamics and ecosystem process on the scale of hours to decades (Westoby *et al.* 1989; Walker 1993; Schwinning *et al.* 2004).

In one study in which an ecosystem process model was applied, cheatgrass productivity responded to changes in moisture available for plant growth; a 10% increase or decrease in total precipitation caused a comparative 10% change in leaf carbon accumulation (Kremer *et al.* 1996). Their simulations suggested that changes in the magnitude of average precipitation indicated by GCMs are within the adaptive capability of cheatgrass and may prevent significant variations in soil water or community composition. The change in timing of precipitation, with an increase of precipitation

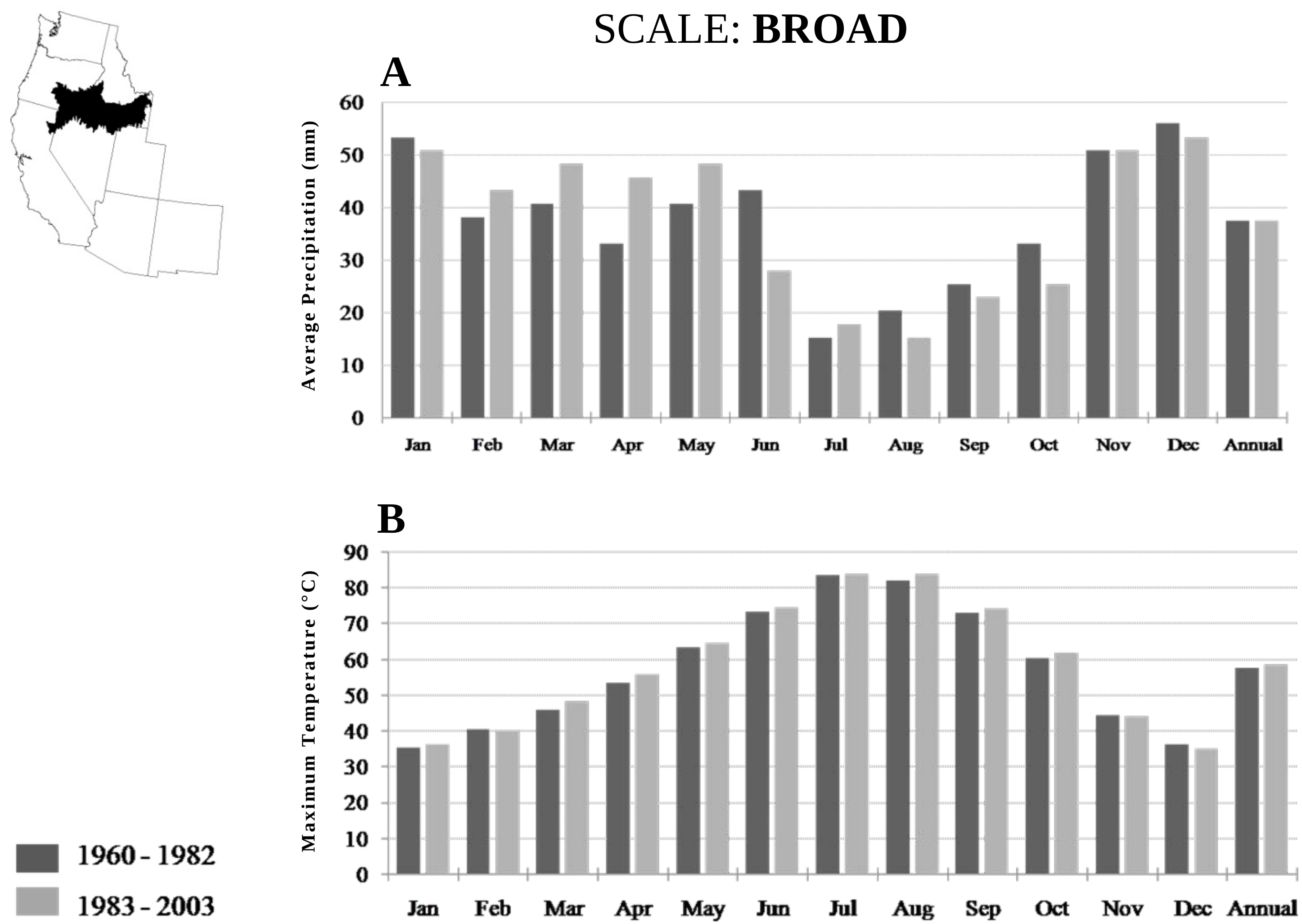

Table 3.1. Average monthly and annual precipitation (Figure A) and maximum temperatures (Figure B), for the periods 1960 – 1982 and 1983 - 2003 in the selected study area to include the Snake River Plain and Basin and Range Ecoregions, and the Idaho National Laboratory.

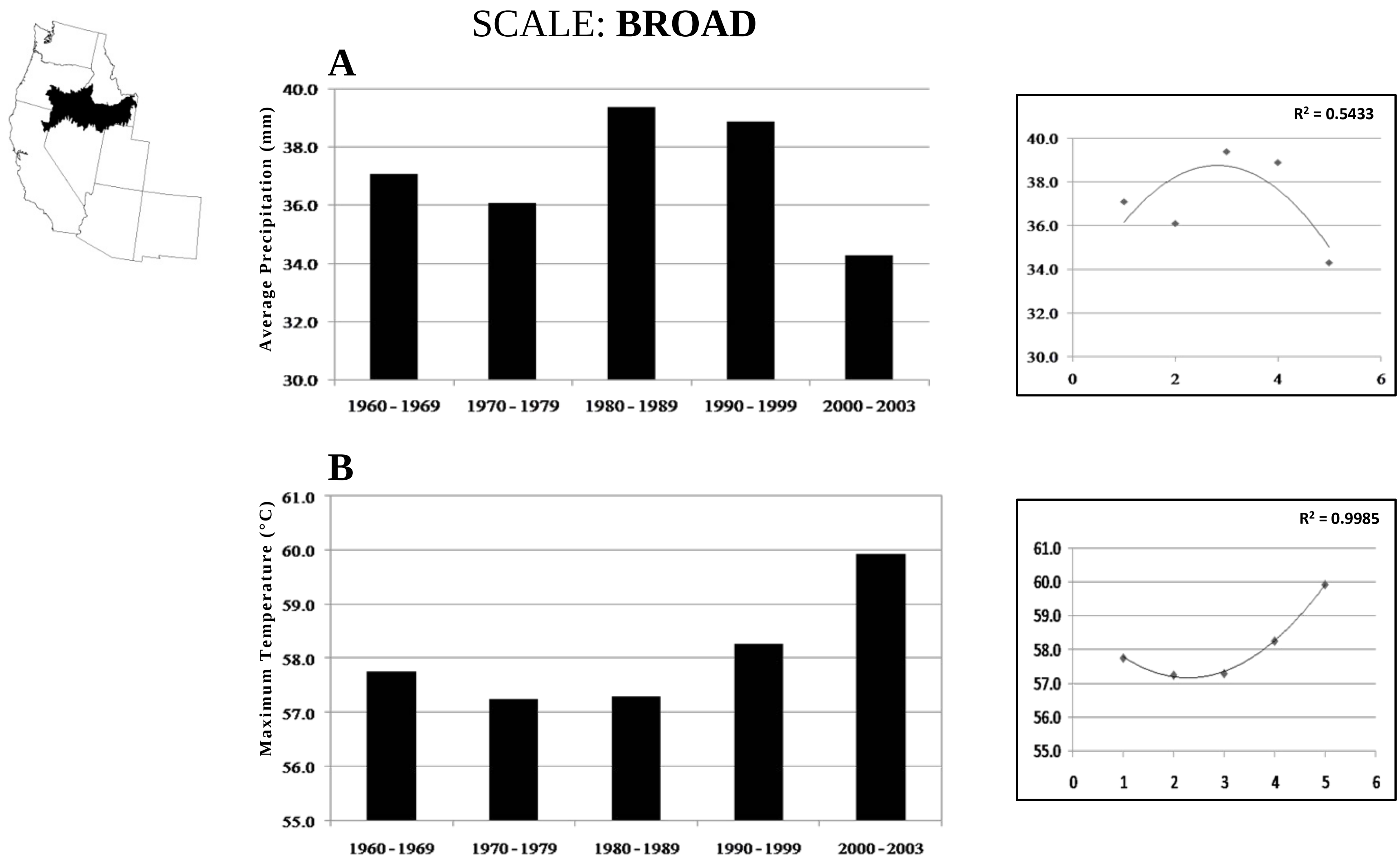

Table 3.2. Average annual precipitation (Figure A) and maximum temperatures (Figure B), by decade in the selected study area to include the Snake River Plain and Basin and Range Ecoregions, and the Idaho National Laboratory. Insets at right show exponential (top) and polynomial (bottom) trend lines with R^2 values.

during the winter and spring seasons, could be beneficial to exotic grasses early in the summer. Warmer, drier summer or fall seasons such as observed in my study, could increase large fire risk in sagebrush steppe because there would be abundant fine, dry fuels.

I found no significant correlation between the number of large fires or fire size, and spring/ summer precipitation the year of the fire event. Lagged effects for average yearly precipitation one year previous to the large fire events were weakly positively correlated (r^2 = 0.054, P = 0.031) (**Table 3.3**). Lagged effects were not observed for the two years prior to the large fire event at any spatial scale.

Temperature

July and September were on average the driest and warmest months. Average seasonal maximum temperature was 0.9 – 1.4°C warmer during 1983 – 2003 than for the first half of the study period, with the exception of winter, which averaged 0.6°C cooler (**Table 3.4**). Averaged annual maximum temperature was still 0.9°C warmer overall despite this slight winter cooling. Warmer spring conditions result in earlier green-up, which may result in an earlier fire season as the vegetation is cured and vulnerable to fire earlier. These conditions result in a high fire potential for a longer period of time. Soil moisture from winter is important for spring green-up and the inception of the growing season is largely influenced by air temperature (Kremer *et al.* 1996). Soil moisture in the Great Basin is generally replenished during the winter and early spring months, providing moisture for the subsequent growing season, and is gradually diminished through surface

Table 3.3. Pearson's product-moment correlations between precipitation (PPT) and maximum temperature (TMAX), and large fire size (ha).

SCALES: **BROAD, MEDIUM and FINE**

	Current year		1-yr Previous		2-yr Previous	
	r^2	P	r^2	P	r^2	P
BROAD SCALE						
Spring/Summer PPT	0.031	0.123	0.089	0.527	0.040	0.153
Spring/Summer TMAX	0.057	0.207	0.017	0.406	0.026	0.441
Yearly PPT	0.031	0.123	0.054	0.031 *	0.053	0.061
Yearly TMAX	0.114	0.038 *	0.064	0.126	0.049	0.247
MEDIUM SCALE						
Spring/Summer PPT	0.202	0.143	0.016	0.693	0.0002	0.943
Spring/Summer TMAX	0.083	0.421	0.020	0.649	0.021	0.524
Yearly PPT	0.016	0.474	0.026	0.312	0.000	0.943
Yearly TMAX	0.002	0.844	0.206	0.161	0.104	0.398
FINE SCALE						
Spring/Summer PPT	0.249	0.667	0.249	0.667	-	-
Spring/Summer TMAX	0.083	0.421	0.057	0.150	-	-
Yearly PPT	0.293	0.636	0.289	0.639	0.028	0.786
Yearly TMAX	0.240	0.667	0.240	0.667	0.606	0.221

* Significant at alpha = 0.05

Table 3.4. Average seasonal maximum temperature and the average precipitation for two time periods: 1960-1982 and 1983-2003.

SCALE: **BROAD**

	Max. Temperature (°C)		Difference (°C)	Precipitation (mm.)		Difference (mm.)
	1960 – 1982	**1983 - 2003**		**1960 – 1982**	**1983 - 2003**	
Spring	43.7	45.0	1.4	40.6	45.7	5.1
Summer	74.4	76.5	1.1	30.5	27.9	- 2.6
Autumn	66.6	67.9	1.3	33.0	27.9	- 5.1
Winter	40.1	39.5	- 0.6	53.3	53.3	0.0
Annual	57.5	58.4	0.9	38.1	38.1	0.0

evaporation and transpiration of shallow-rooted plants as the season progresses (Caldwell *et al*. 1977). Warming is very likely to persist over the next several decades regardless of the effectiveness of greenhouse gas emissions reduction efforts, due to greenhouse gas emissions that have already occurred. Warmer, drier sagebrush communities such as Wyoming big sagebrush may be more susceptible to cheatgrass invasion as most often reported for the Wyoming big sagebrush subspecies in the past (Stewart and Hull 1949).

At the broadest scale, average annual maximum temperature the year of the fire event was weakly positively correlated with fire size ($r^2 = 0.114$, $P = 0.038$) (**Table 3.3**). These effects were not observed at finer spatial scales. Factors such as temperature, humidity, fuel moisture, species composition, and topographic relief, usually measured at relatively fine spatial scales such as hectare, can exhibit substantial spatial and temporal variability (Turner and Romme 1994). The kinds and amounts of fuel vary considerably across the landscape, and each cover type may have a characteristic fuel mosaic (Brown 1985). Air temperature, humidity, and wind also have been shown to vary in relation to elevation and topography in mountainous areas (Mitchell 1976). Finally, the probability of ignition may vary spatially (Turner and Romme 1994).

Using the gradient analysis, only 1.8% of the study region for the big sagebrush probability analysis had no data available. For the remaining area in which data were available, 22.7% of the study region with a probability of occurrence in the range of 0.26 – 1.0 for current climatic conditions will experience a 25-50% decrease in the probability that the species will occur under a doubling of CO_2. Although many climate models provide valuable insight into potential species ranges in novel, stable climates, they do not take into account several variables that will influence the spread and distribution of a species, such as the means by which species disperse, interactions with new assemblages of species, and the direct and indirect responses of the species to other aspects of climate and environmental change. The sagebrush loss may increasingly be replaced by annual grasses such as cheatgrass, which have been identified as the C_3 species most positively responsive to increased CO_2 and thus capable of benefiting from global atmospheric trends (Smith 2000, Smith *et. al.* 1987). Interactions among the elements of global change might affect the prevalence of biological invaders, but remain largely unstudied. Seydack et al. (2007) asserted that for certain vegetation types in low-lying elevations in South African shrub land, fuel characteristics appeared to be the dominant factor controlling fire frequency. If climatic changes result in extended and more severe droughts, total annual grass production may be reduced, though larger fires may continue to occur and even increase across the landscape due to lower fuel moisture and connectivity of fine fuels, provided ignition sources were not limiting. This may offset the possibility of an increase in precipitation, flammability potential, or it may accelerate the conversion to more drought and/or fire- sensitive land cover, particularly at the regional scale.

Temperature, precipitation, vegetation, and albedo (extent to which light from the sun is reflected) interact in determining available soil moisture, which is variable seasonally and geographically (Mitchell 1989). Increasing temperatures can increase evapotranspiration and reduce the growing season by depleting soil moisture sooner. Sagebrush steppe grown at higher temperatures appears to adapt by exhibiting maximum photosynthetic rates at higher temperatures (Adams 1970). Provided that water potential is not limiting, leaves acclimated to higher temperatures tend to have photosynthetic rates higher than those acclimated to lower temperatures (Mooney and Shropshire 1967). In one study in a montane meadow (Perfors *et al.* 2003), the growth rate of mountain big sagebrush near the edge of the high-elevation range for this species was enhanced under experimental warming. The date of snowmelt was the dominant climatic factor determining annual growth, and not soil temperature or moisture. However, temperature increases alone may be sufficient to cause productivity declines in sagebrush communities if vegetation is not permitted to recover annually through natural cycles of cooler, wetter years between warmer, dryer ones that may be detrimental to this vegetation type (Kremner *et al.* 1996). Big sagebrush is one of the most successful plant species in the Great Basin, and likely benefits most by the reoccurrence of wet-dry cycles. If an accelerated warming does occur in the Great Basin accompanied with a decrease in precipitation, particularly during winter and early spring, we can expect a decrease in dominance of sagebrush species. This was demonstrated during the drought of the 1930's, when sagebrush canopy cover experienced a decline in the cold desert (Pechanec *et al.* 1937). Elevated temperatures may benefit lower biomass communities such as cheatgrass in semi-arid systems that are able to better maintain equilibrium between productivity and soil water availability under a range of climates without substantial impacts on the hydrologic balance (Kremer *et al.* 1996).

The interactions between climate and vegetation and fire are complex. The degree to which recent vegetation changes can be attributed to climatic variability is compounded by anthropogenic-influenced change such as non-native species introduction, overgrazing and fire suppression. Although overgrazing and agricultural practices might create areas dominated by fire-susceptible stands of sagebrush, it is predominantly fine fuels that carry fires in the Snake River Plain and surrounding areas (Whisenant 1990). Bergeron and Flannigan (1995) showed decreased fire frequencies in a southern boreal forest of southeast Canada with a doubled CO_2 scenario derived from a GCM. They attributed the reduction in fire frequencies to a reduction in drought frequency combined with an increase in precipitation and relative humidity in the doubled CO_2 climate scenario. The combination of these factors into the future is an unknown and as a result the exact effect of climate change is an unknown. More than likely the effects will vary greatly on a regional basis.

Dominance of exotic grass species in sagebrush steppe alters the fire cycle (Smith *et al.* 1987, Mayeux *et al.* 1994). If precipitation in the form of rainfall were to increase in the future, vegetation growth may be enhanced, resulting in a build-up of fuel. If a precipitation increase coincided with an increase in storms and subsequent lightning strikes, a continuous increase in large fires in this landscape can be expected.

Some evidence that extent of burning was related to climate (as determined by annual maximum temperature and previous year's annual precipitation) indicated that increased large fire occurrences were at least partially linked to relevant changes in these factors over time. With an increase in fine fuels in this lightning-prone region we may expect that large fire occurrence and size are causally linked to the sustainably combustible fuels more easily than the patchier sagebrush fuels. In a warmer, moisture climate, annual grasses and fine woody plants (i.e. sagebrush) are likely to expand as higher levels of effective moisture will favor increased productivity. This is likely to occur even under the more moderate B1 climate scenario. Sagebrush may not be able to adapt to the more extreme A2 scenario, particularly if change is rapid. However, under both scenarios, increases in grass biomass will translate to more fine flammable fuels, promoting more fire which in turn will reduce the cover of sagebrush. Fire will play a crucial role in the adjustment of semi-arid vegetation to altered precipitation regimes, be it slowing or limiting the encroachment of woody vegetation into grasslands under less dry conditions, or hastening the transition from sagebrush steppe communities to grasslands under drier conditions (Lenihan *et al.* 2003). Thus, it seems likely that cheatgrass may be a superior contender for dominance in sagebrush steppe given the B1 or A2 climate scenario. Other studies in this region have found a positive correlation between cheatgrass abundance and fire frequency (Whisenant 1990).

My results suggest that the relationship between large fires and climate may become clearer when assessed on a longer-term multi-annual temporal scale, especially in relation to the extent of burning following climatic cycles of alternating periods with relatively high temperatures (especially annual means of maximum monthly temperatures and annual mean precipitation the previous year). Many impacts on wildfire occurrence and size are moderated or even controlled by climate influences other than precipitation and temperature (i.e. cloudiness, wind or humidity), including socioeconomic development and anthropogenic influences, with climate change acting as an exacerbating factor, not necessarily the primary driver. The relationship of large wildfires to temperature and precipitation is just one part of the story.

The recovery of sagebrush steppe post fire depends in part on species composition and climatic conditions after the burn, and is more rapid in areas receiving higher precipitation if other conditions for recovery remain optimal (Paysen *et al.* 2000). In one study in which vegetation and hydrologic responses to GCM climate changes were simulated (Kremer *et al.* 1996), both sagebrush and cheatgrass showed an increase in leaf area with increased precipitation and subsequent water use efficiency (WUE) and a decrease in leaf area with elevated temperature and decreased precipitation. My results suggest that 'optimal' conditions for sagebrush recovery after fire is increasingly less likely to be met as fire interval decreases and an increasing number of fires occur over time across the landscape with increasing area burned per fire.

Climate and Plant Physiology

Carbon Dioxide

Plant responses to increases in CO_2 depend primarily on photosynthetic pathways. Cool season (C_3 pathway) plants such as cheatgrass and sagebrush will likely have an advantage over warm season (C_4 pathway) and Crassulacean Acid Metabolism (CAM) plants, as C_3 plants are not saturated with carbon. In a mixed community of C_3 and C_4 plants, C_3 species will likely benefit most from an increase in CO_2 concentration, particularly if water is not a limiting factor (McDonald 2003). The primary CO_2 effect on C_3 species is on carbon assimilation, whereas C_4 plants it is on transpiration (Goudriaan and Unsworth 1990). Increases in atmospheric CO_2 should improve WUE of plants, and increase photosynthesis and plant growth (Bazzaz 1990). Although some C_4 and CAM plants do show increased growth or physiologic responses with elevated CO_2, C_3 plants tend toward greater responses initially (Poorter 1993). However, long-term growth at elevated CO_2 results in many C_3 species result in photosynthetic rates that are only marginally enhanced, or even lower than observed prior to elevated CO_2 exposure (Sage 1994). While over the short-term, CO_2 enhancement generally stimulates net CO_2

fixation, long-term exposure is often inhibitory (Sasek *et al.* 1985). The nature of this inhibition is not known for certain, though reduced net CO_2 fixation may be a sign of reduced rubisco activation states which suggests that the mechanism by which the plants photosynthesize is unable to fully acclimate to growth at high CO_2 (Sage *et al.* 1989), or it may be due to a reallocation of nitrogen to processes other than photosynthesis (Field and Mooney 1986). Invasive species generally exhibit positive growth responses to elevated CO_2 (Dukes 2000). Spring drought may be more critical to carbon dynamics than summer drought in the sagebrush-steppe ecosystem (Fay *et al.* 2003, Gilmanov *et al.* 2006).

An increase in CO_2 might also have contributed to the diversion of fire regimes from historical regimes and contributed to the invasion by cheatgrass of the intermountain West (Smith *et al.* 1987, Mayeux *et al.* 1994). Because elevated CO_2 levels stimulate plant growth, it consequently increases fuel loading as well as cause plant leaves and stems to be less sensitive to decay, allowing dead fuels to persist longer (Ziska *et al.* 2005). Under the right conditions, the frequency and severity of fires may increase because of more rapid fuel loading resulting in part by an increase in plant growth due to increased CO_2 levels (Sage *et al.* 1989). Smith (*et al.* 2000) found that an increase in size of individual cheatgrass plants grown under elevated CO_2 also resulted in higher seed production, but with decreased seed mass. Smith (*et al.* 2000) suggested that the increase in seed production may compensate for reduced seed quality. The demographics of some tree and shrub expansion into arid environments offer strong evidence that rising CO_2 contributes to woody plant encroachment in arid ecosystems (Polley *et al.* 1996, 1999, 2002).

Water Use Efficiency and Photosynthesis

The availability of moisture can affect water use efficiency (WUE). WUE refers to the ratio of net primary production (NPP) to evapotranspiration, or amount of carbon acquired per unit of water lost, as water can be conserved through stomatal closure in terrestrial plants, but only at the expense of reduced carbon gain (Toft *et al*. 1989). When moisture is abundant, many plants have high stomatal conductance to maximize carbon gain. As water becomes scarce, stomatal closure may be induced to prevent excessive loss of water by transpiration in terms of atmospheric drought, as determined by temperature and the relative air humidity (Mott and Parkhurst 1991). This would result in a reduction of net ecosystem exchange (NEE), increased WUE (Forseth and Ehleringer 1983; Comstock and Ehleringer 1984; Gollan *et al.* 1985), and decreased photosynthetic rates (Dai *et al*. 1992).

Cheatgrass may be able to maintain biomass at greater soil water deficits than sagebrush communities when WUE is enhanced. In the sagebrush–steppe region, soil temperature influences green up in the spring, whereas soil moisture and absolute humidity deficits influence senescence in the summer (Kremer *et al.* 1996). In a study in northern Utah, big sagebrush photosynthesis was highest in late May/early June during periods of maximum vegetative growth and lowest in August, the driest period (DePuit and Caldwell 1973). Like most plants, sagebrush reduces its overall growth during drought. However, sagebrush and bunchgrass communities, which generally have greater WUE (Toft *et al*. 1989), support greater living biomass at higher soil water deficits than cheatgrass (Kremer & Running 1993). Sagebrush has a lower rate of transpiration than cheatgrass because of its greater living biomass (Donovan & Ehleringer 1992) and greater respiration costs. Sagebrush communities may experience a reduction in photosynthesis if faced with greater absolute humidity deficits and greater maintenance respiration costs (Kremer *et al.* 1996). If water stress is nominal, variations in the rate of photosynthesis may be attributed to irradiation and temperature for sagebrush (DePuit and Caldwell 1973). While cheatgrass thrives under conditions of abundant moisture, it also grows well in areas of low precipitation (Paysen *et al.* 2000). After disturbance and during years with higher precipitation, annual grasses such as cheatgrass and medusahead often invade sagebrush steppe and outcompete native plants. Warmer, drier summers will make sagebrush highly vulnerable to fire and insect outbreaks (Parmesan 2006). Increases in insect outbreaks have already changed the vegetation in much of the arid western U.S. (Hobbie *et al.* 1999). In addition to the drying effects of increased temperatures, projected decreases in summer precipitation and increased fuels resulting from CO_2 fertilization (plant growth as a result of increased atmospheric CO_2) will further increase the trends towards more and larger fires over the next century (Price and Rind 1994; Lenihan *et al.* 1998).

Climate Change and Migration

Sagebrush steppe is often a mixture of other shrub species with an understory of perennial grasses and forbs. At higher elevations, big sagebrush often forms the understory with conifers. The species of sagebrush dominating a particular site depends on topography, soil arrangement, and moisture prevalent at the local scale. Low sagebrush prefers shallow soils and restricted drainage; silver sagebrush prefers soils saturated with moisture in the spring; black sagebrush is found in gravelly soils. At lower elevations and on drier sites, saltbrush (*Atriplex polycarpa*), greasewood (*Sarcobatus vermiculatus)*, creosotebush (*Larrea tridentata)*, and winterfat *(Ceratoides lanata*) are more prevalent in sagebrush steppe, whereas at mid-elevations on more mesic sites (areas with moderate moisture), bitterbrush *(Purshia tridentata)* and curlleaf mountain mahogany *(Cercocarpus ledifolius)* become more prevalent. Cheatgrass establishes dominance within a couple of years if fires occur only a few years apart, and excludes many common shrubs such as and big sagebrush, antelope bitterbrush, and cliffrose *(Cowania mexicana)*. If fires are slightly less frequent, rabbitbrush *(Chrysothamnus* and *Ericameria spp)*, a resprouting perennial, may coexist with cheatgrass (Knapp 1996). In one study in northern Nevada, cheatgrass adversely affected the productivity and water status of green rabbitbrush *(Chrysothamnus viscidiflorus)* and needle-and-thread grass because the root system of cheatgrass monopolized soil resources over the native species (Melzoga & Nowak 1991). Until the early 1980s, cheatgrass was not considered an important species within the Great Basin shadscale zone. The accumulation of cheatgrass phytomass has allowed fires to burn in areas previously unburned, fostering an increase in cheatgrass and other exotic annuals such as Russian thistle *(Salsola australis)* and halogeton *(Halogeton glomeratus)* at the expense of the native shrubs (Young and Tipton 1990). If temperatures continue to increase, woody species (i.e. trees) currently restricted by cooler temperatures may be restricted to higher elevation areas of the Great Basin. If precipitation increases, woody species will likely expand in range, increasing fuel loads. The increase in biomass from heavy woody species will very likely increase fires, further disseminating sagebrush steppe vegetation.

A common misconception in future scenarios of the response of vegetation to climatic changes is that zones of vegetation will advance in a northerly direction and retreat southward as climate warms and/ or cools from current baseline levels. Each species' which composes the larger group of vegetation (i.e. sagebrush steppe) has a unique migratory history (Pielou 1991). Rates of range shifts vary greatly among and within species in part due to different dispersal abilities. The ability and speed at which individual plants migrate is largely species-specific and depends on a multitude of factors including environmental conditions suitable for advancement, age of reproductive maturity, dispersal mechanism, seed weight and wind speed. New arrivals would have to take advantage of gaps in existing vegetation matrices. In addition, the few seeds which do arrive to a new location will likely are in the company of a larger seed base for adjacent vegetation which is already abundant in the area, even if the climate is gradually becoming more suitable to the new arrivals than already established vegetation (Pielou 1991). If large gaps in existing vegetation become available as a result of a large fire, then the newcomers may have a better opportunity to establish, particularly if existing adjacent vegetation lacks a mechanism of dispersal to occupy the gaps. Migrating over long distances requires plant dispersal followed by the establishment and spread of local populations (Clark *et al.* 2001). The chance of a random seed establishing itself is dependent on the existing vegetation and on the type and frequency of disturbances at the local scale. Newly originated outlier populations may be repeatedly unsuccessful, and if they do succeed, there may be a lengthy time period before the originating populations can populate gaps with propagules. For this reason, a species' ability to shift its range may be opportunistic (Pitelka 1997). Altered, fragmented lands resulting from anthropogenic disturbance can also affect migration rates by affecting local population sizes and seed production, creating impassable barriers and by altering and even eliminating the amount of suitable habitat for new populations. This may increase the chance for local extinctions (Fahrig 2003), particularly for species exhibiting low adaptability and capacity for sufficient dispersal. If climate change renders these vestige sites inhospitable, there may not be sufficient numbers of plants left to initiate migrations across hostile landscapes. Given that anthropogenic activity hastens the migration of some species and hinders that of others, we can only expect that, in the future, this reordering would result in communities quite unlike those of today.

Another potential obstacle to northerly or southerly migrations is the length of the photoperiod, or the length of daylight with relation to the length of night (Pielou 1991). This has implications for the migration of plants in response to climate change because the photoperiod is shorter at lower latitudes, which has implications for germination of plants which are accustomed to higher (northerly) latitudes. Seedling mortality and recruitment are closely related to the timing of germination. Meyer et al. (1990) reported that emergence patterns of big sagebrush were correlated with germination response, and seeds planted in habitats most similar to that of the parent population experienced higher emergence and survival.

Organisms, populations and ecological communities respond to regional changes which are spatially heterogeneous, as opposed to changes in global averages (Walther *et al.* 2002). Factors affecting species distribution interact in intricate ways, and shifts in range are often sporadic rather than gradual, and changes may be accelerated during warmer periods. Many species, such as cheatgrass, exhibit first an inactive phase during the invasion process, during which ranges shift very little, followed by an active phase, during which something sparks off a sizeable expansion (Forcella and Harvey 1981). The large fire – annual grass phenomenon is more than likely a positive feedback cycle (Evans and Young 1970). Predictions of species' response in these complex systems is complicated by the variability of temperature and precipitation over the past century, although directional changes in these climatic elements have produced consistent ecological responses in many systems, which often only become apparent when viewed at broader spatiotemporal scales. My results show that only at the broadest spatial scale did patterns of large fire occurrences become clearer when correlated with climatic variables. The finest spatial scale did not display any patterns in fire history due to the small sample size, and patterns only became clearer at the broadest spatial scale.

There exists a challenge of separating the effects of changes in climate from other effects. One of the great challenges of understanding climate change impacts is that these changes are superimposed on an already rapidly changing world. In many instances, ecosystems may respond to a variety of stressors acting upon them simultaneously rather than just one. We know that sagebrush ecosystems have been and will continue to undergo change. In many areas we will likely continue to see an increase in large wildfires and more non-native plant species. Given the anticipated rate of change, it is unlikely that all existing systems will just migrate from one place to another. Not all seeds of a species will migrate at the same time or at the same speed, and not all species comprising a zone of vegetation will migrate at the same time or at the same speed. Some species may not migrate at all. Species at lower elevations and more southerly ranges may shift northward more rapidly than the resident species will shift to lower elevations or southward. This may initially lead to an increase in species richness in a region. Some of the area currently classified as sagebrush steppe may instead be completely new communities of tree and plant species together with different insects and pathogens. If sagebrush does remain on a particular site, similar functional types of insects and pathogens will likely remain, although they may be different species better-adapted to the future climate, which would perform a similar functional role. In some areas, sagebrush as it exists today may continue to exist, while in others, it may disappear. Although a particular subspecies such as basin big sagebrush might disappear from some locations, it might become more widely distributed in another part of its geographic range under the future climate. For example, basin big sagebrush may occupy more of the current mountain big sagebrush habitat. If species comprising sagebrush steppe are unable to persist in their current range, they will be replaced by species to which the system is suitable. Although sagebrush species may grow in warmer and colder areas, subspecies that grow in the colder areas (i.e. mountain big sagebrush) may not be genetically capable of surviving warmer weather. It is important to realize the climate is not the only factor determining suitable habitat and eventual success. Insects, pathogens and other species may immigrate from outside the system and become established, or synchronous relationships between seed fungi and their hosts may be disturbed by climate change. Cheatgrass will probably continue to do very well. The results presented here support the conclusion that fire seasons in the Snake River Plain and Northern Basin and Range will likely continue to become longer, seeing more large fires under both climate scenarios discussed here.

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Appendix 1. Select Biophysical (BpS) Vegetation Descriptions

Sagebrush Steppe
Scientific Name: Inter-Mountain Basins Big Sagebrush Shrubland
Summary: *Atriplex* spp. may be present in some stands. *Ericameria nauseosa, Chrysothamnus viscidiflorus, Purshia tridentata* (not commonly in Montana or Wyoming), or *Symphoricarpos oreophilus* may codominate disturbed stands (e.g., in burned stands, these may become more predominant). Perennial herbaceous components typically contribute less than 25% vegetative cover. Common graminoid species can include *Achnatherum hymenoides*, *Bouteloua gracilis, Elymus lanceolatus, Festuca idahoensis* (not in Montana or Wyoming), *Hesperostipa comata, Leymus cinereus, Pleuraphis jamesii* (not present in northeastern portions of the range), *Pascopyrum smithii, Poa secunda*, or *Pseudoroegneria spicata* (not in Wyoming). Some semi-natural communities are included that often originate on abandoned agricultural land or on other disturbed sites. In these locations, *Bromus tectorum* or other annual bromes and invasive weeds can be abundant. Most *Artemisia tridentata* ssp. *wyomingensis* communities in Wyoming are placed in Inter-Mountain Basins Big Sagebrush Steppe (CES304.778); the shrubland system is more restricted in environmental setting than the steppe. Dunes in the Red Desert have areas of large basin big sage with very dense canopies. In Wyoming, this system is likely to only contain *Artemisia tridentata* ssp. *Tridentate*

Desert Scrub
Scientific Name: Inter-Mountain Basins Mixed Salt Desert Scrub
This extensive ecological system includes open-canopied shrublands of typically saline basins, alluvial slopes and plains across the Intermountain western U.S. This type also extends in limited distribution into the southern Great Plains. Substrates are often saline and calcareous, medium- to fine-textured, alkaline soils, but include some coarser-textured soils. The vegetation is characterized by a typically open to moderately dense shrubland composed of one or more *Atriplex* species, such as *Atriplex confertifolia, Atriplex canescens, Atriplex polycarpa*, or *Atriplex spinifera*. *Grayia spinosa* tends to occur on coppice dunes that may have a silty component to them. Northern occurrences lack *Atriplex* species and are typically dominated by *Grayia spinosa, Krascheninnikovia lanata*, and/or *Artemisia tridentata*. Other shrubs present to codominant may include *Artemisia tridentata* ssp. *wyomingensis, Chrysothamnus viscidiflorus, Ericameria nauseosa, Ephedra nevadensis, Grayia spinosa, Krascheninnikovia lanata, Lycium* spp., *Picrothamnus desertorum*, or *Tetradymia* spp. In Wyoming, occurrences are typically a mix of *Atriplex confertifolia, Grayia spinosa, Artemisia tridentata* ssp. *wyomingensis, Sarcobatus vermiculatus, Krascheninnikovia lanata*, and various *Ericameria* or *Chrysothamnus* species. Some places are a mix of *Atriplex confertifolia* and *Artemisia tridentata* ssp. *wyomingensis*. In the Great Basin, *Sarcobatus vermiculatus* is generally absent but, if present, does not codominate. The herbaceous layer varies from sparse to moderately dense and is dominated by perennial graminoids such as *Achnatherum hymenoides, Bouteloua gracilis, Elymus lanceolatus* ssp. *lanceolatus, Pascopyrum smithii, Pleuraphis jamesii, Pleuraphis rigida, Poa secunda*, or *Sporobolus airoides*. Various forbs are also present

Scientific Name: Sonora-Mojave Mixed Salt Desert Scrub
Summary: This system includes extensive open-canopied shrublands of typically saline basins in the Mojave and Sonoran deserts. Stands often occur around playas and spatial pattern is in large patches. Substrates are generally fine-textured, saline soils. Vegetation is typically composed of one or more *Atriplex* species such as *Atriplex canescens* or *Atriplex polycarpa* along with other species of *Atriplex*. Species of *Allenrolfea, Salicornia, Suaeda*, or other halophytic plants are often present to codominant. Graminoid species may include *Sporobolus airoides* or *Distichlis spicata* at varying densities.

Scientific Name: Chihuahuan Mixed Salt Desert Scrub
Summary: This ecological system includes extensive open-canopied shrublands of typically saline basins in the Chihuahuan Desert. Stands often occur on alluvial flats and around playas, as well as in floodplains along the Rio Grande and Pecos rivers, possibly also extending into the San Simon of Southeastern Arizona. Spatial pattern is in large patches and substrates are generally fine-textured, saline soils. Vegetation is typically composed of one or more *Atriplex* species such as *Atriplex canescens*, *Atriplex obovata*, or *Atriplex polycarpa* along with species of *Allenrolfea, Flourensia, Salicornia, Suaeda*, or other halophytic plants. Graminoid species may include *Sporobolus airoides, Pleuraphis mutica*, or *Distichlis spicata* at varying densities.

Scientific Name: Sonora-Mojave Creosotebush-White Bursage Desert Scrub
Summary: This ecological system forms the vegetation matrix in broad valleys, lower bajadas, plains and low hills in the Mojave and lower Sonoran deserts. This desert scrub is characterized by a sparse to moderately dense layer (2-50% cover) of xeromorphic microphyllous and broad-leaved shrubs. *Larrea tridentata* and *Ambrosia dumosa* are typically dominants, but many different shrubs, dwarf-shrubs, and cacti may codominate or form typically sparse understories. Associated species may include *Atriplex canescens, Atriplex hymenelytra, Encelia farinosa, Ephedra nevadensis, Fouquieria splendens, Lycium andersonii*, and *Opuntia basilaris*. The herbaceous layer is typically sparse, but may be seasonally abundant with ephemerals. Herbaceous species such as *Chamaesyce* spp., *Eriogonum inflatum, Dasyochloa pulchella, Aristida* spp., *Cryptantha* spp., *Nama* spp., and *Phacelia* spp. are common.

Appendix 2. Size class 'D' fires by size (ha) and year

YEAR	AREA (Ha)	YEAR	AREA (Ha)	YEAR	AREA (Ha)
1960	10244	1981	5354	1981	7252
1960	7094	1982	6555	1982	6503
1962	6457	1982	5142	1982	6124
1963	16589	1983	10580	1983	6025
1963	9512	1983	10077	1986	5054
1963	8516	1983	7922	1987	25458
1966	7781	1983	6628	1987	9634
1966	6507	1983	6595	1987	7721
1969	7623	1983	6126	1990	29112
1969	5256	1983	6037	1990	6301
1970	5854	1983	5646	1990	5856
1970	5831	1984	13597	1992	60630
1971	35791	1984	8831	1992	16638
1971	29723	1984	8716	1992	14028
1971	14224	1984	8323	1992	7915
1971	11308	1984	5286	1992	7468
1971	8775	1985	24076	1992	6258
1971	7563	1985	21999	1994	9205
1972	5835	1985	14632	1994	7527
1973	16255	1985	12845	1994	6865
1973	14032	1985	10094	1994	6437
1973	5592	1985	8771	1994	5280
1974	17834	1985	7796	1995	24763
1974	6720	1985	7600	1995	12465
1974	5886	1985	7599	1995	8294
1975	5043	1985	7535	1995	7422
1976	59498	1985	6447	1995	6378
1979	5287	1985	5602	1995	5684
1980	6416	1985	5204	1995	5393
1980	6416	1985	5119	1996	88862
1981	30048	1986	21595	1996	72048
1981	28627	1986	21444	1996	22727
1981	18284	1986	21197	1996	14670
1981	17666	1986	21158	1996	13373
1981	15212	1986	13407	1996	10555
1981	14278	1986	12548	1996	9928
1981	8615	1986	11047	1996	9805
1981	8484	1986	10197	1996	9765
1981	8045	1986	9339	1996	9531
1981	7669	1986	9108	1996	9250
1981	6738	1986	7430	1996	8975

1996	8009	2000	23064	2000	8201
1996	6996	2000	22328	2000	7098
1996	6465	2000	22173	2000	7094
1996	6037	2000	21889	2000	7092
1996	5299	2000	21888	2000	7076
1996	5232	2000	21325	2000	7010
1998	9308	2000	18291	2000	6900
1998	9307	2000	16343	2000	6728
1998	7738	2000	16245	2000	6681
1998	7570	2000	16143	2000	6656
1999	56534	2000	16086	2000	6637
1999	46824	2000	15980	2000	6628
1999	20127	2000	15977	2000	6592
1999	20123	2000	15956	2000	6529
1999	17538	2000	14726	2000	6113
1999	17496	2000	14713	2000	6041
1999	16623	2000	14396	2000	5939
1999	15827	2000	14277	2000	5755
1999	14184	2000	14209	2000	5675
1999	13202	2000	13946	2000	5176
1999	13201	2000	12735	2001	32585
1999	13198	2000	12601	2001	32583
1999	13142	2000	12562	2001	27237
1999	13122	2000	12412	2001	17420
1999	12857	2000	12409	2001	16941
1999	12838	2000	10901	2001	16814
1999	9552	2000	10745	2001	16807
1999	9551	2000	10740	2001	14603
1999	8622	2000	9491	2001	13098
1999	6580	2000	9488	2001	7803
1999	6579	2000	9092	2001	6255
1999	6064	2000	8611	2001	6056
2000	29461	2000	8375	2001	6007
2000	29448	2000	8334	2001	16941
2000	23263	2000	8208	2002	17263
2000	23105	2000	8207	-	-