

THESIS

HERBACEOUS AND AVIFAUNA RESPONSES TO PRESCRIBED FIRE AND  
GRAZING TIMING IN A HIGH-ELEVATION SAGEBRUSH ECOSYSTEM

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## ABSTRACT

### HERBACEOUS AND AVIFAUNA RESPONSES TO PRESCRIBED FIRE AND GRAZING TIMING IN A HIGH-ELEVATION SAGEBRUSH ECOSYSTEM

Changes in land use over the last two centuries have been linked to reduced geographic distributions of sagebrush (*Artemisia spp.*) habitats and sagebrush associated avifauna. Livestock grazing is one of the principle land uses of publicly administered sagebrush ecosystems. Prescribed fire and other sagebrush control methods are often implemented in an attempt to increase the quantity or quality of available livestock forage. These treatments have also been recommended by some as a tool for enhancing habitat to meet seasonal forage requirements for greater sage-grouse (*Centrocercus urophasianus*) or other wildlife species. In this thesis, I examine differences in: 1) herbaceous productivity (peak standing crop biomass), 2) relative habitat use by sage-grouse, and 3) habitat suitability for migratory songbirds related to prescribed fire and summer grazing timing treatments in a high-elevation sagebrush community. Increased livestock forage availability in burns occurred only during one of three post-burn years investigated and was further limited to only one of three grazing treatment pastures (early summer). Graminoid peak standing crop in burn treatments with later summer grazing never surpassed unburned big sagebrush plots subjected to the same grazing treatment. Habitat suitability and use by avian species appeared to be largely unaffected by post-fire

grazing timing. Although sage-grouse use of burn treatments was greater when burn configuration was more heterogeneous, use was minimal across all burn treatments the first four years after burning. Sagebrush obligate songbirds, such as Brewer's sparrow (*Spizella breweri*) and sage thrasher (*Oreoscoptes montanus*), also strongly avoided burn treatments, particularly with increasing distance to intact big sagebrush (*A. tridentata*) nesting substrate. Although ground nesting species, such as vesper sparrow (*Pooecetes gramineus*), preferred reduced shrub cover associated with burn treatments, this species also responded negatively to more uniform patterns of big sagebrush removal. These results suggest that avian species are minimally impacted by summer livestock grazing at the light to moderate intensity levels resulting from my grazing treatments, regardless of timing. However, sage-grouse and migratory songbirds displayed clear seasonal avoidance of burn treatments. These results demonstrate that negative avifauna responses to sagebrush removal may strongly outweigh limited short-term gains in livestock forage production resulting from prescribed fire in some high-elevation big sagebrush systems.

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## **CHAPTER 1 : INTRODUCTION TO THE THESIS**

### **1.1 INTRODUCTION**

Sagebrush steppe is one of the most imperiled ecosystems in the western United States (Noss and Peters 1995, Mac et al. 1998, Anderson and Inouye 2001) as a direct result of region-wide changes in land-use practices over the past two centuries. These changes have directly and indirectly resulted in loss of sagebrush habitat and degradation of proper ecological function in many remaining sagebrush ecosystems. The most common land uses currently impacting sagebrush habitat include conversion to agriculture, urbanization, oil/gas development, off-road recreation, and livestock grazing (Baker et al. 1976, Noss et al. 1995, Hann et al. 1997, Crawford et al. 2004). Direct and indirect hazards associated with these activities include, exotic species introduction, habitat fragmentation, sagebrush control/removal, and shifts in vegetation species composition (McArthur and Plummer 1978, Young and Evans 1978, Braun 1998, Christensen et al. 1996, Knick 1999, Miller and Eddleman 2000).

Sagebrush rangelands managed by state and federal agencies encompass approximately 70% of all remaining sagebrush habitats in the Intermountain West (Knick et al. 2003). Livestock grazing has been one of the most consistent land-uses of publicly managed sagebrush ecosystems since just after the Civil War (Clawson 1983). Over-grazing on public and private rangelands prior to the 1920's resulted in dramatic changes in vegetation composition and soil erosion across the western United States (Borman and

Johnson 1990). Rangelands began to recover with changes in grazing management practices that followed the Taylor Grazing Act of 1934. However, because management of a large proportion of remaining sagebrush habitat is administered by public agencies, understanding the impact of current grazing practices on sagebrush steppe vegetation and associated fauna is of paramount importance for conserving remaining sagebrush habitats and the wildlife species that they support.

In the past, sagebrush control programs have gone hand in hand with grazing on public lands (Vale 1974). There has been a long held belief that competition for resources with shrubs limits herbaceous forage production within shrub dominated landscapes (Harniss and Murray 1973, Tanaka and Workman 1988, Bastian et al. 1995). Beginning in the mid-1940's extensive sagebrush removal programs using mechanical methods were implemented in an attempt to increase herbaceous production (Baker et al. 1976). Efforts to control sagebrush intensified in the late 1950's when herbicide application became the preferred method for treating large tracts of sagebrush. However, after the use of 2,4-D was banned on public lands in the early 1980's, prescribed fire became the most common large-scale method of sagebrush control (Braun 1987).

Many research efforts have assessed the response of herbaceous production after big sagebrush removal (Blaisdel 1953, Harniss and Murray 1973, Peek et al. 1979, Van Dyke et al. 1991, Peterson 1995, Wambolt et al. 2001, Van Dyke and Darragh 2006) and grazing treatments (Van Poolen and Lacey 1979, Holechek and Stephenson 1983, West et al. 1984, Vallentine 1989) with varying results. In the case of prescribed fire, these responses range from multiple years of increased production (Harniss and Murray 1973, Davies et al. 2007) to no increase in herbaceous biomass (Peek et al. 1979). Some



reported effects of growing season grazing timing treatments include increased production, with high intensity late summer grazing due to removal of residual biomass (Laycock 1967), reduced production with heavy early summer grazing (Crawford et al. 2004), and reduced residual height and cover with grazing anytime after peak standing crop production (Gregg et al. 1994). However, limited research has documented how production is affected when prescribed fire and grazing treatments are applied simultaneously (but see Bunting et al. 1998, Bruce et al. 2007, Bates et al. 2009). Additionally, most of the studies to date have been conducted within Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) communities and inferences from these studies are not necessarily transferable to communities dominated by mountain big sagebrush (*A. t.* ssp. *vaseyana*).

Prescribed fire has recently been recommended as a tool to enhance habitat for greater sage-grouse (*Centrocercus urophasianus*; Martin 1990, Pyle and Crawford 1996, McDowell 2000, Wirth and Pyke 2003). The proposed enhancement mechanism is increased production or availability of forb species used by sage-grouse chicks during the early brood-rearing period. When preferred forbs are present in the pre-burn vegetation community, prescribed fire has been documented to result in increased short-term production in at least some cases (Pyle and Crawford 1996). However, as with herbaceous community responses, effects of prescribed fire and grazing timing on sage-grouse habitat is frequently limited to conclusions drawn from research within Wyoming big sagebrush habitats. Many of these studies have inferred sage-grouse habitat quality indirectly by measuring vegetation responses but most have not directly addressed sage-

grouse use of prescribed fire areas (but see Robertson 1991, Byrne 2002, Lowe et al. 2009). Direct effects of grazing timing include sage-grouse avoidance of pastures when cattle are present during the breeding season (Holloran 1999, Lupis et al. 2006). Other potential ramifications of vegetation change associated with removal of residual herbaceous cover from late-season grazing include reduced nest site selection and success the subsequent breeding season (Gregg et al. 1994). To my knowledge no direct assessments of sage-grouse use of prescribed fire areas across different grazing timing strategies have been conducted.

Regardless of whether the goal of a prescribed fire treatment within a sagebrush community is to increase forage production for livestock or enhance sage-grouse habitat, resulting alterations in habitat characteristics are likely to impact other organisms that utilize these habitats. Several migratory songbird species use the sagebrush steppe during the summer breeding season. It is likely that populations of sagebrush obligate nesting species like Brewer's sparrow (*Spizella breweri*), and sage thrasher (*Oreoscoptes montanus*), as well as shrub obligate nesting species like green-tailed towhee (*Pipilo chlorurus*) will be disrupted by removal of required nesting substrate (big sagebrush and associated vegetation) that results from large-scale sagebrush control programs (Rotenberry and Wiens 1978, Wiens and Rotenberry 1981). However, some studies have documented little change in sagebrush obligate species densities or nest success following prescribed fire (Peterson and Best 1987). In contrast, ground nesting species such as vesper sparrow (*Pooecetes gramineus*) might be expected to increase in density following a prescribed fire (Peterson and Best 1987). It has also been documented that

livestock grazing can indirectly alter avian species composition (Ryder 1980, Bock et al. 1993) by modifying plant species composition, herbaceous biomass, and vegetation structural components such as plant height and cover (Holecheck 1989). Songbirds are often highly responsive to alterations in structural habitat components (Cody 1985). Drastic changes in structural components that can follow overgrazing events, including reductions in foraging resources such as seeds and insects (Ryder 1980, Putnam et al. 1989) and removal of adequate nesting cover, have been documented to reduce nest success for numerous avian species (Koerth et al. 1983, Barker et al. 1990, Bowen 1993). Thus, differences in herbaceous response to alternate grazing timing strategies following a prescribed burn are likely to translate into associated changes in both bird densities and reproductive success (Castrale 1982, Kerley and Anderson 1995). Again, little research has investigated how the timing of grazing after prescribed fire might affect songbird densities and nest success within mountain big sagebrush communities.

In this thesis, I examine the impacts of prescribed fire and grazing timing treatments on select ecosystem components within a high-elevation sagebrush system. In Chapter 2, I assess direct responses in herbaceous productivity as measured by dry weight of graminoid and forb functional classes. I present an indirect assessment of sage-grouse relative habitat selection related to fire and grazing treatments in Chapter 3. I indirectly evaluated habitat use (selection) by sage-grouse using the abundance of sage-grouse fecal pellets detected along permanent survey routes. Finally, in Chapter 4, I present breeding songbird habitat selection responses represented by population densities estimated using line-transect sampling and reproductive success estimated through monitoring of

individual nests. In each of the avian response analyses described above, response metrics were related to differences in habitat composition across multiple spatial scales, as well as categorical representations of grazing and burn treatment type.

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## **CHAPTER 2 : HERBACEOUS BIOMASS PRODUCTION IN RESPONSE TO PRESCRIBED FIRE AND GRAZING TIMING TREATMENTS IN HIGH- ELEVATION SAGEBRUSH**

### **2.1 INTRODUCTION**

Livestock grazing has been a consistent use of public lands in the western United States since the end of the Civil War (Clawson 1983). Nearly three-quarters of the 364 million acres of publicly managed land in the Intermountain West is accessible to livestock grazing (Hess and Holechek 1995). A large proportion of this land is sagebrush habitat (57 million acres, Connelly et al. 2004) administered by the Bureau of Land Management (BLM). Prior to the Taylor Grazing Act (TGA) of 1934, the health of many rangeland systems was negatively impacted by overgrazing. After enactment of the TGA, public land management agencies began to limit grazing practices that intensified soil erosion and diminished herbaceous productivity within public rangelands (Borman and Johnson 1990). Management objectives further shifted towards improving rangeland health in the late 1970's with the Federal Land Policy and Management Act of 1976, and the Public Rangelands Improvement Act of 1978. These new management objectives sought to reverse effects of fire suppression and/or overgrazing which had altered vegetation community composition resulting, in many cases, in either a monoculture of mature sagebrush stands and/or high proportions of shrub decadence (> 25% dead branches). However, the most common objective of range restoration projects

during this early period was to increase herbaceous production (Bunting et al. 1987) by removing shrubs and releasing herbaceous vegetation from competition with woody vegetation (Clark and Starkey 1990). Herbicide treatment was the most common sagebrush removal method for large-scale projects prior to the ban of 2,4-D on public lands in the early 1980's (Braun 1987). Since that time, burning has become the favored method for woody vegetation control in sagebrush ecosystems (Braun 1987). Prescribed fire remains a widely used management tool across many publicly administered sagebrush rangelands used to achieve various management goals. Some of these targeted outcomes include halting conifer encroachment (Miller and Rose 1999), enhancing habitat characteristics for greater sage-grouse (*Centrocercus urophasianus*) or other native wildlife species (Pyle and Crawford 1996), and increasing forage availability/quality to enhance grazing for domestic livestock (Laycock 1979).

The effects of grazing intensity, rotational grazing, and grazing exclusion on the composition and productivity of the herbaceous plant community have been studied extensively (Van Poolen and Lacey 1979, Holechek and Stephenson 1983, West et al. 1984, Vallentine 1989). One such study indicated that excluding grazing for 25 years can result in an increase in basal cover of perennial grasses up to 5.8% (Anderson and Holte 1981), while another found no increase in herbaceous standing crop with 13 years of grazing rest (West et al. 1984). Van Poolen and Lacey (1979) presented a review of grazing intensity effects on herbaceous production in sagebrush systems and suggested that herbage production could be increased by 5 – 21% above production in ungrazed areas with grazing systems implemented at moderate (40 – 60% offtake) use levels. Other research suggests that late spring/early summer grazing in sagebrush systems may

negatively impact yield of late-seral grasses and forbs while benefiting early-seral species such as the invasive cheatgrass (*Bromus tectorum*; Crawford et al. 2004). However, I was unable to find any published studies comparing differences in herbaceous productivity across mid-summer, late summer, and early fall grazing timing treatments.

Numerous investigations have also examined the response of herbaceous plant community production following burns in sagebrush ecosystems (Blaisdel 1953, Harniss and Murrah 1973, Peek et al. 1979, Van Dyke et al. 1991, Peterson 1995, Wambolt et al. 2001, Van Dyke and Darragh 2006). Fire effects on herbaceous components of sagebrush systems have been shown to vary widely. Some investigators have described long-term (>9 yrs) increases in herbaceous productivity (Wambolt and Payne 1986), while others have found short-term (2 – 3 yrs) increases, no measurable increase, and even negative effects on herbaceous production resulting from fire events. Publically administered rangelands of the Intermountain West typically receive one to two years of grazing rest after fires to promote herbaceous recovery (BLM 2007). However, some BLM field offices reapply grazing the growing season immediately following fire if deemed appropriate by an environmental assessment. It has been suggested that while late-season defoliation the summer after a burn may have little effect on herbaceous recovery, intensive spring season grazing may inhibit recovery (Bunting et al. 1998). Several studies have attempted to document the effects of fire and post-fire grazing immediately after fire. Results indicate that moderate or lower grazing intensity, immediately after prescribed fire (Bruce et al. 2007) or after a single growing season of rest (Bunting et al. 1998, Bates et al. 2009) does not limit herbaceous recovery.

Inferences from many of these investigations are drawn from low or mid-elevation (< 2,000 m) Wyoming big sagebrush communities (*A. t. ssp. wyomingensis*).

I evaluated the influence of summer grazing timing without post-fire grazing deferment on herbaceous production after a prescribed burn in high-elevation (> 2,000 m) mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) vegetation community. It is not uncommon to observe lower grass production in burns the growing season immediately following a prescribed fire (Bates et al. 2009). However, fire releases nutrients contained in woody and residual herbaceous biomass, increasing nutrient availability for uptake in subsequent vegetative growth. This, along with increased ultraviolet light availability after removal of shrub over-story can result in short-term increases in herbaceous production and/or forage quality if favorable moisture conditions occur after a fire (West and Hassan 1985). For this reason, I expected short-term herbaceous production in prescribed fire treatments to increase with time since burn and surpass production levels in unburned mountain big sagebrush habitats. However, I expected this increase in production to be of limited duration (1 – 2 years), before returning to pre-burn production levels, as the surplus nutrients released by the fire become increasingly incorporated in standing biomass, litter, or removed by cattle and thus, no longer remain available to supplement normal site potentials. Further, heavy late spring/early summer grazing has been found to negatively impact recruitment of late-seral native grasses and substantially damage adult plants (Clark et al. 1998, Crawford et al. 2004), thus, I expected biomass production in burns might be lower in pastures grazed earlier in the summer compared to those grazed later in the growing season.

## 2.2 METHODS

### 2.2.1 Study Area

Prescribed fire and grazing timing treatments were conducted at the Stratton Sagebrush Ecological Research Site (Stratton) in south-central Wyoming (Figure 2.1). Stratton is managed by the Bureau of Land Management (BLM) and encompasses approximately 2,250 ha. An additional 485 ha of adjoining state land was also included in the study area. The average elevation at the site is 2,400 m with terrain characterized by gently rolling hills with ~100 m relief occurring between ridge tops and valley bottoms. A riparian drainage (Beaver Creek) runs through the valley bottom bisecting the study area. Soils are Argic Cryoborolls formed from underlying Brown's Park formation (Miocene), cross-bedded sandstone bedrock. The area averages 500 mm of precipitation annually with two-thirds resulting from winter snowfall.

Vegetation communities dominated by mountain big sagebrush with pockets of sparse bitterbrush (*Purshia tridentata*) are found on drainage slopes and other less well-drained areas. High-slope and ridgetop vegetation communities are dominated by black sagebrush (*A. nova*) with Wyoming big sagebrush as a secondary shrub. Other sparsely distributed shrubs occurring within the study site include winterfat (*Krascheninnikovia lanata*), mountain snowberry (*Symphoricarpus oreophilus*), and green rabbitbrush (*Chrysothamnus viscidiflorus*). Shrub communities have a well-developed herbaceous understory. Graminoids at the site include Idaho fescue (*Festuca idahoensis*), Rocky Mountain fescue (*F. saximontana*), needle-and-thread (*Hesperostipa comata*), and bluegrass species (*Poa* spp.). In general, forbs are a minor component of the herbaceous

understory and include flowery phlox (*Phlox multiflora*), ballhead sandwort (*Arenaria congesta*), Great Basin wild buckwheat (*Eriogonum microthecum* var. *laxiflorum*), silvery lupine (*Lupinus argenteus*), yarrow (*Achillea millifolium*), thistle (*Cirsium* spp.) and mat penstemon (*Penstemon caespitosum*).

### **2.2.2 Prescribed Fire and Grazing Timing Treatments**

Three prescribed burns were implemented in October of 2005, primarily within mountain big sagebrush vegetation communities. Fire treatments burned approximately 10 percent of the study area (295 ha). Each burn resulted in high sagebrush mortality (~85% sagebrush removal). However, high spatial variation in the shape of the fire perimeter left a large number of unburned big sagebrush patches in proximity to some burn areas while other burn areas were left devoid of any shrub cover within  $\geq 200$  m.

Fences partitioned the study site into three distinct pastures. In the three years prior to manipulation of grazing timing, which began in the summer of 2006, the grazing rotation moved from the most western pasture to eastern pasture each summer. Grazing began in early July and concluded by early August. During these years, stocking rates in the pastures that would become the late and mid-summer grazing treatments were reduced ( $0.11$  animal unit months (AUM) \*  $\text{ha}^{-1}$ ) to allow fine fuel build-up that would help carry the prescribed fire. Following the prescribed fire, the timing of cattle grazing was manipulated within each of the three pastures through three subsequent growing seasons (2006 – 2008). Each pasture was grazed for approximately a two-week period. Start dates for grazing treatments varied annually. Early summer grazing occurred in the eastern most pasture of the study site starting June 21 – July 3. The mid-summer grazing

treatment occurred in the central pasture starting July 25 – July 31 and late summer grazing occurred in the western-most pasture starting Aug 24 – Aug 28. Authorized AUMs varied between pastures based on BLM allotment tabulations of historic production and expected forage conditions within each pasture (Mike Calton, personal communication). Because expected forage availability varied by pasture, target stocking rates were 0.64 AUM \* ha<sup>-1</sup> for early grazing, 0.53 AUM \* ha<sup>-1</sup> for mid-summer grazing, and 0.60 AUM \* ha<sup>-1</sup> for the late grazing pasture.

There were no replications of treatments per se; that is, there was only one plot per unique treatment category. However, I treated the multiple measurements collected per plot each year as replicates; in the statistical sense, they are subsamples. I pursued the analysis in this fashion because each of the treatment plots occurred within isolated stands of vegetation and I adjusted for pretreatment differences with the subsequent analysis of covariance (see below). Therefore, results from my study are focused on plot-level comparisons, and I recognize that inferences beyond my study area may be limited. Sampling is often constrained by time and money for ecological studies, however, I feel results of these analyses provide valuable insights into effects that may result from burns and grazing on herbaceous production within high elevation sagebrush communities.

### **2.2.3 Vegetation Measurements**

Shrublands subjected to prescribed fire treatments were predominantly mountain big sagebrush prior to treatment. Two paired grazing exclosures (burn and unburned) were established in lower-elevation areas near the base of the drainage within each pasture prior to the grazing season in 2006. A third exclosure within each pasture was



placed in wind-swept upland habitat where black and Wyoming big sagebrush were dominant (hereafter referred to as black sagebrush). In addition to exclosures, five 1-m<sup>2</sup> grazing cages were installed in random locations within a paired plot (no exclosure) adjacent to each exclosure. In order to document baseline biomass production before experimental treatments, data was collected on vegetation characteristics for two seasons (2004, 2005) prior to the prescribed burn and installation of grazing exclosures. Prior to prescribed fire it was determined that all mountain big sagebrush plots (future burn and unburned) were similar in ground cover, shrub size, soil N, and C content (Schoenecker et al. 2005).

For the purposes of my study, herbaceous vegetation responses to treatments were evaluated using measurements of peak standing crop. Herbaceous vegetation clippings were collected within three to ten 0.25 m<sup>2</sup> circular plots frames per plot each sampling period. The number of samples varied by year due to time and manpower constraints. These vegetation measurements were collected during two pre-burn (2004, 2005) and three post-burn years (2006 – 2008). Clipping conducted in late June was used as a measure of peak standing crop and clippings collected after completion of livestock grazing were used to assess livestock grazing offtake. All herbaceous vegetation within circular plots was clipped to near-ground level (stubble height) and sorted by graminoid and forb functional groups. Vegetation was oven-dried in a forced-air oven at 55°C for 24–48 hours and weighed by functional group (Schoenecker et al. 2005).

Percent offtake was calculated from the September vegetation clipping measurement using the following formula:

$$100*(W_i - W_o)/W_i,$$

where  $W_i$  is the weight inside the grazing cage and  $W_o$  is the weight outside the cage (Bonham 1989). Cages were randomly relocated after the peak standing crop sampling in June to capture consumption and/or compensatory production after grazing and moved again after the fall clipping to avoid impacts of clipping removal on the subsequent year's measurements.

#### **2.2.4 Statistical Analysis**

All predictor variables were examined for normality and I used square-root transformations where appropriate to normalize the data. I examined potential pre-treatment differences in peak standing crop biomass, for grass (GRASS) and total herbaceous production (ALL; both grass and forb), between grazed and ungrazed control plots within individual pastures. I also tested for pre-treatment differences between all proposed burn plots, and between all proposed unburned mountain big sagebrush plots across grazing pastures using single degree-of-freedom tests after a cell means ANOVA. A cell means ANOVA model was also used to examine individual post-treatment yearly differences between burn/unburned habitats and across grazing timing treatments. Using a Null Hypothesis Significant Testing (NHST) framework, I considered plots to have significantly different production at  $P < 0.05$  (or less than 5 % probability that the difference occurred by pure chance alone), and marginally significant at  $0.05 < P < 0.10$ .

While the structuring of the experimental manipulations were straightforward other inputs into annual production, such as precipitation and site potential (i.e., pretreatment productivity), could not be controlled. For this reason I also performed an

analysis of covariance (ANCOVA) using the *lm* function of the stats package in program R (R Development Core Team, 2009) as an attempt to control for co-variation between treatment and non-experimental covariate effects on the herbaceous responses (grass or total production). My goal in performing this additional analysis was to determine if implications resulting from the rudimentary NHST analysis described above could be more appropriately examined when non-experimental covariates were addressed in the modeling process. The ANCOVA was conducted using all five years of data collection (two pre-fire and three post-fire years). While the duration of the study encompassed three post-fire years, annual grazing events took place each year after measurement of the response variable (peak standing crop). Thus, my data consists of three post fire treatment measurements but only two post-grazing treatment measurements. Treatment factors in the model were habitat treatment (HAB), grazing timing (PASTURE), and year following grazing treatment (Y1, Y2 respectively). Covariates explored in the modeling process included precipitation (FWS), and pretreatment graminoid standing crop (PRE) obtained from the 2004 growing season. Prior to model development the most predictive precipitation variable was selected by Akaike information criterion (AIC) using simple linear regression models. The candidate set of precipitation variables included individual season and combined season precipitation measurements obtained from an onsite gauge maintained by the BLM, and combined monthly precipitation measurements obtained from the nearby weather station in Saratoga (e.g., Oct. – Mar., Oct. – Apr.). After AIC selection of the precipitation variable, all models containing two-way interactions of variables and three-way interactions implicit in the study design were considered as

competing models. Final model selection was based on lowest AICc (AIC corrected for small sample sizes; Hurvich and Tsai 1989).

## **2.3 RESULTS**

### **2.3.1 Comparison of mean pretreatment conditions**

Prior to treatments, mean standing crop herbaceous biomass within enclosure plots (no grazing permitted) did not differ significantly in grass production compared to paired grazing treatment plots located just outside the enclosure. However total herbaceous production (grass and forbs) was greater outside the proposed enclosure (subsequently to be grazed) in the unburned mountain big sagebrush plot of the early summer grazing treatment pasture ( $t_{14,99} = 3.46$ ,  $P < 0.05$ ) in 2004 compared to the paired enclosure plot.

A cross-comparison of pretreatment measurements across all proposed grazing treatment plots (HabitatXPasture) indicated significant differences ( $P < 0.001$ ) in grass production ( $F_{9,126} = 170.5$ ,  $F_{9,120} = 220.21$ ) and total herbaceous production ( $F_{9,123} = 206.73$ ,  $F_{9,119} = 309.34$ ) between proposed treatment plots in each of the pre-treatment years (2004 and 2005, respectively). Cell means did not differ among the three proposed burn plots, but differences in herbaceous production were evident among the three mountain big sagebrush plots that were to remain unburned. Differences were also indicated between proposed burn plots and paired mountain big sagebrush plots (to remain unburned) within each pasture in one or both pre-burn years (Table 2.1). These results reinforced the need to control for inherent difference in productivity across sites when examining differences attributable to treatment effects.

### **2.3.2 Grazing Utilization**

Grazing utilization across the nine habitat and grazing treatment combinations varied by year. Variation in reported AUMs ranged from nine percent over target levels to 50 percent under target levels (based on reported animals and grazing duration within pastures). Mean stocking rates  $\pm$  standard deviation during grazing treatment years were below target stocking rates for the early and mid-summer grazing treatments ( $0.45 \pm .058$  and  $0.33 \pm 0.063$  AUM \* ha<sup>-1</sup>, respectively). However, annual reported stocking rates in the late grazing pasture were near target rates ( $0.52 \pm 0.16$  AUM \* ha<sup>-1</sup>). Cattle had simultaneous access to all three habitat types (burn, mountain big sagebrush, and black sagebrush) within each grazing treatment. Therefore, I was unable to differentiate within pasture use levels (stocking rate) for burns, mountain big sagebrush, or black sagebrush habitats. However, comparisons of offtake across habitat plots can give a general idea about where cattle preferentially grazed when given access to all habitat types simultaneously. Annual average consumption across treatments, as measured by percent offtake, was low to high, ranging from 0-83% with a mean of 50%. In general, offtake was lowest in black sagebrush plots and greatest within burn areas across all post-fire years, but due to high variation differences were not significant.

### **2.3.3 Herbaceous Peak Standing Crop**

#### **Comparison of ANOVA means**

A simple analysis of variance testing for treatment effects alone (habitat and grazing) on annual peak herbaceous standing crop, suggests strong effects for habitat (sagebrush type or burn), grazing treatment, and time since treatment (habitat:  $F_{2,245} =$

34.42,  $P < 0.001$ ; pasture:  $F_{2,245} = 4.56$ ,  $P < 0.05$ ; year:  $F_{2,245} = 34.50$ ,  $P < 0.001$ ). Based on two sample mean comparison tests, the first year after the prescribed fire (prior to grazing), grass and total production (grass and forbs combined) in all unexclosed mountain big sagebrush was greater than the burn treatment plot within the same pasture (Table 2.2). The same pattern occurred inside exclosures but significance was marginal in some cases (Table 2.3). Production in burn plots did not differ from each other across the three pastures. However, total production (grass and forbs) was lower in the early grazing pasture's unburned mountain big sagebrush plot compared to mid- and late grazing pastures, although in the comparison with the late grazing pasture the difference was only marginal. Within exclosures (ungrazed), all burn plots had lower production than paired unburned plots the first post-burn year, but there were no differences in production between unburned plots. However, first year production was lower in the late summer grazing pasture burn exclosure compared to other exclosed burn plots (Table 2.3). Again, measurements for this initial year following prescribed fire were collected prior to any changes in livestock grazing. Therefore, differences in production are attributable to either burn treatment or intrinsic differences in productivity between plots rather than response to grazing.

Measurements for the second post-fire year (2007) corresponded to biomass production after a single year of grazing timing manipulation. Mean grass and total herbaceous production in this year were greater in all burn plots than the previous year. Burn plot grass production remained lower than unburned mountain big sagebrush in the late grazing pasture but surpassed unburned mountain big sagebrush in early and mid-

summer grazing pastures (Table 2.2). This pattern was duplicated within ungrazed control plots (Table 2.3). Combined grass and forb productivity (ALL) in burns versus unburned mountain big sagebrush differed only in the early grazing pasture with higher production in the burned plot ( $F_{1,75} = 23.6$ ,  $P < 0.001$ ). Productivity differences were also evident within burn treatment and controls across grazing treatments. Across burn treatments, grass production was significantly lower in the late grazing pasture compared to burns in the early ( $F_{1,75} = 18.59$ ,  $P < 0.001$ ) and mid-summer ( $F_{1,75} = 10.52$ ,  $P < 0.01$ ) grazing pastures. However, grass and total herbaceous production were greater in unburned mountain big sagebrush subjected to mid-summer grazing. There were no significant differences between unburned/ungrazed control plots across pastures the second post-fire year ( $F_{1,36} < 0.4$ ,  $P > 0.5$  in all cross-comparisons).

By the third year, a simple comparison of cells means indicated that herbaceous production in burn treatments no longer differed by pasture. However, unburned mountain big sagebrush in the early grazing pasture had lower grass and/or total production than counterparts in the other two pastures. When each of the burns was compared to paired unburned sagebrush plots, no production differences were evident in either the early ( $F_{1,36} = 0.9$ ,  $P = 0.35$ ) or mid-summer ( $F_{1,36} = 0.01$ ,  $P = 0.94$ ) grazing pastures. However grass production in the late grazing burn remained lower than its unburned counterpart ( $F_{1,36} = 4.17$ ,  $P < 0.05$ ).

### **ANCOVA Results**

Variation in herbaceous peak standing crop explained by variables in the top model is summarized in Table 2.4. Habitat treatment (Burn, unburned mountain big

sagebrush, or black sagebrush) explained the largest portion of variation in the model. Considering all factors in an ANCOVA, burn plots had lower grass production the first year (2006) following the prescribed fire (prior to implementation of grazing timing treatments) than paired unburned mountain big sagebrush and lower production than black sagebrush plots within the early and late grazing pastures (Figure 2.3). However, this initial post-burn year, the burn plot in the pasture that would subsequently be treated with mid-summer grazing had greater production than burn plots in the other two pastures.

The first summer following grazing timing manipulation (2007), grass production in the late-summer burn plot remained significantly lower than burn plots in the other pastures and marginally lower than the corresponding unburned mountain big sagebrush plot (Figure 2.3). Grazing timing also appeared to impact unburned habitats, with greater grass production in the mid- and late grazing mountain big sagebrush plots, compared to the early grazing mountain big sagebrush plot (Figure 2.3). Adjusted means for all black sagebrush plots decreased from pretreatment means, although due to high variability differences were not significant. By the second post grazing treatment growing season (2008) the only difference that remained between burned and unburned mountain big sagebrush plots was marginally greater biomass production in the mid-summer burn (Figure 2.3). However, the adjusted mean for the mid-summer black sagebrush plot was lower than either burns or unburned mountain big sagebrush plots in all pastures.

Significant main effects in the analysis of covariance were habitat (HAB), Year 1 (Y1), pretreatment productivity (PRE), and fall through spring precipitation (FWS).



Habitat type (HAB = Burn, unburned big sagebrush, or black sagebrush) accounted for most of the variation in the model (Table 2.4). While PASTURE did not have a significant main effect, it did interact significantly in year 2 (Y2) and marginally with HAB in two-way interactions. PASTURE didn't have a significant interaction with Y1 by itself; however, a three-way interaction which also included HAB was significant. Main effects, together with interaction effects, indicate that the response in peak standing crop was highly dependant on time since treatment (burn), but that the response within burns and unburned habitats was dependant on the pasture.

Here I present some key coefficients from the top ANCOVA model. The main effects of pretreatment productivity and precipitation resulted in a positive increase in grass production (Coef = 0.54; Coef = 0.099). However, pretreatment productivity had a negative relationship with grass production in interactions with both grazing treatment years (Coef = -0.77; Coef = -0.65), indicating that plots with higher pre-treatment productivity exhibited proportionally less increase in productivity following grazing than plots where pretreatment productivity was lower. The intercept for burn treatment (Coef = -1.68) was lower (confidence interval did not overlap zero) than unburned mountain big sagebrush habitat (Coef = 1.00), indicating that unburned mountain big sagebrush had greater production than burns the initial post-fire year prior to grazing manipulation. Conversely, the intercept for black sagebrush habitat decreased significantly the first post-grazing year (Coef = -1.67), signifying reduced grass production. The intercept for burn habitats in the early and mid-summer (Mid) pastures increased the first post grazing year (Coef = 4.29; Coef = 1.61, respectively) indicating a spike in grass production that

year. However, this effect is somewhat mediated by lower overall production in the early grazing pasture the first post grazing year (Coef = -1.59).

## **2.4 DISCUSSION**

My research suggests that short-term enhancement of livestock forage availability following prescribed fire in high-elevation big sagebrush may be limited. One of the proposed benefits of prescribed fire in big sagebrush rangelands is the release of herbaceous vegetation from competition with woody vegetation for nutrients, light and soil moisture theorized to result in increased livestock forage production (Laycock 1979). Many sagebrush rangelands are given two growing seasons after fire before grazing is reintroduced. Grazing deferment is based on the assumption that post-fire grazing rest promotes herbaceous recovery (BLM 2007). However, the duration of grazing rest may be a less important factor in determining post-fire herbaceous production than the timing, intensity, and duration of use during the growing seasons that immediately follow the fire (Bunting et al. 1998, Bruce et al. 2007, Bates et al. 2009). Under grazing pressure similar to my study (50% utilization), Bates et al. (2009) reported that herbaceous yield in ungrazed burns and burns grazed in either spring or summer the first two years after fire, or the second and third year after fire (six total treatments), all exceeded unburned controls by the third post-fire year. However, Bates et al. (2009) remarked that seed production was negatively affected by grazing during the early growing season, which, in theory, could inhibit herbaceous recovery in some cases. Like much of the research on post-fire grazing in sagebrush systems, inferences from their study came from data collected in Wyoming big sagebrush communities. The spike in production with the

earliest of my grazing treatments contrasts with the potential inhibition of recovery suggested above. Additionally, forage production in the late summer grazing burn never exceeded production in unburned mountain big sagebrush during any of the three post-fire years sampled. This suggests that late summer grazing may inhibit the rate of herbaceous recovery and negate the one year spike in forage production following fire that was observed in pastures grazed earlier in the growing season. Lower production with late season grazing at my study site contrasts with findings noted by Laycock (1979) that grass production is stimulated by late summer or early fall grazing.

I collected measurements for peak standing biomass in mid-June. My objective in choosing this collection date was to obtain a standing crop measurement prior to cattle entering the study site in late June. However, due to the high-elevation at Stratton, additional production likely occurred after this collection date. Thus, more forage may have been available to livestock in the late grazing burn in the initial year than was available to them when they occupied either of the other pastures. If this was indeed the case, cattle may have spent more time grazing in that particular burn the first year, consuming seed heads in addition to vegetative plant parts, reducing recruitment of herbaceous components in that pasture the following year. However, my measurement of forage availability in the late grazing burn was not significantly different than unburned habitats in either of the post-grazing years. Harniss and Wright (1982) found that herbaceous yield was not impacted by late summer grazing by sheep. In contrast, my results suggest that late-summer grazing by cattle immediately following fire may have inhibited enhancement of herbaceous productivity. Thus, late summer grazing

immediately after a fire may conflict with management objectives if a short-term increase in forage production that exceeds pre-burn production levels is desired (Figure 2.3).

My results also suggest that grazing timing in pastures that contain a combination of burned and unburned patches may have short-term impacts on forage production in unburned habitats (both mountain big sagebrush and black sagebrush). Prior to alteration of grazing timing, herbaceous production was similar in all plots that were not part of proposed burn treatments. However, the second post-fire year after a single year of post-fire grazing, production within early grazing unburned mountain big sagebrush was lower than mid- and late summer pastures (Figure 2.3). Low first year production in burns during early summer the year prior to this measurement may have forced livestock to concentrate grazing pressure within unburned habitats where herbaceous components were more readily available. This in turn may have impacted production within these unburned habitats the first post-grazing growing season. However, because production in the early grazing burn increased dramatically the second summer, cattle likely shifted their focus to the burn area in that pasture, allowing unburned habitat to recover to pre-burn production levels by the third growing season. Others have documented that cattle often congregate in burn patches when the forage quality in these patches is adequate (Mitchell and Villalobos 1999, Vermeire et al. 2004).

The impacts of grazing timing within black sagebrush habitats were more subtle. As with mountain big sagebrush habitats, grazing pressure the first post-fire year likely increased as cattle dispersed away from burns where herbaceous vegetation was sparse. In general, black sagebrush areas are less productive and less attractive to livestock

(Shown et al. 1969). As a result, these areas typically receive less grazing pressure when more productive riparian or mountain big sagebrush habitat is available. While production in the black sagebrush plot was not significantly different than the mountain big sagebrush plot within the same pasture, there was a pattern of lower mean peak standing crop and greater variability in production within black sagebrush, as evidenced by the wide confidence intervals in Figure 2.3. However, as was the pattern within mountain big sagebrush habitats, these effects seem to have subsided by the third growing season as herbaceous components in the burns recovered. To my knowledge no previous research has been conducted on herbaceous response to grazing timing within black sagebrush habitats.

## **2.5 MANAGEMENT IMPLICATIONS**

Desired objectives of prescribed fires in sagebrush habitat include, but are not limited to, enhancement of wildlife habitat, recovery of vegetation components lost to past over-grazing and fire suppression, and transition to preferred plant communities (i.e., increase herbaceous production; Laycock 1979). In some cases, fire treatments seek to meet multiple objectives simultaneously (see Braun 1987, Crawford et al. 2004). By the third year after prescribed fire, all burn plots at my study site had similar herbaceous peak standing crops compared to paired unburned mountain big sagebrush plots within each of the grazing pastures. Thus, similar to conclusions drawn by Augustine et al. (2010) in sagebrush steppe, my study demonstrated that grazing need not be limited immediately after prescribed fire for herbaceous components to recover to pre-burn levels. However, immediate grazing of prescribed fire treatments in high-elevation sagebrush may not be

advised to meet a management objective of increased herbaceous biomass for livestock forage. A large increase in herbaceous production was limited to one year in my study and was only significantly greater than unburned habitats within a single pasture. This short-term increase in forage production may not outweigh the negative impacts of prescribed fire on sagebrush dependant wildlife species resulting from the near complete mortality of sagebrush plants within prescribed fire treatments (see Chapters 3 & 4 of this thesis).

The growing season at high-elevation sagebrush sites is often abbreviated due to later snowmelt and a lower number of growing degree days compared to lower elevation habitats (Hunter and Grant 1971, Chambers et al. 2007). Thus, results from my study may not be directly comparable to results obtained within some Wyoming big sagebrush systems. In fact, shorter growing seasons might require a longer time period for enhancement of herbaceous production to become evident in high-elevation mountain big sagebrush. Because lower elevation big sagebrush habitats are often moisture limited, a longer growing season might result in substantially greater herbaceous production if the initial post-fire growing seasons coincidentally coincide with above average precipitation cycles (Rhodes et al. 2010). My study was short-term (three years post-fire), and additional insights may have been garnered with additional years of data collection at this study site. Therefore, while I did not observe reduced recovery with immediate post-fire grazing, I am not suggesting that the prevailing recommendation for grazing deferment after fire is unwarranted. On the contrary, the results of my investigation lead me to recommend longer-term monitoring of herbaceous productivity in both grazed and

ungrazed scenarios following prescribed fire in high-elevation sagebrush systems in order to fully understand post-fire plant community dynamics in concert with post-fire grazing. Monitoring of post-fire herbaceous production across a broader geographic range would also likely capture differences in post-fire response attributable to variation in precipitation regimes immediately following fire events. Further, I caution that results of such investigations may be misleading if differences in pretreatment conditions between experimental units and in annual precipitation are not properly addressed within the analytical framework. Clearly, my results suggest that within high elevation sagebrush communities, like the one represented in my study, prescribed fire may not lead to increased herbaceous productivity as has been suggested (Wright et al. 1979). The limited duration of short-term increases in forage production certainly does not merit large-scale sagebrush removal programs that often adversely impact sagebrush dependent species.

## **2.6 LITERATURE CITED**

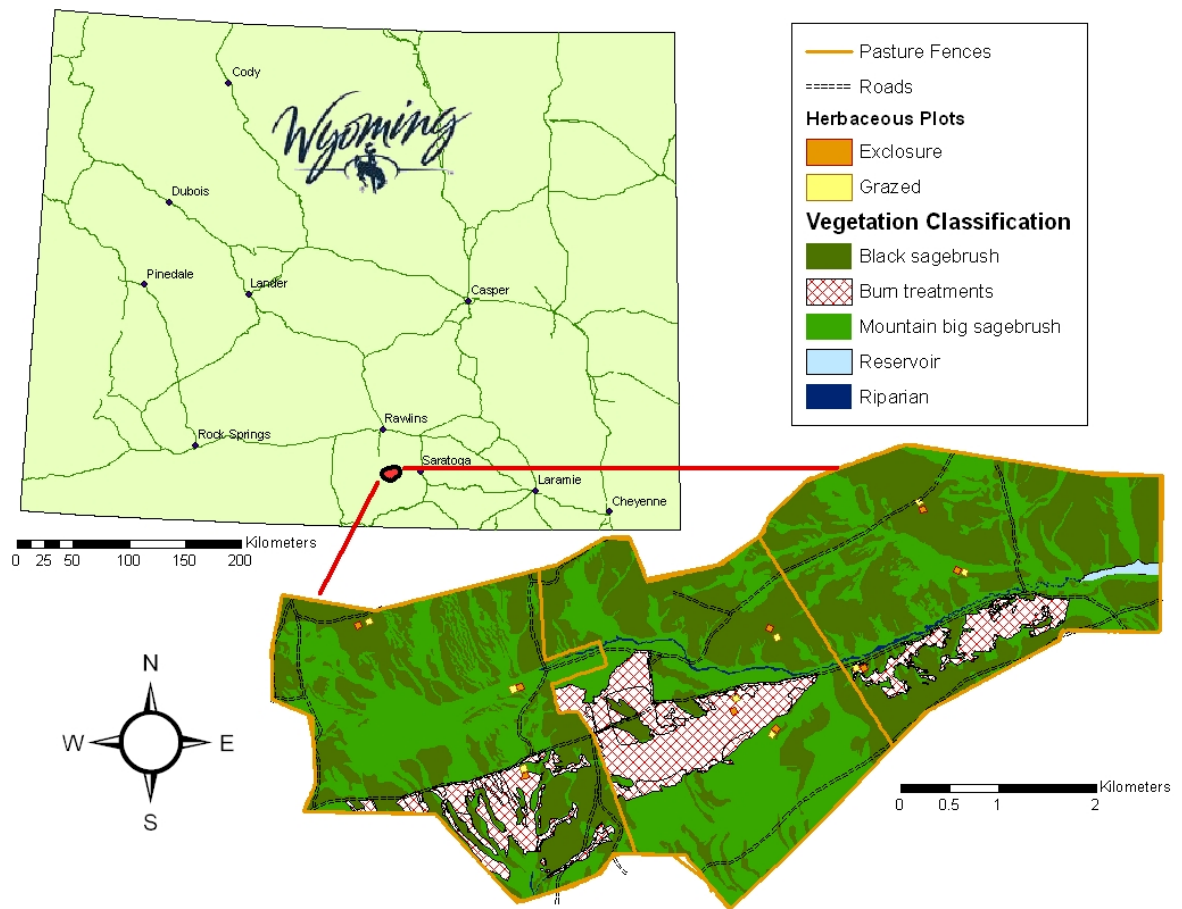
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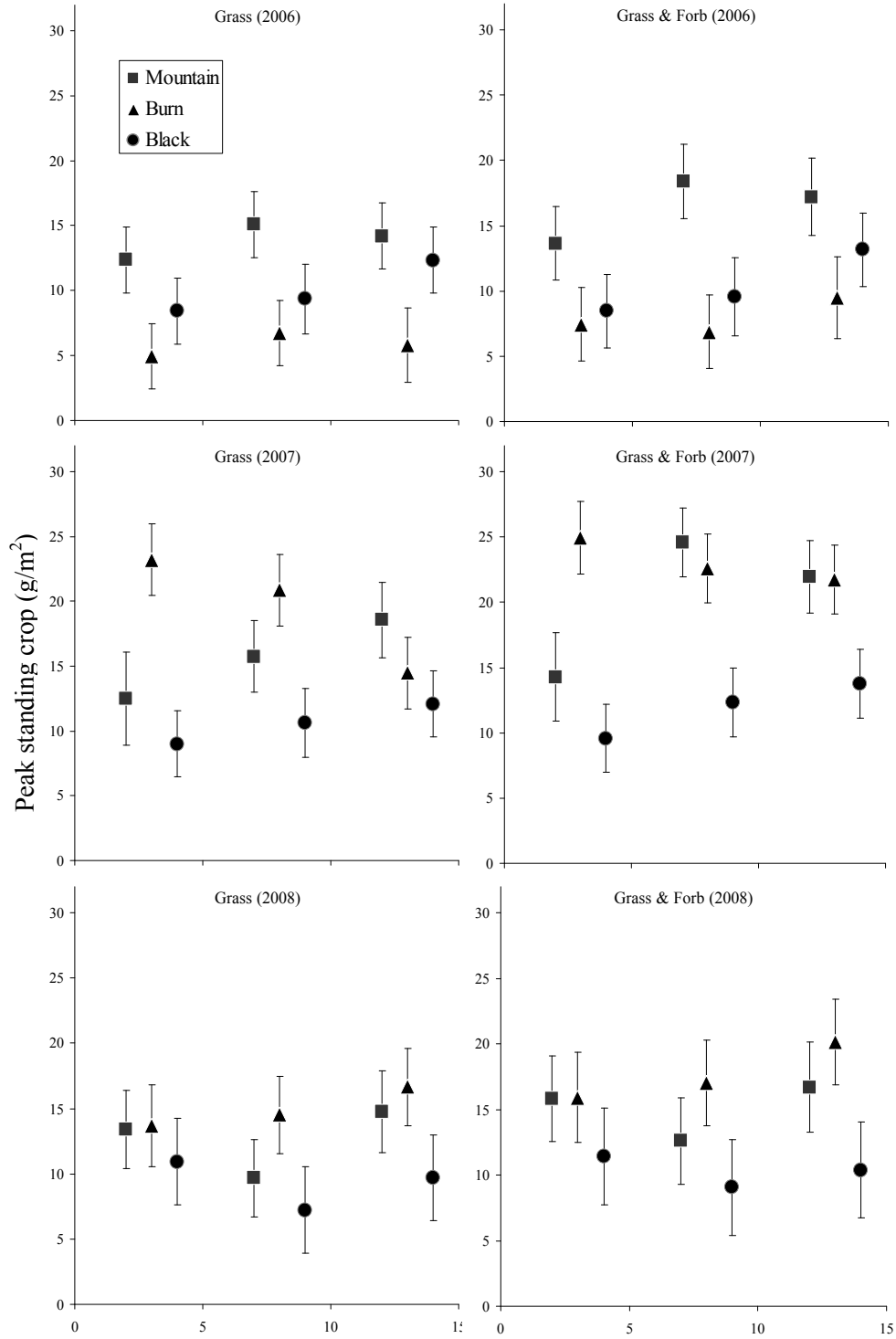


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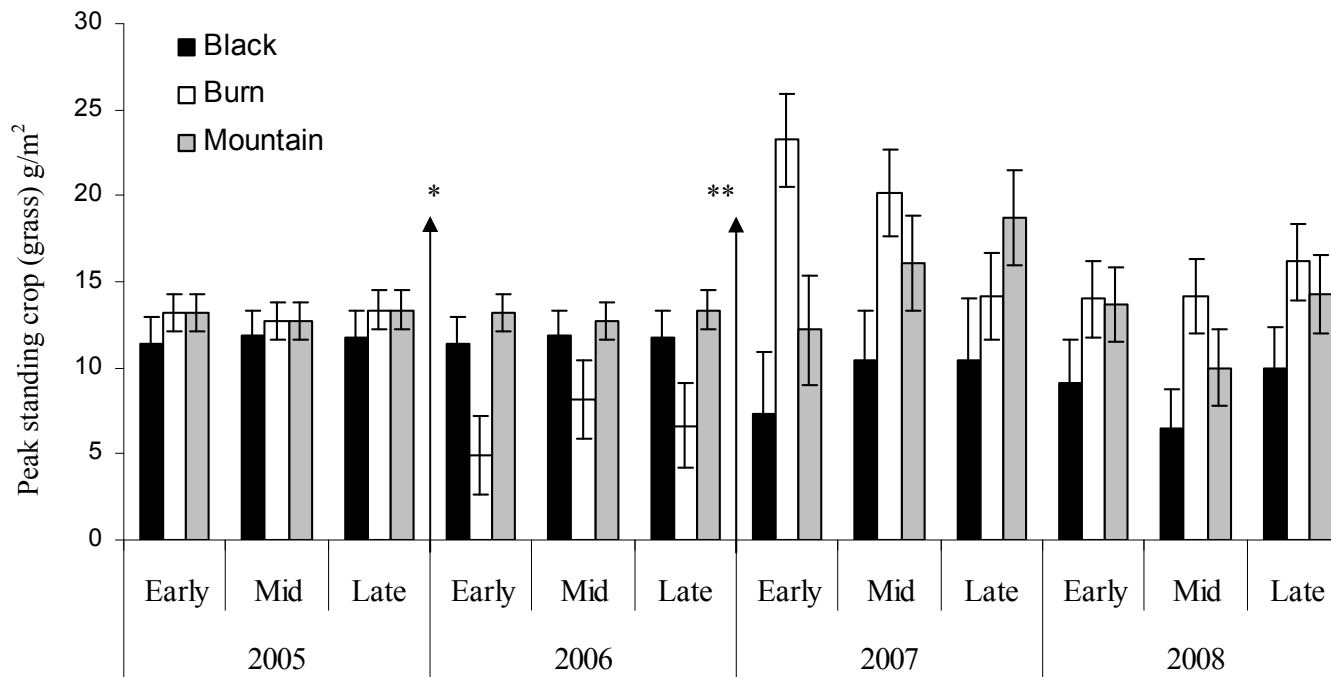
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**Figure 2.1.** Location map for the Stratton Sagebrush Ecological Research Site (Stratton) located 29 km west of the town of Saratoga in south-central Wyoming. The site is managed by the Rawlins Field Office of the Bureau of Land Management.



**Figure 2.2.** Mean peak standing crop biomass for grass (left side) and total herbaceous standing crop (right side) by year (2006 – 2008) across habitat (Mountain = mountain big sagebrush, Burn = burn treatment, Black = black sagebrush) and grazing treatments (Early = late June, Mid = late July, Late = late August) at Stratton Sagebrush Ecological Research Site following prescribed fire (2005).



**Figure 2.3.** Adjusted means (by relative pre-treatment productivity and precipitation) for grass production ( $\text{g/m}^2$ ) by summer grazing timing pasture (Early, Mid, Late) and sagebrush habitat (Mountain, Burn, Black). Burn treatments occurred in October 2005 (indicated by one asterisk (\*)), and initiation of grazing treatments is indicated by a double asterisk (\*\*) prior to the summer of 2007 measurement. Typically, early grazing occurred in late June, mid-summer grazing in late July, and late grazing in late August.

**Table 2.1.** Pre-treatment differences in grass production across grazing treatment pastures within burn treatments and unburned big sagebrush plots and within grazing treatments between paired burn/unburn plots (2004 – 2005, prior to fire or grazing treatments) at Stratton, Wyoming. Differences are based on square-root transformed variables. Because differences between proposed grazing and paired enclosure plots were minimal, pre-treatment measurements from both plots were combined to increase the sample sizes for the analysis. F statistics and P values are displayed only when  $P < 0.10$ .

I Plot Comparison	2004		2005	
	Grass production	Total production	Grass production	Total production
BurnEarly vs. BurnMid	N.S. <sup>2</sup>	N.S.	N.S.	N.S.
BurnEarly vs. BurnLate	N.S.	N.S.	N.S.	N.S.
BurnMid vs. BurnLate	N.S.	N.S.	N.S.	N.S.
MtEarly vs. MtMid	F <sub>1,126</sub> = 4.03, P < 0.05	F <sub>1,123</sub> = 7.57, P < 0.05	N.S.	N.S.
MtEarly vs. MtLate	F <sub>1,126</sub> = 454.38, P < 0.001	F <sub>1,123</sub> = 569.04, P < 0.001	F <sub>1,120</sub> = 673.34, P < 0.001	F <sub>1,119</sub> = 844.45, P < 0.001
MtMid vs. MtLate	N.S.	F <sub>1,123</sub> = 4.10, P < 0.05	N.S.	N.S.
BurnEarly vs. MtEarly	N.S.	N.S.	F <sub>1,120</sub> = 5.84, P < 0.05	F <sub>1,119</sub> = 3.06, P < 0.10
BurnMid vs. MtMid	F <sub>1,123</sub> = 7.70, P < 0.05	F <sub>1,123</sub> = 18.08, P < 0.001	N.S.	F <sub>1,119</sub> = 9.77, P < 0.05
BurnLate vs. MtLate	F <sub>1,123</sub> = 3.78, P < 0.10	N.S.	F <sub>1,120</sub> = 5.30, P < 0.05	F <sub>1,119</sub> = 5.20, P < 0.05

<sup>1</sup> Plot comparisons prior to grazing and fire treatments for proposed burns (Burn) and unburned mountain big sagebrush (Mt) plots across grazing timing pastures (Early = late June, Mid = late July, Late = late August) and between burns and paired mountain big sagebrush controls (not burned) within individual pastures.

<sup>2</sup> Differences between cell means are based on post-hoc ANOVA comparisons. N.S. indicates non-significant.

**Table 2.2.** Cross-comparison of herbaceous production within experimental plots for three years following prescribed fire (2006 – 2007) at Stratton, Wyoming. Differences are based on square-root transformed variables. F statistics and P values are displayed only when significant ( $P < 0.10$ ).

Plot Comparison	2006		2007		2008	
	Grass production	Total production	Grass production	Total production	Grass production	Total production
BurnEarly vs. BurnMid	N.S. <sup>2</sup>	N.S.	N.S.	N.S.	N.S.	N.S.
BurnEarly vs. BurnLate	N.S.	N.S.	F <sub>1,75</sub> = 18.59, P < 0.001	F <sub>1,75</sub> = 2.78, P < 0.10	N.S.	F <sub>1,73</sub> = 1.8, P < 0.10
BurnMid vs. BurnLate	N.S.	N.S.	F <sub>1,75</sub> = 10.52, P < 0.01	N.S.	N.S.	N.S.
MtEarly vs. MtMid	N.S.	F <sub>1,77</sub> = 5.65, P < 0.05	N.S.	F <sub>1,75</sub> = 22.91, P < 0.001	F <sub>1,73</sub> = 3.16, P < 0.10	N.S.
MtEarly vs. MtLate	N.S.	F <sub>1,77</sub> = 3.01, P < 0.10	F <sub>1,75</sub> = 6.85, P < 0.05	F <sub>1,75</sub> = 12.22, P < 0.001	N.S.	N.S.
MtMid vs. MtLate	N.S.	N.S.	N.S.	N.S.	F <sub>1,73</sub> = 5.61, P < 0.05	F <sub>1,73</sub> = 2.96, P < 0.10
BurnEarly vs. MtEarly	F <sub>1,78</sub> = 16.89, P < 0.001	F <sub>1,77</sub> = 9.43, P < 0.01	F <sub>1,75</sub> = 21.33, P < 0.001	F <sub>1,75</sub> = 23.6, P < 0.001	N.S.	N.S.
BurnMid vs. MtMid	F <sub>1,78</sub> = 21.52, P < 0.001	F <sub>1,74</sub> = 32.9, P < 0.001	F <sub>1,75</sub> = 6.85, P < 0.05	N.S.	F <sub>1,73</sub> = 5.35, P < 0.05	F <sub>1,73</sub> = 3.64, P < 0.10
BurnLate vs. MtLate	F <sub>1,78</sub> = 21.52, P < 0.001	F <sub>1,77</sub> = 12.55, P < 0.001	F <sub>1,75</sub> = 4.07, P < 0.05	N.S.	N.S.	N.S.

<sup>1</sup> Plot comparisons of burns (Burn) and unburned mountain big sagebrush (Mt) plots across grazing timing pastures (Early = late June, Mid = late July, Late = late August) and between burns and paired mountain big sagebrush controls within the same pasture.

<sup>2</sup> Differences between cell means are based on post-hoc ANOVA comparisons. N.S. indicates non-significant.

**Table 2.3.** Cross-comparison of herbaceous production within control (ungrazed) plots for three years following prescribed fire (2006 – 2008) at Stratton, Wyoming. Differences are based on square-root transformed variables. F statistics and P values are displayed only when significant.

I Plot Comparison	2006		2007		2008	
	Grass production	Total production	Grass production	Total production	Grass production	Total production
BurnEarly vs. BurnMid	N.S. <sup>2</sup>	N.S.	N.S.	N.S.	N.S.	N.S.
BurnEarly vs. BurnLate	F <sub>1,18</sub> = 4.99 P < 0.05	F <sub>1,18</sub> = 5.94 P < 0.05	F <sub>1,36</sub> = 11.03 P < 0.01	F <sub>1,36</sub> = 4.06 P < 0.1	N.S.	N.S.
BurnMid vs. BurnLate	N.S.	F <sub>1,18</sub> = 6.31 P < 0.05	F <sub>1,36</sub> = 10.90 P < 0.01	F <sub>1,36</sub> = 3.25 P < 0.1	N.S.	N.S.
MtEarly vs. MtMid	N.S.	N.S.	N.S.	F <sub>1,36</sub> = 3.98 P < 0.1	N.S.	F <sub>1,36</sub> = 4.22 P < 0.05
MtEarly vs. MtLate	N.S.	N.S.	N.S.	N.S.	F <sub>1,36</sub> = 4.55 P < 0.05	F <sub>1,36</sub> = 5.17 P < 0.05
MtMid vs. MtLate	N.S.	N.S.	N.S.	F <sub>1,36</sub> = 3.98 P < 0.1	N.S.	N.S.
BurnEarly vs. MtEarly	F <sub>1,18</sub> = 3.32 P < 0.10	F <sub>1,18</sub> = 10.4 P < 0.01	N.S.	N.S.	N.S.	N.S.
BurnMid vs. MtMid	F <sub>1,18</sub> = 9.48 P < 0.001	F <sub>1,18</sub> = 4.28 P < 0.10	N.S.	N.S.	N.S.	N.S.
BurnLate vs. MtLate	F <sub>1,18</sub> = 12.68 P < 0.001	F <sub>1,18</sub> = 18.31 P < 0.001	F <sub>1,36</sub> = 7.05 P < 0.05	N.S.	F <sub>1,36</sub> = 4.17 P < 0.05	F <sub>1,36</sub> = 2.95 P < 0.10

<sup>1</sup> Plot comparisons of burns (Burn) and unburned mountain big sagebrush (Mt) plots across grazing timing pastures (Early = late June, Mid = late July, Late = late August) and between burns and paired mountain big sagebrush controls within the same pasture.

<sup>2</sup> Differences between cell means are based on post-hoc ANOVA comparisons. N.S. indicates non-significant.



**Table 2.4.** Analysis of covariance table testing burning and grazing timing effects on peak standing crop of grass at Stratton, Wyoming for two pre-treatment (2004, 2005), one post-burn/pre-grazing (2006), and two post grazing timing treatment years (2007, 2008).

Source of variation	df	Sum Sq	Mean Sq	F value	Significance
HAB	2	111.58	55.79	56.4484	***
PASTURE	2	2.56	1.28	1.2965	
Y1	1	65.77	65.77	66.5415	***
Y2	1	4.28	4.28	4.3299	*
PRE	1	22.25	22.25	22.5151	***
FWS	1	37.84	37.84	38.2869	***
HAB*PASTURE	4	8.28	2.07	2.0936	.
HAB*Y1	2	31.67	15.84	16.0237	***
PASTURE*Y1	2	1.71	0.85	0.864	
HAB*Y2	2	55.33	27.67	27.9926	***
PASTURE*Y2	2	11.93	5.97	6.0369	**
Y1*PRE	1	9.55	9.55	9.666	**
Y2*PRE	1	22.57	22.57	22.8386	***
HAB*PASTURE*Y1	4	30.78	7.7	7.7857	***
Residuals	390	385.47	0.99		

Significance codes: \*\*\* P < 0.001, \*\* P < 0.01, \* P < 0.05, . P < 0.1

Variable definitions: HAB = Habitat (Burn, mountain big sagebrush, or black sagebrush); PASTURE = grazing timing treatment (early, mid, or late summer); Y1 = one year post-grazing; Y2 = two years post-grazing; PRE = pre-treatment productivity measurement; FWS = combined precipitation across fall, winter and spring prior to growing season.

## **CHAPTER 3 : HABITAT USE BY GREATER SAGE-GROUSE IN RESPONSE TO PRESCRIBED FIRE AND SUMMER GRAZING TIMING TREATMENTS IN HIGH-ELEVATION SAGEBRUSH**

### **3.1 INTRODUCTION**

Nearly half of all native sagebrush habitats in North American have been lost as a direct result of land-use change following European colonization (Knick et al. 2003). Habitats that remain have demonstrated changes in vegetation community compositions resulting from poorly managed grazing regimes (McArthur and Plummer 1978), management initiatives involving sagebrush removal (Young et al. 1981), intensified drought conditions (climate change; Nielson et al. 2005), invasive species (Miller and Eddleman 2000), and habitat fragmentation (Vale 1974, Braun 1998). As a result, sagebrush-obligate species such as the sage-grouse (*Centrocercus* spp.), have experienced large reductions in geographic range and population abundance (Braun et al. 1976, Schroeder et al. 2004). Declining populations have led to the recent decision (March 2010) to designate the greater sage-grouse (*C. urophasianus*) as a candidate for Endangered Species Act protection (U. S. Department of Interior 50 CFR Part 17 2010). The Bureau of Land Management (BLM) administers roughly half (about 45 million hectares) of the remaining sagebrush steppe habitat in the western United States. An additional 20% is managed by other state or federal land management agencies. The

large proportion of remaining sagebrush habitats under the jurisdiction of public land management agencies (~ 70%) is a compelling reason to re-examine the effectiveness of management objectives and address the impact of widespread rangeland management initiatives on sage-grouse populations.

Livestock grazing has been one of the dominant land uses across sagebrush habitats since the Civil War (Clawson 1983) and is currently the most wide-spread use of publicly managed sagebrush rangelands, occurring on more than 53 million hectares. In many of these areas, stocking rates prior to 1900 far exceeded carrying capacity resulting in substantial degradation of ecosystem function within these systems (Young and Sparks 1985). Impacts of over-grazing were recognized by the early part of the 20th century and stocking rates on public lands have been significantly reduced, particularly during the last 40 years, generally followed by observable improvements in rangeland conditions (Crawford et al. 2004).

Prescribed fire is used by many private and public land managers in shrub-steppe vegetation communities to achieve a variety of objectives. Some of these objectives include halting advancement of woodland encroachment following prolonged periods of fire suppression (Miller and Rose 1999, Allen et al. 2008), reducing fuel loads where extensive build-up of these materials could lead to catastrophic wildfire events (Crawford et al. 2004, Perchemlides et al. 2008), increasing herbaceous forage production for cattle by releasing grass species from competition with shrubs (Bastian et al. 1995, Holecheck et al. 2004), and recently, as a potential method to increase forb production for sage-grouse habitat enhancement (Martin 1990, McDowell 2000). Between 1997 and 2002, prescribed burns were conducted on average across 37,000 hectares (ha) of BLM

administered lands annually (Connelly et al. 2004). There have been many efforts investigating potential vegetation enhancements with prescribed fire in sagebrush systems including assessments of potential benefits for sage-grouse brood-rearing habitat (i.e., increases in essential brood-rearing forb production). However, both positive (Wirth and Pyke 2003) and negative effects (Nelle et al. 2000) on forb production have been reported. Previous research has documented negative effects from prescribed fire on sage-grouse attendance at display grounds (leks; Connelly et al. 2000), and avoidance of burns has been documented in winter habitat selection (Robertson 1991). However, little research has been conducted to determine if sage-grouse are actually using prescribed fire areas during the brood-rearing season when benefits from increased forb yield should be most apparent (but see Byrne 2002).

In many cases, fire within publicly managed sagebrush habitats is followed by livestock grazing, often after one or two years of post-fire deferment (Bureau of Land Management 2007). Both positive and negative effects of livestock grazing on sage-grouse habitat have been reported (see Beck and Mitchell 2000 for a review). However, most research has focused on indirect effects resulting from habitat alteration related to sagebrush removal programs geared towards enhancement of livestock forage. Indirect negative effects that have been reported include lek abandonment (Hullet 1983), cheatgrass invasion (Vallentine 1989) and failure of herbaceous components to rebound after prolonged over-grazing (West et al. 1984). Evidence for positive indirect effects have also been found, such as reintegration of nutrients (e.g. nitrogen) consumed by ungulates via urine and feces (Hobbs 1996) and the development of new sage-grouse leks

on livestock salting grounds (Hullet 1983). Though limited, direct evidence concerning negative changes in sage-grouse habitat use related to livestock grazing itself include nest abandonment, trampling (Rasmussen and Griner 1938) and increased densities of nest predators following heavy grazing (Giesen 1995), all leading to reduced nest success.

The main objective of my study was to determine effects of prescribed fire on habitat use by sage-grouse within a high-elevation mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) system. I was also interested in determining if differences in habitat use might be related to the timing of grazing in both burned and unburned habitats. Many previous studies examining sage-grouse habitat selection have utilized radio-telemetry to assess resource use and quantify selection. Telemetry studies have enhanced our understanding of both winter (Eng and Schladweiler 1972, Doherty et al. 2007, Carpenter et al. 2010) and summer (Wallestad 1971, Aldridge and Boyce 2007) habitat use by sage-grouse. However, implementing telemetry monitoring requires substantial financial investments in equipment and personnel in order to collar and obtain re-location fixes on an adequate number of individuals to make inference on population response. Telemetry studies necessitate the capture and outfitting of birds with telemetry equipment, which is invasive and can be disruptive to both individuals and populations. Further, individuals outfitted with telemetry equipment usually represent a small fraction of the population, potentially making it difficult to understand use related to localized treatments, especially for a species like sage-grouse with a large maximum home range (2,975 km<sup>2</sup>, Connelly et al. 2004). Inference about population-wide responses to experimental treatments is thus dependant on the behavior of a subset of individuals.

Pellet counts have been used to assess habitat use by multiple species ranging from large ungulates (Trammell and Butler 1995, Marques et al. 2001) to lagomorphs (Palomares 2001, Calvete et al. 2004). This methodology can usually be conducted quickly and with limited expense. Additionally, pellet accumulation within survey areas may be more representative of population-wide use of treatment areas compared to telemetry data from a small subset of individuals. However, few peer reviewed studies have used this method to document sage-grouse habitat use of experimental treatments (but see Dahlgren et al. 2006). Studies have demonstrated that pellet counts have limited value as a direct population census technique for ungulates due to variability in deposition rates related to diet and variability in decomposition rates resulting from factors such as pellet composition (diet), temperature, moisture, and invertebrate disturbance (Eberhardt and Van Etten 1956, Cochran and Stains 1961). However, several investigators have shown that pellet counts can provide a reasonable relative index of wildlife habitat use directly attributable to differences in use between areas with distinct habitat characteristics (Leckenby 1968, Leopold et al. 1984), particularly if the goal is to differentiate between low-, medium-, and high-use habitats on a seasonal temporal scale (Loft and Kie 1988).

Thus, I developed a study to assess seasonal habitat use based on the accumulation of sage-grouse pellets across different grazing (timing) treatments, after burns, and within unburned sagebrush, at a high-elevation sagebrush site in southern Wyoming. I hypothesized that sage-grouse use of burn treatments would be limited the first year after burns because those areas are initially devoid of both shrub and herbaceous vegetation components. However, I expected sage-grouse use of burns to

increase, at least during the summer (i.e., brood-rearing), once forb species diversity and abundance were reestablished. Habitat use within burns is also likely to be greater in areas closer in proximity to unburned big sagebrush, where intact shrubs provide refugia (Aldridge 2005, Aldridge and Boyce 2007). Because late summer grazing may decrease residual cover preferred for nesting and early brood-rearing habitat the following spring (Hockett 2002), I expected lower relative use might follow later grazing treatments. Additionally, the strong dependence of sage-grouse on sagebrush as a winter food source is likely to result in strong avoidance of all burn areas during the winter season.

## **3.2 METHODS**

### **3.2.1 Study Area**

The Stratton Sagebrush Ecological Research Site (Stratton) is located 29 km west of Saratoga in south-central Wyoming (Figure 3.1). Stratton encompasses approximately 2,250 ha managed by the Rawlins Field Office of the Bureau of Land Management (BLM). The average elevation at the site is 2,400 m with terrain characterized by gently rolling hills with 100 m relief between ridge tops and valley bottoms. A riparian drainage (Beaver Creek) runs west to east through the valley bottom bisecting the study area. Soils are Argic Cryoborolls formed from underlying Brown's Park formation (Miocene), cross-bedded sandstone bedrock. The area averages 500 mm of precipitation annually with two-thirds typically resulting from winter snowfall.

Vegetation communities dominated by mountain big sagebrush with pockets of antelope bitterbrush (*Purshia tridentata*) predominate on side- and toe-slopes. High-slope and ridgetop vegetation communities are dominated by black sagebrush (*A. nova*)

with Wyoming big sagebrush (*A. tridentata* ssp. *wyomingensis*) as a secondary shrub. Other sparsely distributed shrubs occurring within the study site include winterfat (*Krascheninnikovia lanata*), mountain snowberry (*Symphoricarpos oreophilus*), and green rabbitbrush (*Chrysothamnus viscidiflorus*). Shrub communities have well-developed herbaceous understory vegetation. Graminoids at the site include Idaho fescue (*Festuca idahoensis*), Rocky Mountain fescue (*Festuca saximontana*), needle-and-thread (*Hesperostipa comata*), and bluegrass species (*Poa* spp.). In general, forbs are a minor component of the herbaceous understory and include flowery phlox (*Phlox multiflora*), ballhead sandwort (*Arenaria congesta*), Great Basin wild buckwheat (*Eriogonum microthecum* var. *laxiflorum*), silvery lupine (*Lupinus argenteus*), yarrow (*Achillea millifolium*), thistle (*Cirsium* spp.) and mat penstemon (*Penstemon caespitosum*).

### **3.2.2 Prescribed Fire & Grazing Treatments**

Fences partition the study site into three grazing pastures. A separate prescribed burn was implemented within each of these pastures in October of 2005, primarily within mountain big sagebrush vegetation communities. The burns comprised approximately 10 percent of the study area (295 ha). Each of the burns resulted in high sagebrush mortality (~85% sagebrush removal) although configurations (burn pattern) varied considerably between pastures. The burn in the central pasture was a larger, more contiguous burn compared to those in the other two pastures (Figure 3.1).

The summer following the prescribed fire, timing of livestock grazing within pastures was manipulated to incorporate early summer (east pasture), mid-summer (middle pasture), and late summer (west pasture) grazing treatments for each of four



subsequent growing seasons (2006 – 2009). In general, early summer grazing occurred in late June to early July, mid-summer grazing occurred late July to early August, and late summer grazing occurred late August to early September. The annual grazing dates in each pasture varied by up to a week depending on the year, but each treatment occurred annually for a duration *ca* 14 days.

Cattle stocking rates within each pasture were calculated using BLM allotment tabulations of historic production and expected forage conditions within each of the pastures (Mike Calton, personal communication). Targeted stocking rates were 0.64 animal unit months (AUM) \* ha<sup>-1</sup> for the early grazing pasture, 0.53 AUM \* ha<sup>-1</sup> for the mid-summer grazing pasture, and 0.60 AUM \* ha<sup>-1</sup> for the late grazing pasture.

### **3.2.3 Pellet Transects**

I conducted pellet counts along permanent 500-m transects to assess relative habitat use by greater sage-grouse within treatments. Transects were placed across the study site in a stratified random design, so that 15 transects were placed in each of the three grazing pastures. Within each pasture five transects each were placed within burn treatment (formerly dominated by mountain big sagebrush), unburned mountain big sagebrush habitat, and unburned black/Wyoming sagebrush habitat (hereafter referred to as black sagebrush habitat). To aid spatial consistency of sampling across survey periods, painted rebar markers ~ 1 meter tall, were placed every 100 meters (including start/end locations) and flagged nails were placed in the ground every 50 meters. During pellet counts, observers walked slowly along transect lines with the aid of a Garmin e-trex hand-held GPS (Garmin International, Inc., Olathe, Kansas, USA). All sage-grouse

pellets detected within one meter of the transect line (1,000 m<sup>2</sup>) were counted and removed from the sampling area.

Transects were initially cleared of pellets in October, 2005, immediately after the burns. My intent was to conduct pellet surveys twice annually through October, 2009. Counts in late October were used to assess relative summer habitat use by sage-grouse. This time period includes a majority of the nesting phase and all of the brood-rearing phase. A second annual count in late May is representative of the entire winter use period, but also includes a majority of the lekking season and a small portion of the nesting phase. Unfortunately, sampling did not occur in October, 2006 due to unfavorable weather conditions which made the study site inaccessible. Thus, the May, 2007 count consisted of a one-year accumulation of pellets and was discarded from the analysis. Therefore, data used in my analyses included the first post-burn winter followed by a one-year gap. The remaining data included pellet counts for the second through fourth post-burn summers as well as third and fourth post-burn winters.

### **3.2.4 Model Variable Development**

Wildlife often select habitat components at multiple spatial scales based on variation in the spatial structure and distribution of these components within the landscape (Mayor et al. 2009). In order to relate pellet counts to treatments, I examined data on habitat characteristics that may be important predictors of sage-grouse habitat selection/abundance at multiple scales. Line intercept measurements (Canfield 1941) for mountain big sagebrush, Wyoming big sagebrush, black sagebrush, and antelope bitterbrush were collected annually in late July or early August. These measurements

were collected along four 30-m sub-transects, radiating from the pellet transect at the 100, 200, 300, and 400 meter marks in alternating directions. Measurements included only live or green vegetation and gaps  $\geq 5$  cm were not included in the measurement to avoid over-estimation of cover values (Connelly et al. 2003). Along each of these sub-transects, shrub heights, excluding inflorescences, were measured for each shrub species occurring within one meter of a focal point located every 10 meters along the sub-transect (four measurements per sub-transect).

In addition to field measurements, I also explored the predictive ability of vegetation variables derived from remotely sensed products and geographic information systems (GIS) software. I performed a maximum likelihood supervised habitat classification of the study area in ArcGIS Version 9.2 (ESRI 2006) using 1-m resolution color infrared (CIR) National Aerial Photography Program (NAPP) imagery collected in 2001. Based on my extensive knowledge of the study site, I selected training data for the classification from visual interpretation of the imagery. The classification included seven habitat types; water, deciduous trees, antelope bitterbrush, mountain big sagebrush, black sagebrush, grass, and bare ground. I then subjected the product to seven passes of a majority filter using the eight nearest neighbors for each pixel in order to remove speckling from the classification (Mas et al. 2010). Because the training imagery for this classification was taken prior to the prescribed fire, I captured burn perimeters with GPS and replaced pre-burn values in the underlying classification. I calculated mean proportions of mountain big sagebrush, black sagebrush, all sagebrush, and herbaceous (classified as grass or burn) binary habitat layers at the pixel scale (1 m) and five circular

moving window neighborhood scales. I chose neighborhood scales that have previously been found to be important for sage-grouse across life stages; 0.25, 1, 6.25, 25, and 100 ha (Aldridge and Boyce 2007, Doherty et al. 2008). I considered the various neighborhood scales for my habitat variables as capturing three biological scales for sage-grouse. Local scales (pixel value, 0.25 ha and 1 ha) are comparable to scales of selection for roost and/or nesting sites and will hereafter be referred to as local scale. Intermediate scales analogous to vegetation patches or potential foraging site selection by sage-grouse (6 and 25 ha), will henceforth be referred to as patch scales. The landscape scale considered for this product (100 ha) provides a broader scale context within which all smaller scales are imbedded and approximates minimum home range estimates for sage-grouse (60 – 1, 820 ha during winter; Schroeder et al. 1999).

Computation time for generating each of the 1-m big sagebrush neighborhood layers was excessive (5+ days for smaller neighborhoods). For that reason I chose to create 10-m resolution neighborhood statistics for all other habitat layers derived from the supervised classification in order to reduce computation time and required memory storage. Variables used in subsequent analyses are the mean GIS pixel value at each calculated neighborhood scale, summarized with a 60 meter buffer around pellet survey transects. This buffer distance was selected for straight forward comparison with cover measurements collected in the field

Recently, a map of various sagebrush habitat components (30 m cell size) was produced across the state of Wyoming (Homer et al. in review). I used these products, which map continuous percent cover estimates for sagebrush (all species), big sagebrush,

total shrub, and herbaceous cover, as additional habitat covariates with which to assess sage-grouse habitat use. Imagery used in development of these products for my study site was captured in May and September, 2006, the first post-fire growing season. Homer et al. (in review) did not collect classification training data within recently burned areas, and when compared to my field measurements, there was considerable over prediction of shrub/sagebrush cover within burn areas. Therefore, in a procedure similar to the one described for the NAPP classified product discussed above, I zeroed out all shrub cover estimates within the burn perimeter, but retained the herbaceous cover estimates as calculated.

Due to the coarser resolution of the cover estimate products (30 m) from Homer et al. (in review), I was unable to generate one of the smaller local neighborhood scales (0.25 ha) for these variables. However, the spatial resolution of this product did allow efficient calculation of an additional, larger landscape neighborhood scale (400 ha), which has been shown to be relevant for sage-grouse winter habitat selection in northeast Wyoming (Doherty et al. 2008). Hereafter I refer to my NAPP classified product as habitat (i.e., sagebrush habitat), and the Homer et al. (in review) vegetation cover estimates as cover (i.e., sagebrush cover).

Because relative habitat use is likely to vary annually based on the local sage-grouse population size, I also wanted to account for potential changes in use related to fluctuations in local sage-grouse abundance. Thirteen known sage grouse leks are located within 15 km of the study site (Wyoming Game and Fish Department). Using local lek count data, I calculated the average maximum male lek attendance each year for

all leks occurring within 5, 10, and 15 km of the study site, and used these as potential model covariates. Three, six, and thirteen leks occurred within the three distance buffers, respectively, including a lek located ~ 700 m from the eastern perimeter of the study site.

Topographic characteristics have been shown to influence habitat selection in many species including sage-grouse (Aldridge and Boyce 2007, Doherty et al. 2008, Carpenter et al. 2010). To assess how topography affects sage-grouse habitat selection at my study site, I derived topographic ruggedness (*tri*; Riley et al. 1999) and compound topographic (*cti*) indices from a 1-m resolution digital elevation model (DEM) obtained from the National Map Seamless Server (<<http://seamless.usgs.gov/>>). Details concerning formulas used to calculate these indices can be obtained by referring to scripts available online at the Environmental Systems Research Institute ArcScripts website: <[www.esri.com/arcscripts](http://www.esri.com/arcscripts)>. Terrain ruggedness describes the degree of elevation difference between adjacent DEM cells and *cti* is related to both slope and upstream contributing area via orthogonal flow direction, and has been correlated with soil moisture and vegetation productivity (Gessler et al 1995). I calculated zonal statistics for both of these indices at each of the scales discussed above. Definitions, scales and values ranges for all GIS derived variables are listed in Table 3.1.

### **3.2.5 Statistical Analysis**

I used the number of sage-grouse pellets counted during individual surveys (pellet abundance) as a relative measure of habitat use/selection by sage-grouse in proximity to surveyed areas. Poisson regression is often used to model count data to explanatory variables. However, count data in ecological datasets often display a high degree of

overdispersion relative to the Poisson distribution (Hoef and Boveng 2007). Indeed, my pellet count data contained a large number of zero counts, as well as variance that exceeded the mean. I chose to use negative binomial regression rather than zero inflated negative binomial modeling to determine if zero counts were attributable to systematic components within the model rather than implementing a more complicated zero inflated model where zeros are incorporated into a random model component. Justification for this choice is based on the fact that in many cases such abundance distributions arise from negative binomial distributions that have small means (Bliss and Fisher 1953, McCullagh and Nelder 1989). When this is the case, the extra term for zero inflation is unwarranted. Additionally, the implication of zero inflated models is that replicate observations are comprised of observations partitioned into distinct areas where the objects of the count (in this case, sage-grouse pellets) do not occur, and areas where they do occur in accordance with a specified distribution (Warton 2005). Because my survey areas were comprised entirely of habitats known or thought to be used by sage-grouse, I felt it inappropriate to assume that pellet occurrence is narrowly defined in this manner, thus, I chose not to explore zero inflated models.

Since counts are repeated over time, autocorrelation among repeated measures can lead to inflated Type I error when not properly considered during analysis of longitudinal panel structured data (Schielzeth and Forstmeier 2009). Thus, I explored the impact of various correlation structures within a Generalized Estimating Equation (GEE) framework as suggested by Cui and Qian (2007). No serial correlation was found in the response variable across sampling locations and models with independent, exchangeable

and 1<sup>st</sup> order autoregressive correlation structures returned nearly identical model coefficients. Because model estimates were consistent across correlation structures, I collapsed the analysis to a negative binomial model within a generalized linear model (GLM) framework, removing any consideration of correlation structure.

After exploring the appropriateness of longitudinal correlation structure and count distribution dispersion my analysis proceeded using the `glm.nb()` function of the MASS package (Venables and Ripley 2002) in Program R. I conducted separate analyses for fall and summer counts, hereafter referred to as summer and winter use/selection, respectively. I also assessed annual relative habitat use using both annual counts and examined differences in habitat selection between seasons by specifying season (Season = difference in winter compared to summer) as an additional covariate in a year-round model. For each of the analyses (summer, winter, year-round), I conducted a three stage model selection process (Figure 3.2). First, I conducted a univariate analysis of all potential explanatory variables (Hosmer and Lemeshow 2000). Previous studies have suggested that sage-grouse may select sagebrush habitat/cover at intermediate levels (Aldridge and Brigham 2002, Aldridge 2005), thus quadratic effects for vegetation variables were also permitted to compete in the univariate analyses to assess whether non-linearities improved explanatory power. Because I included quadratic effects at this stage of the analysis, the preliminary analysis phase shall henceforth be referred to as functional form analysis rather than univariate analysis. In my year-round analysis I allowed additive and multiplicative seasonal functions (e.g.,  $\text{Sage1m} * \text{Season}$ ) to compete along-side univariate and quadratic models.



Following functional form analysis, the second stage in model development proceeded with the building of four sub-models as a means to reduce the number of candidate variables prior to final model selection. Each of the four sub-models encompassed distinct explanatory categories that might affect sage-grouse habitat use/selection. These categories included vegetation characteristics (both GIS and field measurements), relative sage-grouse abundance in the local area (lek counts), categorical experimental treatments, and topographic characteristics. There were a large number of potential variables and scales available for inclusion in the vegetation sub-model (112, including quadratics). I examined all subsets that included top ranked functional forms ( $< 2$  AIC) and additional variables not highly correlated ( $r < |0.65|$ ) with the initial or subsequent variables (Figure 3.2). However, to avoid over-fitting the models the number of predictor variables in a single model was limited to 10% the number of data points (Nunnally 1978, Marasculio and Levin 1983). Because interaction effects for continuous variables can be difficult to interpret, particularly since my candidate sub-model sets were already quite large, I did not explore interaction effects between vegetation covariates directly while building the vegetation sub-model. However, because I wished to explore all potential interaction effects implicit in my experimental design, candidate sets for treatment sub-models included all three-way habitat treatment (burned, unburned big sagebrush, black sagebrush), grazing treatment (early, mid-, late summer) and time since burn (tb) interactions. Because a known lek site is located inside the study site perimeter, the grazing pasture covariate was highly correlated with distance to the nearest sage-grouse lek (LekDist). Therefore, treatment candidate sub-models also included

variants with interactions between habitat treatment categories and distance to nearest lek (LekDist) without grazing pasture. The best seasonal sub-model in the candidate set for each of the assessments was subsequently selected using Akaike's Information Criteria (AIC, Burnham and Anderson 2002). Variables included in the top sub-model then moved forward to the final model selection phase.

The candidate set in final model selection included all combinations of components within the best sub-models (within 2AIC of top sub-model) in each explanatory category. In some cases, high correlation was observed between select variable(s) from the best vegetation sub-model and a variable from the best treatment sub-model. In this case, subset combinations that included both of these sub-models contained variants with correlated vegetation variables substituted into the treatment sub-model and vice versa. Because I included this type of variant combination in final model selection, some candidate models included interactions between treatment and continuous variables and between two continuous variables. In a comparison of all sub-model combination candidates the final model characterizing habitat use/selection by sage-grouse was selected using an information theoretic approach. Following final model selection, I plotted the interaction effects for select covariates that were of particular interest to facilitate interpretation. These plots display the fitted values for each term with marginal main effects absorbed into the interaction term while values of other predictors are fixed at their means (Fox 2003).

The model selection process included competition between categorical treatment variables and continuous GIS variables representing habitat characteristics. However, I

suspected that some of my GIS derived variables might exhibit geographically distributed patterns associated with my categorical habitat treatments (Burn, Big sagebrush, or Black sagebrush), or grazing treatment pastures (Early, Mid-, or Late summer) despite lack of high correlation. Therefore, prior to analysis, I graphically examined patterns in the data related to habitat characteristics across habitat treatments and grazing pastures using a bootstrap hypothesis test of equality (subsequently referred to as kernel density plots) for select covariates across treatment categories using the `sm.density.compare()` function within the `sm` package of Program R (R Development Core Team 2009).

### **3.3 RESULTS**

#### **3.3.1 Functional Form Analysis (Univariates and Quadratics)**

All field measurements for cover and shrub height were highly correlated ( $r > 0.70$ ) with local scale zonal means (pixel scale, 0.25 ha, and 1 ha) for all big sagebrush habitat and shrub cover GIS variables (i.e., proportion of big sagebrush habitat and estimated big sagebrush cover). At larger scales ( $\geq 6$  ha), percent sagebrush habitat was highly correlated with all derived cover estimates at similar and larger scales. For this reason, shrub habitat/cover variables available for inclusion in model development were often limited to one or two vegetation variables at each of the local, patch and landscape scales. Conditional kernel density plots for select variables also displayed distinct vegetation patterns at multiples scales within habitat treatment categories (Figure 3.3) and associated with grazing treatment pastures (Figures 3.4 – 3.5). All survey sites within burn treatments had much lower proportions of fine scale sagebrush habitat, local scale shrub cover, patch scale shrub cover, and landscape scale big sagebrush cover (Figure

3.3). The early grazing pasture had lower proportions of landscape scale black sagebrush habitat, was closer to the nearest lek, and had more landscape scale big sagebrush habitat than the other two pastures. Additionally, survey transects within the early and late grazing pastures varied in amount of landscape scale burn edge, whereas the amount of burn edge in the mid-summer pasture was unimodal (Figure 3.4). Vegetation patterns across grazing pastures were particularly evident between burn survey transects. Burn transects in the mid-summer pasture contained very low proportions of fine scale sagebrush habitat, burns in the early grazing pasture varied from zero to low proportions, while burns in the late grazing pasture displayed high variability (Figure 3.5).

A positive response to average fine-scale proportion of sagebrush habitat (pixel scale, 1 m) was the top explanatory variable for both winter and year-round relative sage-grouse habitat use in proximity to survey transects. In both of these analyses the nearest competitor was a positive curvilinear quadratic functional form of the same variable and scale at  $\geq 2$  AIC (Table 3.2). In contrast, summer vegetation selection was best characterized by a positive linear relationship with amount of sagebrush habitat at a larger, but still local, neighborhood scale (0.25 ha). However, several other local scale vegetation variables also displayed good predictive performance ( $< 2$  AIC, Table 3.2) in the summer functional form analysis. A negative relationship with burn edge (25 ha) best characterized patch scale winter habitat selection followed closely by a similarly negative impact of burn edge at the smaller patch scale (6 ha; Table 3.2). While not within 2 AIC of the best patch scale predictor, the next best ranked models indicated asymptotic selection for either total shrub cover or big sagebrush cover at slightly above average

available cover levels. Negative impacts of burn edge (25 and 6 ha) were also top patch scale predictors in summer, followed by curvilinear quadratic effects indicating selection for greater proportion of habitat dominated by sagebrush at 6 ha, and intermediate levels of total sagebrush or big sagebrush cover at 6 ha, although these last three were not within 2 AIC of the top covariate functional form (Table 3.2). Not surprisingly, top performing patch scale variables for year-round habitat use had similar patterns as the seasonal models with greatest explanatory ability associated with avoidance of burn edge and selection for intermediate shrub cover (Table 3.2).

At landscape scales (100 and 400 ha) sage-grouse pellet abundance was greatest at intermediate levels of total shrub cover in winter (100 ha) followed by avoidance of burn edge. Other quadratic effects for total shrub and sagebrush cover were also strong competitors at this scale (Table 3.2). The negative relationship between pellet abundance and landscape burn edge was even more pronounced in summer, where this covariate outperformed all other landscape variables ( $\Delta AIC > 4$ ). The top three covariate functional forms for year-round selection at the landscape scale were negative impacts of burn edge as a univariate or with additive or multiplicative interactions with season.

### **3.3.2 Sub-models**

#### **Winter**

Model definitions for top vegetation and treatment sub-models ( $\leq 2 \Delta AIC$ ) for winter, summer and year-round analyses are summarized in Tables 3.3 – 3.6. Reported incidence rate ratios (IRR) in the following text represent exponentiated coefficients from the model with the lowest AIC value and indicate the relative percent change in pellet

accumulation predicted for each 1 percent increase in the explanatory variable or in the case of categorical variables the expected percent change in comparison to the reference category. For instance; an IRR value of 1.2 indicates a 20% increase in the expected number of sage-grouse pellets for each 1 percent increase in the predictor, while an IRR of 0.8 predicts a 20% reduction ( $= 1 - 0.8$ ) in accumulated pellets.

AIC-selected winter, summer and year-round vegetation sub-models were each dominated by variables at the local scale (average pixel value, 0.25 or 1 ha). In winter, all top vegetation sub-models indicated a positive influence of average proportion of habitat dominated by sagebrush, and landscape scale big sagebrush cover on habitat use. All top vegetation sub-models also portrayed a negative association with greater levels of patch scale shrub cover, and landscape-scale proportions of black sagebrush, herbaceous, and combined sagebrush habitat. The first and second ranked models differed only in the inclusion of a negative response to the amount of patch scale burn edge in the top model.

Top treatment sub-models for winter habitat selection (Table 3.3) contained an interaction between burn treatment (main effect IRR = 0.011) and distance to the nearest lek (main effect LekDist IRR = 1.00, interaction IRR = 0.999). I used unburned mountain big sagebrush as the reference category, therefore, during winter, burn areas had 99% less relative use than unburned big sagebrush. This effect was magnified by an additional 0.1% with each 1 m increase in distance from the nearest lek. The study site in general received less relative sage-grouse use with time since burn (IRR = 0.43), but use in burns increased over the study period (interaction IRR = 5.01). However, this effect was limited to burns that were relatively close to the lek as a three-way interaction

between burns, time since burn, and distance to lek indicated lower use with increases in the applicable continuous covariates (IRR = 0.999). The top treatment sub-model indicated greater selection for transects categorized as black sagebrush habitat (IRR = 14.84) diminishing with distance from the on site lek (interaction = 0.999). These interactions indicate a 1% decrease in habitat use with each 1-m increase in distance to the nearest lek suggesting habitat use near the lek was not necessarily consistent with habitat use elsewhere at the study site due to concentrated sage-grouse activity in proximity to the lek during the spring.

All topographic variables had similar performance in the univariate analysis ( $\leq 0.5 \Delta AIC$ ). However, compound topographic index at the 400 ha scale had the lowest AIC value and moved forward as the best topographic variable. The sage-grouse population metric with the lowest univariate AIC in the winter analysis was the average of counts at leks within 5 km of the study site. However, in all three seasonal analyses, confidence intervals for the best lek metric overlapped zero indicating very weak support for this variable as a univariate.

### **Summer**

Because multiple, correlated local scale (1 or 30 m) vegetation variables displayed similar predictive performance in functional form analysis, my candidate model set for summer vegetation use/selection was quite large. As a result, 18 candidate sub-models were within 2 AIC of the top model (Table 3.4). Variables appearing in all top summer vegetation sub-models were avoidance of both patch scale burn edge (6 ha) and landscape scale proportion of sagebrush dominated habitat (100 ha), but positive

association with proportion of patch scale sagebrush habitat (25 ha). Positive curvilinear quadratics for local scale sagebrush, big sagebrush or shrub cover (all 30 m) appeared in more than half of the top models. Indeed, many of the top models were simply variants of each other (i.e., the additional or removal of a single predictor). Thus, individual models in this analysis had low AIC weight even though the cumulative weight of very similar models would suggest support for variables that appeared in a majority of top models. In top models that did not include a shrub cover estimate variable, there was either a positive effect of local scale proportion of sagebrush habitat (linear or curvilinear) or a negative relationship with local proportions of burn area (1 m). An additional variable appearing in the top model (and seven others) represented a positive effect for burn edge at a landscape scale (400 ha). In all top models that did not contain landscape scale burn edge, there was instead either a positive association of patch scale herbaceous habitat (burns and grassy riparian habitat) or a negative association with landscape scale herbaceous habitat. Local and patch scale effects taken together highlight the additive importance of sagebrush habitat in proximity to non-sagebrush (or herbaceous) habitat.

The best summer treatment sub-model indicated 84% less relative use in burn treatments (IRR = 0.136) compared to unburned habitats (both mountain and black sagebrush). The top model also indicated that habitat use in the mid-summer grazing pasture (IRR = 0.428) was 57% less than habitat use in early and late summer grazing pastures. Other variables included in top ranked treatment sub-models (Table 3.4), but exhibiting weaker support, included a negative trend with time since burn, greater use of



the early grazing pasture compared to the late grazing pasture, lower use of survey transects dominated by black sagebrush compared to mountain big sagebrush, and a negative interaction between burns and time since burn.

### **Year-round**

All top AIC-selected vegetation sub-models characterizing relative year-round habitat use by sage-grouse (Table 3.5) indicated positive selection (IRR = 1.026) for sagebrush habitat at the finest scale (1 m) and lower overall use of the entire study site in winter compared to summer (IRR = 0.247). Treatment models also contained an interaction effect indicating greater importance of local-scale sagebrush habitat in winter habitat selection (IRR = 1.022). Other factors appearing in a majority of the top year-round vegetation models included a negative effect of patch scale burn edge (IRR = 0.999), particularly during the winter (IRR = 0.998), positive selection for landscape scale herbaceous cover (IRR = 1.256), and negative selection for both landscape scale habitat dominated by black sagebrush (IRR = 0.797) and total shrub cover (IRR = 0.823).

All top year-round treatment sub-models included avoidance of burns (IRR = 0.625) compared to unburned habitats, particularly in the mid-summer (main IRR = 2.30, interaction IRR = 0.060) and late-summer (main IRR = 1.020, interaction IRR = 0.080) grazing pastures. However, although there was an overall negative trend in use at the study site with time since burn (IRR = 0.755), use within the late grazing pasture increased 38% each additional year subsequent the burn (IRR = 1.381). As with the vegetation sub-model, top year-round treatment sub-models indicated lower study site use in winter compared to summer (IRR = 0.617), particularly within burn treatment

areas (interaction IRR = 0.171). Additional interactions in the highest ranking model included greater use within survey areas dominated by black sagebrush vegetation (IRR = 2.622), particularly in winter (interaction IRR = 2.675).

The best year-round topographic sub-model included selection for increasing levels of both patch scale compound topographic index (IRR = 1.006) and landscape scale terrain ruggedness (IRR = 1.007).

### **3.3.3 Final Model Selection**

The top winter habitat selection model contained a combination of vegetation, treatment, population index and topography variables (Table 3.6) with a maximum-likelihood pseudo  $r^2$  ( $pr^2$ ) of 0.52. This model indicated that relative use decreased with patch scale (25 ha) burn edge, patch scale (25 ha) total shrub cover, landscape scale proportions (100 ha) of both black sagebrush and habitat dominated by herbaceous vegetation, and landscape scale (400 ha) compound topographic index. However, use increased with landscape scale big sagebrush cover (400 ha), fine scale proportion of habitat dominated by sagebrush (1 m), and sage-grouse lek attendance within 5 km of the study site. Local scale proportion of habitat dominated by sagebrush was particularly important with increasing distance from the nearby lek (strong interaction effect). All top models (< 2 AIC) for winter habitat selection contained the same core set of covariates. In fact, the other competitive models either lacked topography, population index, and/or landscape scale proportion of herbaceous habitat, or contained an added categorical variable from the treatment model indicating greater use of survey transects classified as dominated by black sagebrush vegetation (Table 3.7).

The final summer sage-grouse habitat selection model contained a combination of vegetation, topographic, and local population metric components ( $pr^2 = 0.38$ , Table 3.8). While treatment variables did not appear in the top model, one of the models within 2 AIC did contain a negative effect for the mid-summer grazing pasture (Table 3.9). However, the absence of any pasture covariates in any of the other top models suggests that differences in summer habitat selection might be better explained by within pasture variations in shrub vegetation characteristics rather than grazing effects. All top models displayed a positive curvilinear relationship between relative habitat use and estimated sagebrush cover at the finest available local scale (30 m), a negative relationship to patch scale burn edge (6 ha), but a positive relationships to patch scale proportion of sagebrush habitat (25 ha) and landscape scale burn edge (400 ha). Use across the study site was also influenced by the number of sage-grouse attending leks within 10 km of the study site (average of maximum count). Also appearing in the top model, and at least two other top models, were negative associations with landscape scale proportion of sagebrush habitat (100 ha) and increased use with patch scale compound topographic index (6 ha). As with top winter models, all top summer models were simply variants of the top model.

The final year-round habitat selection model ( $pr^2 = 0.44$ ) had greater complexity than either of the seasonal models, accommodating differences in habitat selection between the two seasons. All of the top models contained negative effects for patch scale burn edge (25ha; with greater effect in winter), and patch scale proportion of herbaceous habitat (25 ha), as well as positive effects for landscape scale estimated herbaceous cover (400 ha), patch scale compound topographic index (6 ha), and landscape scale terrain

ruggedness ((400 ha; Table 3.10). Additionally, all top models included positive selection for fine scale proportion of sagebrush habitat (1 m). Although the main effect for this variable was negative in the top model, interaction effects with season and landscape scale proportion of black sagebrush habitat predict increasing pellet abundance with larger proportions of sagebrush habitat, particularly in winter. Effects of seasonal variable interactions in the top model, while holding all other variables at their means, are displayed in Figure 3.6. Similar to top models for separate analyses of winter and summer habitat selection, the top year-round model also indicates greater overall sage-grouse use of the study site with increasing local sage-grouse population size. All other models within 2 AIC of the top model (Table 3.11) differed from the top model only in the exclusion of the sage-grouse population metric, exclusion of the proportion of landscape scale black sagebrush habitat (and/or its interaction with local sagebrush habitat), inclusion of an interaction of these two variables, or some combination of these.

### **3.4 DISCUSSION**

At my study site relative habitat use by sage-grouse during the winter was strongly associated with increasing big sagebrush cover at the landscape scale (400 ha). During the breeding season (summer), areas with greater local scale (30 m) shrub cover and patch scale sagebrush habitat were used most frequently. Despite seasonal variation in relative habitat use, my investigation indicates that sage-grouse select summer and winter habitat at a combination of local, patch and landscape scales. This supports previous research suggesting that sage-grouse select habitat across multiple spatial scales (Aldridge and Boyce 2007, Carpenter et al. 2010, Doherty et al. 2010).

Functional form (univariate) analyses indicate that local scale proportion of sagebrush habitat (1 m or 0.25 ha; see table 3.2) is important for sage-grouse during both seasons. In winter, sage-grouse most often used habitats with greater proportions of local sagebrush dominated vegetation. While this predictor variable was not a direct measure of percent cover, it indicates dominance of sagebrush vegetation within a very small area (1 m pixels) and when averaged across the 30 m sampling buffer it was highly correlated with sagebrush cover estimates produced by the USGS (Homer et al. in review). Therefore, my results are consistent with results of previous research indicating sage-grouse select winter habitats with greater sagebrush cover than available randomly (Eng and Schladweiler 1972, Wallestad 1975, Robertson 1991, Carpenter et al. 2010). Sage-grouse feed almost exclusively on sagebrush leaves during the winter (Wallestad et al. 1975) and several studies have described big sagebrush as dominating winter diet throughout most of the species geographic range (Wallestad et al. 1975, Remington and Braun 1985, Welch et al. 1991). Positive selection for patch scale big sagebrush cover in the functional form analysis and landscape scale big sagebrush cover in the final winter habitat selection model support this tenet. My study site also contains large areas of black sagebrush habitat, and as documented in other studies (Dalke et al. 1963, Beck 1975, Beck 1977) these areas also receive substantial use during the winter when they are available based on the best winter treatment sub-model. Black sagebrush sites tend to occur on windswept high-slopes where sagebrush vegetation might be more accessible during periods of prolonged winter snow accumulation compared to denser big sagebrush areas generally found down slope. Sage-grouse at my study site displayed strong

avoidance for landscapes with large proportions of herbaceous habitat (burns and areas classified as grass habitat). This indicates that burns resulting in larger proportions of herbaceous dominated habitat reduce winter habitat quality for sage-grouse. The negative effect of patch scale burn edge further indicates the detrimental effect of large, contiguous burns. However, the positive effect of landscape scale burn edge supports findings of Carpenter et al. (2010) suggesting that patchy sagebrush at larger scales can be important in winter habitat selection. The prescribed burns at my study site resulted in high sagebrush mortality. As a result, there was a strong negative correlation between pellet survey sites that were burned and total sagebrush habitat proportions and cover estimates at multiple spatial scales. Strong selection for covariate functional forms with increasing sagebrush habitat or cover supports my hypothesis that sage-grouse would display near complete avoidance of burn areas in winter during the first four post-fire years.

In some cases, increasing biomass production or availability of forbs required during early sage-grouse brood rearing is used as a justification for fire treatments (Crawford et al. 2004). However, my study provides evidence that prescribed fire may adversely affect sage-grouse habitat use in these areas during the time period suggested to provide the most benefit. Previous research has documented that percent cover of brood-rearing forbs often increases following prescribed fire when relevant forb species are present in the pre-fire vegetation community (Young and Evans 1978). I did not quantify the increase in preferred forbs used by sage-grouse broods in burn areas post-fire, but sage-grouse palatable forbs were present. Although my year-round habitat selection

model indicated that avoidance of burn areas was stronger in winter than summer, burns still had very little use during the summer when habitat enhancement is theorized to occur. Therefore, whether or not, essential early brood rearing forbs increase as a result of reduced competition with removed shrubs, and/or, short term increases in nutrient availability produced by the fire itself (Pyle and Crawford 1996), sage-grouse strongly avoided these areas at my study site.

My research also suggests that the pattern of burn resulting from prescribed fire applications impacts relative use during the summer. Sage-grouse pellet abundance was reduced with increasing patch scale burn edge indicating that all burns significantly reduced habitat selection. However, at a landscape scale, sage-grouse selection was positively impacted by burn edge, although with a much smaller coefficient. These two effects taken together, along with overwhelming selection for greater proportions of fine scale sagebrush habitat, indicate that larger, more contiguous burns likely reduce habitat quality while patchy burns that leave large portions of sagebrush vegetation intact are less detrimental (Fisher et al. 1996, Knick et al. 2005). However, even when patches of sagebrush were retained in the burn units, sage-grouse use of burn treatments at my study site can be characterized as minimal, at best.

Pellet counts are a coarse measure of relative use and abundance compared to telemetry data. However, I propose that the pellet count method can be effectively used to ask targeted questions about relative habitat use by sage-grouse in response to treatments. Telemetry studies may not capture such localized use patterns if only a few individuals are collared. Indeed, telemetered individuals often represent a small fraction

of the local population whose movements may not effectively overlap treatment locations, possibly limiting inference about localized treatment effects on populations. Alternatively, pellet transects, in theory, represent a relative measure from the entire localized population, allowing for broader sampling.

Few research efforts have examined both winter and summer sage-grouse habitat selection simultaneously. My year-round habitat selection model highlights how habitat devoid of shrubs at both the local and landscape scale results in large reductions in relative habitat use by sage-grouse. This is not to dismiss the importance of herbaceous vegetation components for sage-grouse, particularly during brood rearing (see Crawford et al. 2004). Indeed, my analysis indicates that year-round sage-grouse habitat use increases with landscape scale herbaceous cover. However, my results reinforce the importance of sagebrush vegetation components throughout the entire year for sage-grouse. Taken together this suggests that areas containing large proportions of shrub habitat with well developed herbaceous understory provide the highest quality habitat. Additionally, if relative habitat use is in any way indicative of habitat quality then increases in local scale herbaceous cover resulting from management initiatives with removal of competitive shrub cover does not necessarily translate into sage-grouse habitat enhancement, even when unburned sagebrush is nearby. Seasonal interactions with fine scale sagebrush habitat in my year-round habitat model emphasizes seasonal variation in habitat requirements, supporting the need for both local and patch scale sagebrush in winter and summer (see Figure 3.6). The winter assessment, in particular,



indicates nearly complete avoidance of areas denuded of sagebrush vegetation by burn treatments, even four years post-burn.

In addition to investigating the effects of prescribed fire on sage-grouse abundance, I was also interested in determining if differences in habitat use/pellet abundance could be related to post-fire timing of livestock grazing. It has been suggested that livestock grazing can negatively impact sagebrush habitat (Connelly and Braun 1997). Beck and Mitchell (2000) produced a synthesis of published research concerning the effects of grazing (by both cattle and sheep) on sage-grouse habitat and habitat use. They described both positive and negative effects, quantified both directly and indirectly, resulting from grazing. However, most of the research discussed measured changes in habitat characteristics, and specifically, impacts on forbs eaten by sage-grouse broods, and inferred how this might affect sage-grouse use rather than directly measuring sage-grouse use. Lupis et al. (2006) did document that sage-grouse (male and females without young broods) avoid heavily stocked pastures while cattle are present. During the development of my winter and summer treatment sub-models, there appeared to be evidence for sage-grouse avoidance of the mid-summer grazing pasture during summer and even stronger avoidance of burns grazed during mid-summer in winter. However, variables associated with grazing pasture did not appear in the final model selected for either season. Distinct vegetation patterns across the three grazing pastures potentially complicated inference concerning grazing effects (see Figures 3.4 – 3.5). Thus, differences in sage-grouse habitat use across grazing pastures were better explained by differences in habitat distribution patterns expressed by the vegetation variables, either

pasture-wide or within the three separate burn treatments. High grazing intensity has been documented to reduce sage-grouse nest success by reducing residual grass cover surrounding nests making them more visible to predators (Gregg et al. 1994, Delong et al. 1995, Hockett 2002). However, stocking rates in my experimental treatments were based on current BLM tabulations of historic range condition and expected forage for the area, and were well below those reported in other assessments of grazing impacts (see Lupis et al. 2006). Thus, it may simply be that grazing intensity was low enough that grazing timing had relatively minor impact on vegetation characteristics relevant to sage-grouse. In turn, this may have resulted in little discernable grazing impact on sage-grouse habitat selection at my study site.

Snow cover at the study site limited my ability to separate habitat use based on pellet accumulation into temporal scales smaller than season-long summer and winter use; late May and mid-October sampling. Telemetry research has documented differences in habitat selection dependant on brood stage (i.e., early versus late brood-rearing periods), as chicks shift from dietary requirements consisting of insects to forbs at a young age, and then to sagebrush as juveniles (Connelly et al. 2004, Kaiser 2006). However, when forb availability is limited, difference in habitat selection based on the age of broods may not exist (Aldridge and Brigham 2002, Aldridge and Boyce 2008). Others have suggested that early brood rearing is the most limiting life stage in sage-grouse population persistence and growth (Lyon 2000, Crawford et al. 2004). This is when sage-grouse would theoretically benefit the most from the increased forb production resulting from a prescribed fire. If summer pellet counts had been separated

into discrete measurements encompassing distinct brood rearing periods, I might have been able to further elucidate the impact of prescribed fire on summer brood habitat use. However, even in ideal conditions this would have been difficult, given that population-wide hatching can span several months for sage-grouse (Schroeder et al. 1999). At lower elevation sites, conducting additional summer counts may also increase detection of less persistent chick pellets, providing additional information on brood use within treatments. This was not feasible at my high-elevation study site due to ground snow conditions that often persisted well into late-May or early-June; typically corresponding to the early brood rearing period. Future studies investigating the impacts of prescribed fire and grazing timing may also benefit from a more detailed comparison of forb species composition within treated and untreated areas, assessing how that might relate to habitat use and sage-grouse abundance.

### **3.5 MANAGEMENT IMPLICATIONS**

At least one study has demonstrated that highly mobile sage-grouse (i.e., adults without young broods) avoid using pastures with a large cattle presence (Lupis et al. 2006). However, short duration (two weeks) timing of summer grazing at low to moderate intensity had no discernable impact on sage-grouse seasonal habitat use at my study site. This suggests that current stocking rate practices utilized by the BLM at this site, regardless of summer grazing timing, are compatible with season-long habitat use by sage-grouse, within unburned sagebrush habitat.

Based on my research and results reported by others, I suggest that land managers use extreme caution when considering prescribed fire as a tool for sage-grouse habitat

enhancement in high-elevation sagebrush systems. Potential benefits resulting from increased forb production within burns appear to be completely negated by the wholesale removal of shrub cover. In fact, prescribed fires implemented in big sagebrush systems most often result in high levels of shrub mortality at small spatial scales. It has been suggested that a patchy mosaic of sagebrush habitat at the landscape scale is optimal for sage-grouse (Aldridge and Boyce 2007, Carpenter et al. 2010). Indeed, my results suggest that a heterogeneous configuration of habitat patches (herbaceous and shrub) plays an important role in habitat use by sage-grouse. However, achieving a burn that conforms to an optimal configuration would not be easy to implement on a large scale. Such an endeavor would require a substantial amount of pre-burn planning in order to achieve desired landscape scale characteristics.

Summer is theoretically when sage-grouse would benefit most from short-term increases in forb production resulting from prescribed fire treatments. However, I have demonstrated that sage-grouse use of burns over the length of the summer season is very low. I suggest that further research may be needed to discern differences in habitat use of burn treatments that may affect within season habitat use (i.e., early versus late brood rearing periods). Additionally, monitoring sage-grouse use of fire treatment areas over a time scale that spans pre-burn to reestablishment of sagebrush dominance is necessary to determine which, if any, successional stage of the plant community might benefit sage-grouse the most. During the winter, sage-grouse at my study site displayed near complete avoidance of burn treatments directly attributable to removal of sagebrush cover at a small spatial scale (6 ha). Therefore, at the very least, prescribed fire treatments,

regardless of management objective, should not be implemented in areas considered to be important wintering habitat for declining sage-grouse populations.

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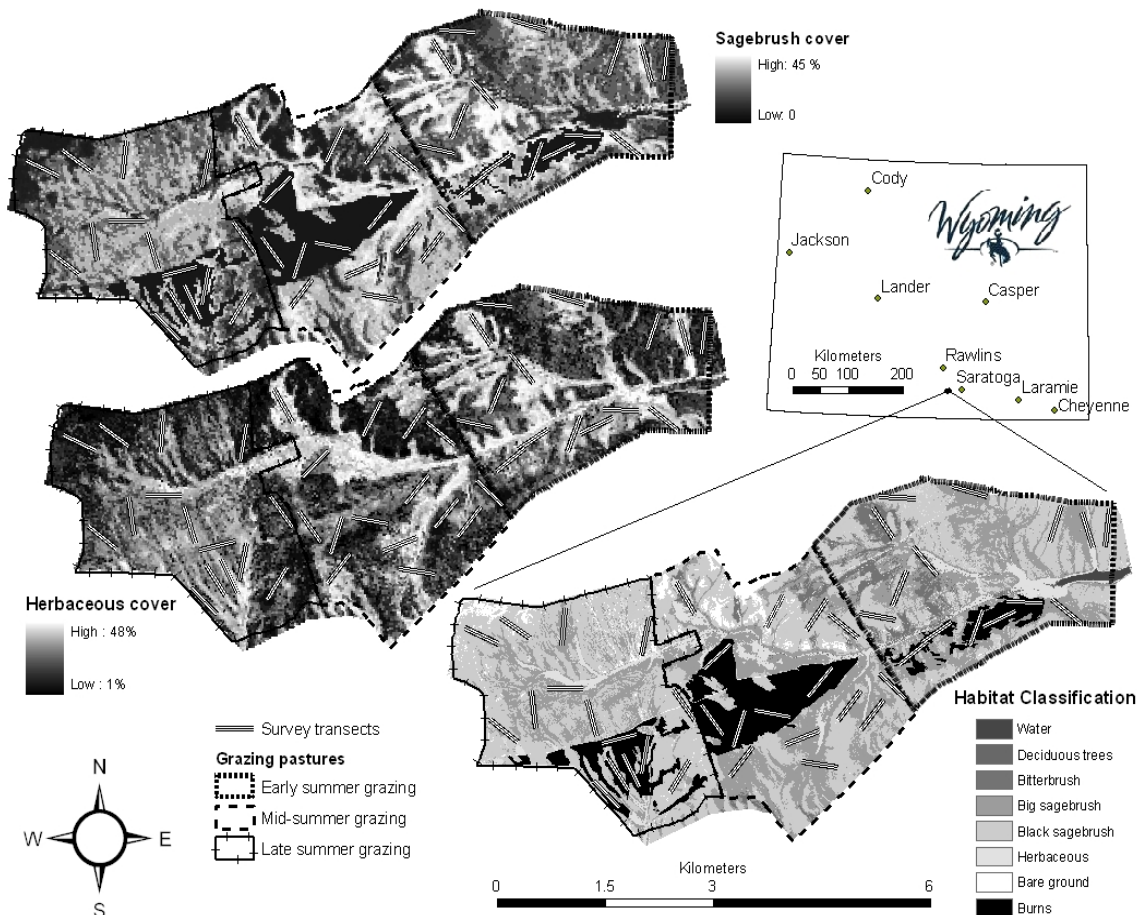
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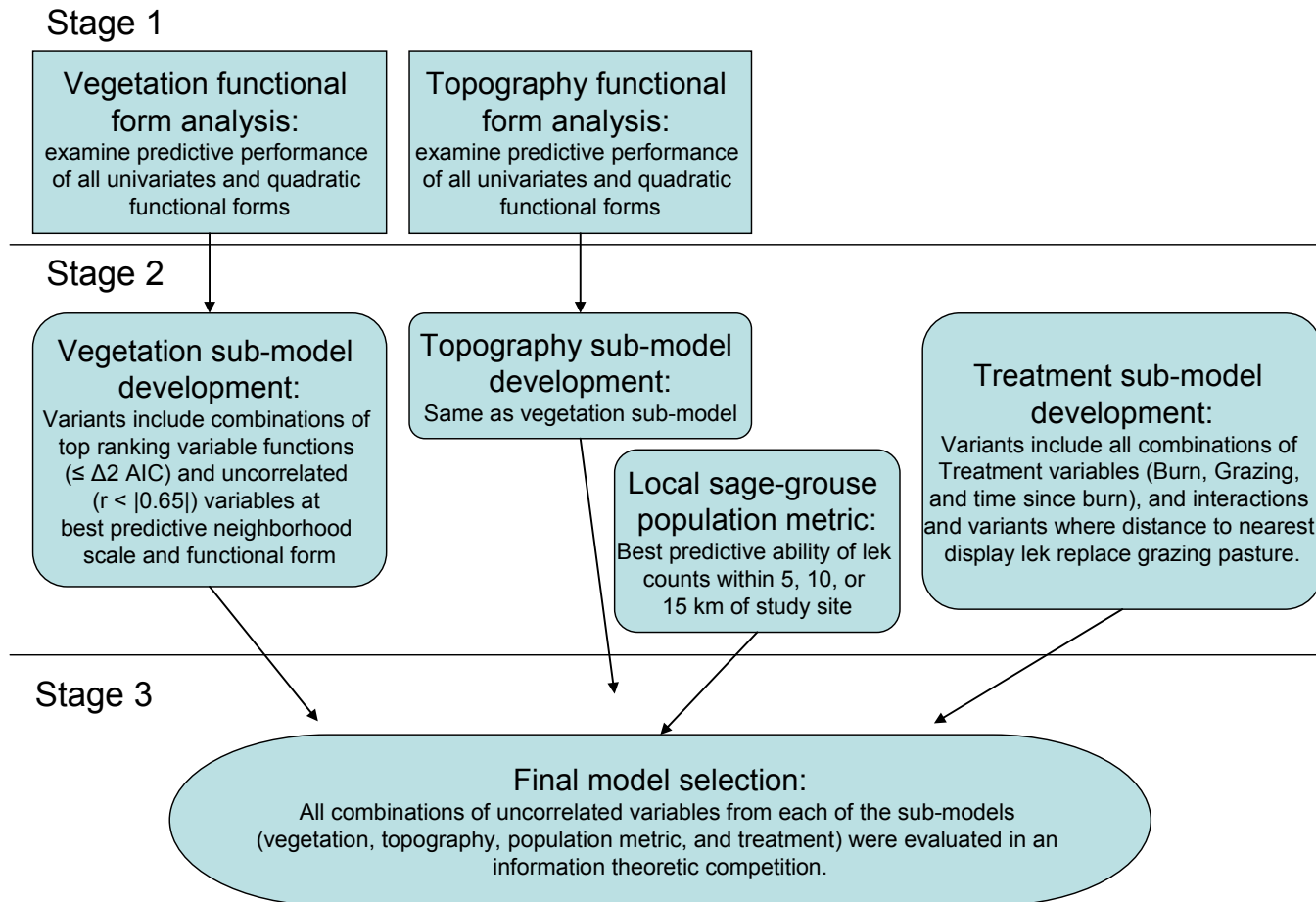
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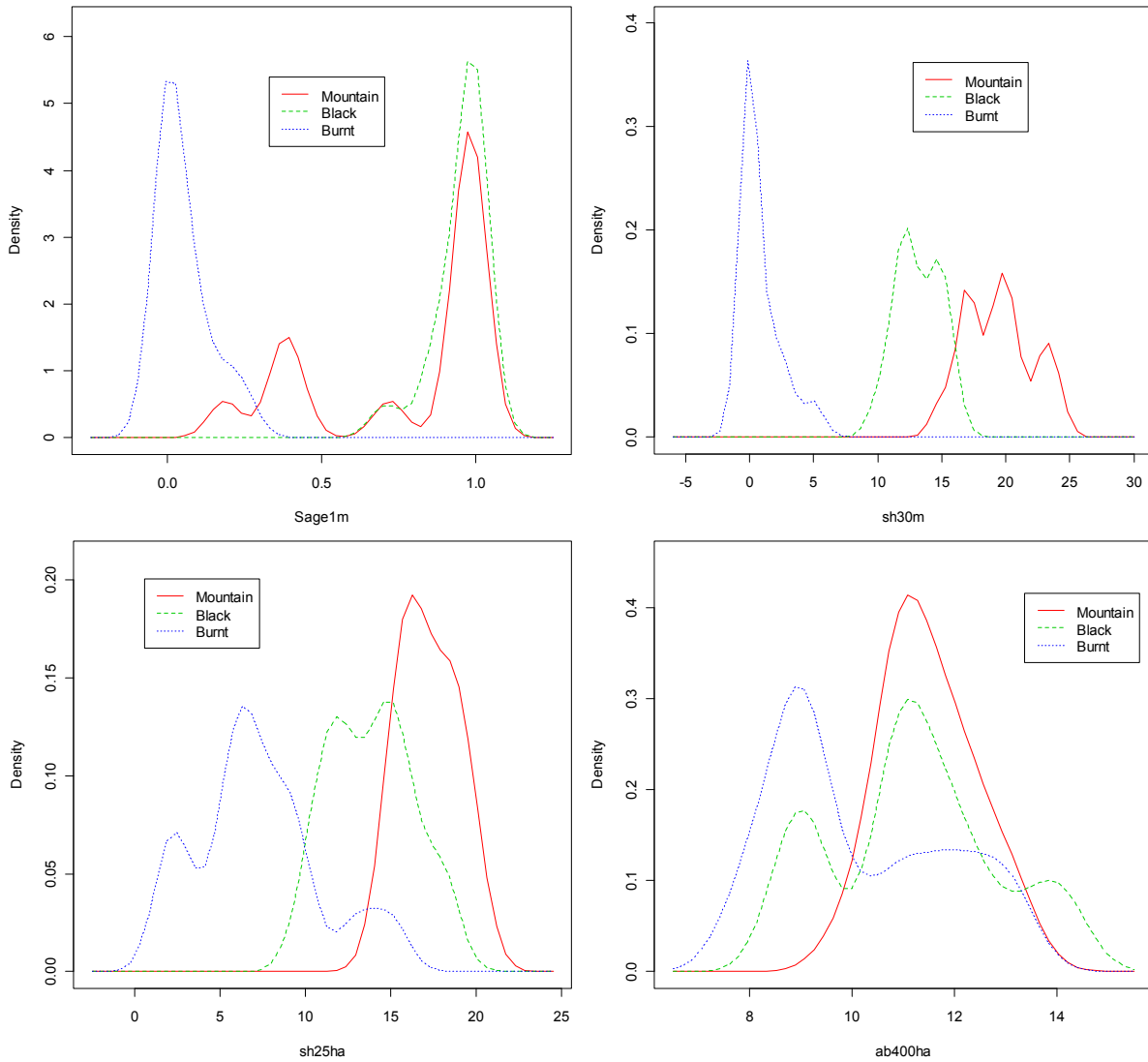
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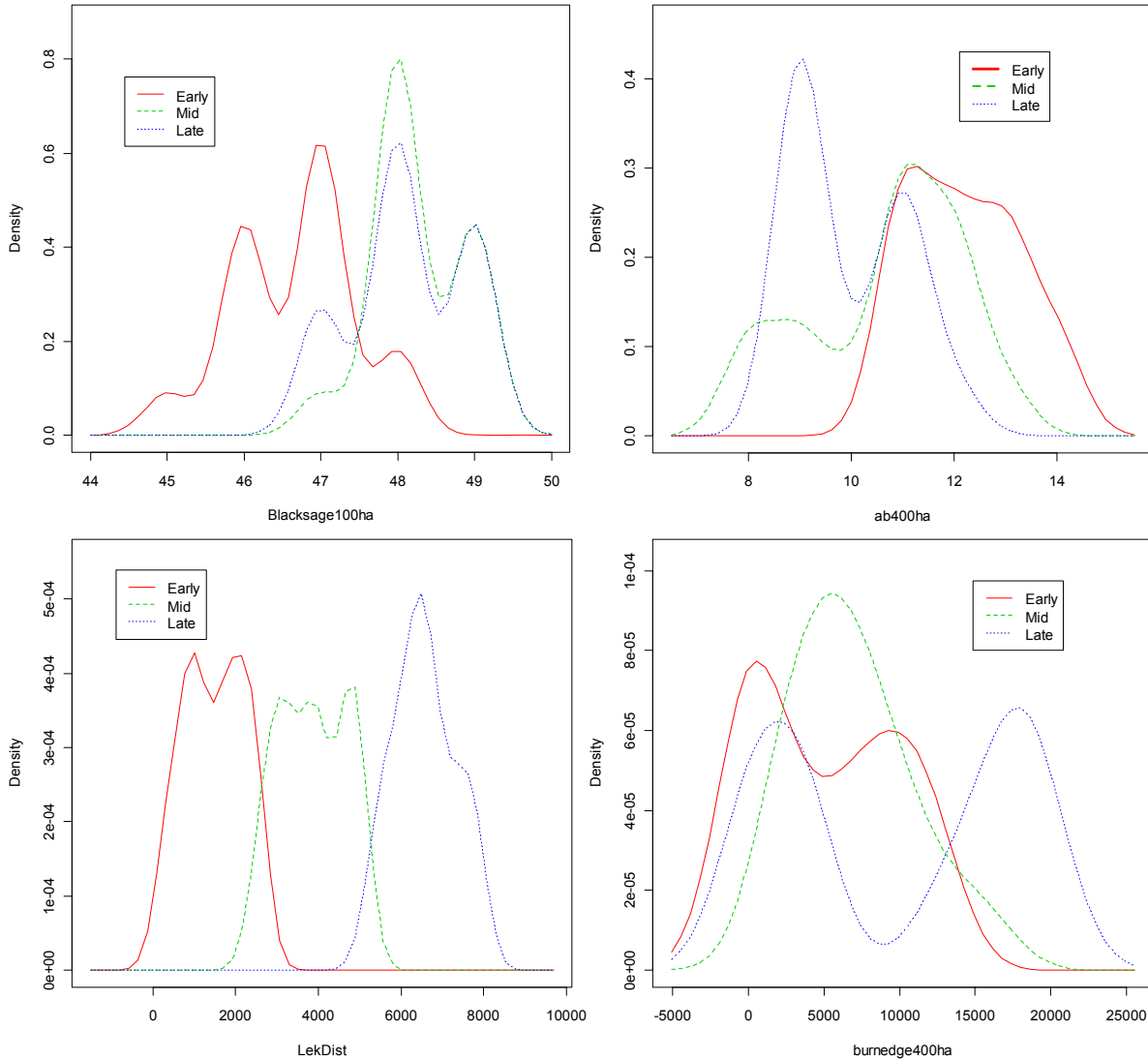
**Figure 3.1.** The Stratton Ecological Research Site is located in south-central Wyoming. The displayed habitat classification (bottom right) is the result of a supervised classification of 1-m resolution color infrared (CIR) National Aerial Photography Program (NAPP) imagery collected in 2001. A prescribed burn was implemented in October, 2005, after NAPP image acquisition. Burn perimeters, delineated by hand-held GPS, were overlaid on the habitat classification and a spatial replacement of values in the underlying habitat classification was performed. Sagebrush and herbaceous cover estimates (top left and center, respectively) are from a USGS habitat assessment product (see Homer et al., in review). Note: Habitat classified as deciduous trees occurs just outside the study site but does not appear on this map.



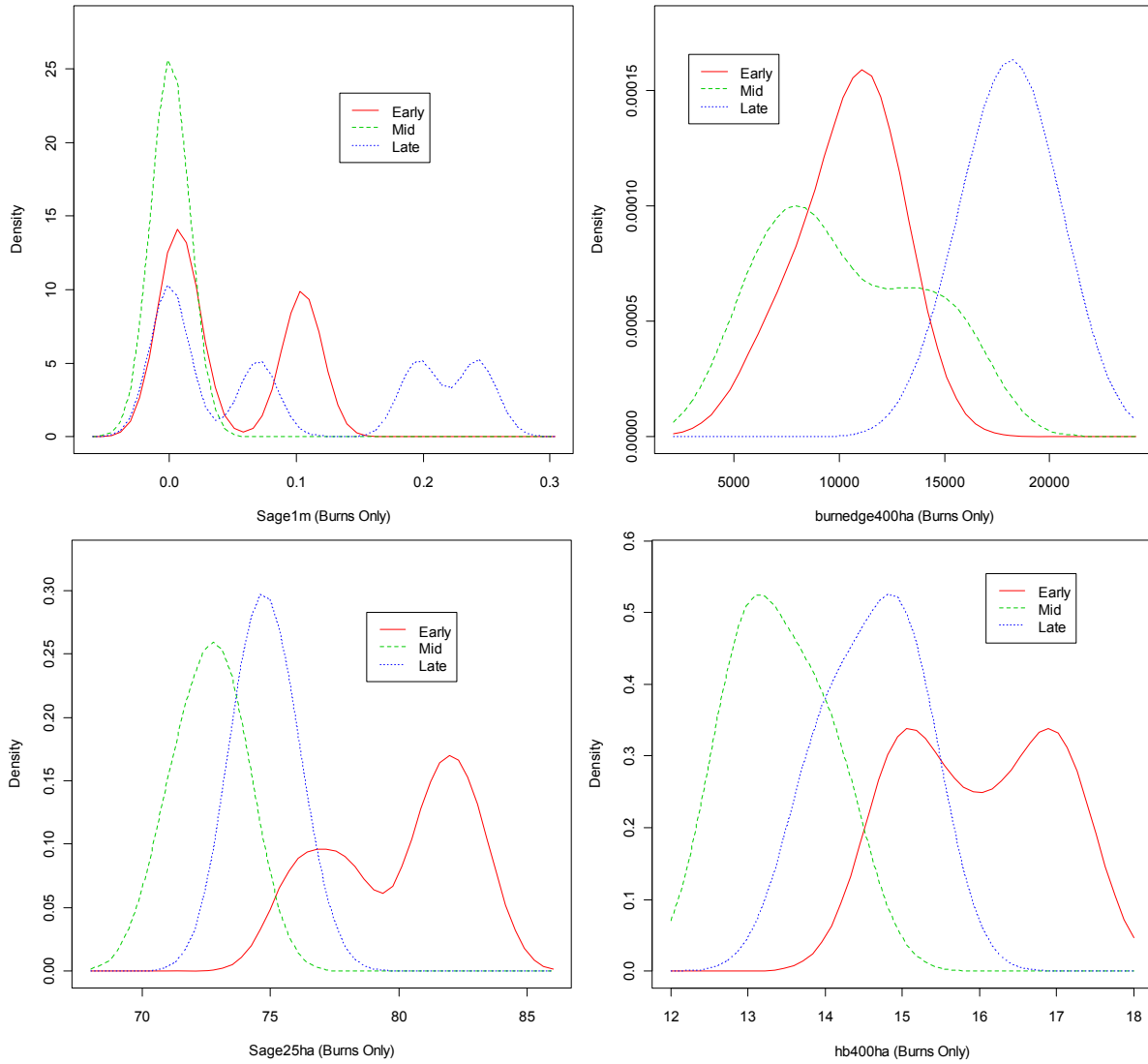
**Figure 3.2.** Three-stage model development strategy implemented in an examination of sage-grouse winter, summer, and year-round habitat selection at the Stratton Ecological Research Site in south-central Wyoming.



**Figure 3.3.** Kernel density plots examining patterns in fine scale (1 m) proportion of sagebrush dominated habitat (top left), local scale (30 m) estimated shrub cover (top right), patch scale (25 ha) estimated shrub cover (bottom left), and landscape scale (400 ha) estimated big sagebrush cover across three habitats (burn treatment, big sagebrush, and black sagebrush; labeled Burnt, Mountain, and Black, respectively) at Stratton Ecological Research Site. All cover estimates derived from product created by USGS and proportion sagebrush habitat derived from classification of NAPP CIR imagery. All measures are static, representing values at time of image acquisition.

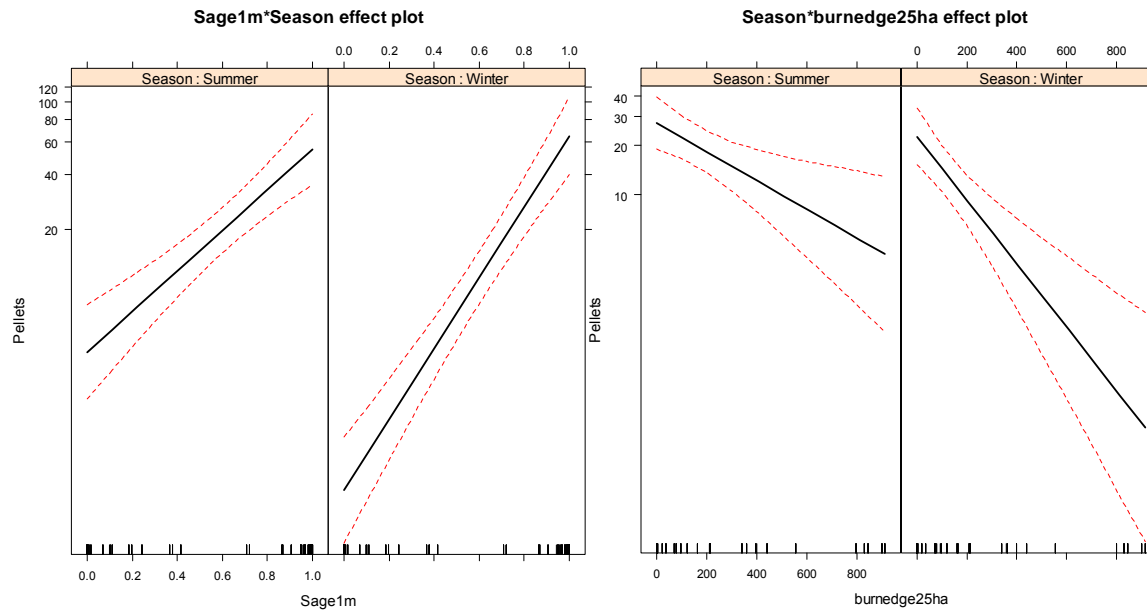


**Figure 3.4.** Kernel density plots describing patterns in distribution of landscape scale (100 ha) proportion of black sagebrush dominated habitat (top left), landscape scale (400 ha) estimated big sagebrush cover (top right), straight line distance to nearest sage-grouse lek (LekDist; bottom left), and landscape scale (400 ha) burn edge across three grazing treatment pastures (Early, mid-, and late summer) at Stratton Ecological Research Site. All cover estimates derived from product created by USGS and proportion sagebrush habitat derived from classification of NAPP CIR imagery. All measures are static, representing values at time of image acquisition.



**Figure 3.5.** Kernel density plots describing patterns in distribution of fine scale (1 m) proportion of sagebrush dominated habitat (top left), landscape scale (400 ha) burn edge (top right), patch scale (25 ha) proportion sagebrush dominated habitat (bottom left), and landscape scale (400 ha) estimated herbaceous cover within burn treatments across three grazing treatment pastures (Early, mid-, and late summer) at Stratton Ecological Research Site. All cover estimates derived from product created by USGS and proportion sagebrush habitat derived from classification of NAPP CIR imagery. All measures are static, representing values at time of image acquisition.





**Figure 3.6.** Effect plots depicting seasonal (winter versus summer) differences in sage-grouse habitat selection for fine scale (1 m) proportions of habitat dominated by sagebrush (left) and avoidance of patch scale (25 ha) burn edge (right) based on top AIC selected year-round sage-grouse pellet abundance model.

**Table 3.1.** Naming conventions, definitions, and value ranges for predictive covariates used to assess sage-grouse relative habitat use at Stratton Ecological Research Site from 2006 – 2009. GIS pixel level resolutions are 1 m for the habitat classification product and 30 m for the cover estimation product. Neighborhood scales investigated were; 0.25, 1, 6, 25, 100 and 400 hectare neighborhoods.

Derived variables	Variable description	1, 10, or 30 m	Variable value range at scale:					
			1	2	3	4	5	6
Bigsage	% of habitat at neighborhood scale classified as habitat dominated by big sagebrush	0 - 95	0 - 94	0 - 94	1 - 90	2 - 77	8 - 55	NA
Black-sage	% of habitat at neighborhood scale classified as habitat dominated by black sagebrush	0 - 97	2 - 80	13 - 69	32 - 61	39 - 57	45 - 49	NA
Sage	% of habitat at neighborhood scale classified as habitat dominated by sagebrush (either big or black)	0 - 100	10 - 98	33 - 96	58 - 97	67 - 83	72 - 79	NA
Herb	% of habitat at neighborhood scale classified as habitat dominated herbaceous vegetation	0 - 1	0 - 89	0 - 66	3 - 34	9 - 17	10 - 12	NA
ab	Estimate of % cover represented by big sagebrush species as derived from Homer et al. (in review)	0 - 20	NA	0 - 19	0 - 18	2 - 16	4 - 16	8 - 14
as	Estimate of % cover represented by all sagsbrush species as derived from Homer et al. (in review)	0 - 22	NA	0 - 21	0 - 19	2 - 17	5 - 16	9 - 14
sh	Estimate of % cover represented by all shrub species as derived from Homer et al. (in review)	0 - 24	NA	0 - 23	0 - 22	2 - 20	6 - 19	10 - 17
hb	Estimate of % cover represented by herbaceous species as derived from Homer et al. (in review)	5 - 28	NA	5 - 28	5 - 24	6 - 21	7 - 18	9 - 17
burnedge	Amount of burn edge calculated via a line statistics algorithm (line length within neighborhood)	NA	0 - 37	0 - 37	0 - 151	0 - 914	0 - 9941	0 - 20418
tri	Topographic ruggedness index is a comparison of elevation differences in immediately adjacent cells (Riley et al. 1999)	NA	42 - 415	44 - 414	46 - 400	57 - 343	81 - 311	92 - 275
cti	Compound topographic index calculated as; $\ln(A_s/\tan(s))$ , where $A_s$ = (flow accumulation +1)*pixel area (m <sup>2</sup> ) and s = slope in radians	NA	411 - 806	410 - 776	406 - 721	430 - 681	427 - 625	461 - 614
tricot	Average percent cover measured along four line-intercept transects per pellet survey transect	0 - 24	NA	NA	NA	NA	NA	NA

**Table 3.2.** Top functional forms for vegetation predictors at local, patch and landscape scales within winter, summer and year-round habitat selection (pellet abundance) analyses for greater sage-grouse at Stratton with associated measures of variable support (AIC weight). Only the top five models for each scale and season are shown. Note: AIC weights for local scale include comparison to all spatial scales, while patch and landscape scales only include comparison to functional forms at similar neighborhood scales.

Scale	Winter			Summer			Year-round		
	Univariate	dAIC	weight	Univariate	dAIC	weight	Univariate	dAIC	weight
Local	Sage1m	0	0.53243	Sage0.25ha	0	0.12667	Sage1m*Season	0	0.7589
	Sage1m + Sage1m_2	2	0.19814	Sage1m	0	0.12547	Sage1m*Season + Sage1m_2*Season	3.7	0.1181
	Herb1m + Herb1m_2	2.7	0.13564	Sage0.25ha + Sage30.25ha_2	0.4	0.1015	as30m*Season + as30m_2*Season	4.6	0.0765
	Herb1m	2.8	0.12863	tricov + tricov_2	1.4	0.06199	Herb1m*Season	5.7	0.0444
	Burn1m	11.8	0.00148	Burn1m	1.6	0.05652	ab30m*Season + ab30m_2*Season	13.9	<0.001
Patch	burnedge25ha	0	0.44545	burnedge25ha	0	0.49555	burnedge25ha*Season	0	0.63737
	burnedge6ha	0.4	0.36423	burnedge6ha	0.7	0.35051	burnedge6ha*Season	1.2	0.35327
	sh6ha + sh6ha_2	3.5	0.07623	Sage6ha + Sage6ha_2	6.7	0.01729	sh6ha*Season + sh6ha_2*Season	11.6	0.00192
	ab25ha + ab25ha_2	3.5	0.07623	as6ha + as6ha_2	7.4	0.01228	as6ha*Season + as6ha_2*Season	11.7	0.00185
	ab6ha + ab6ha_2	8.2	0.00739	sh6ha + sh6ha_2	7.5	0.01189	ab6ha*Season + ab6ha_2*Season	11.7	0.00179
Landscape	as100ha + as100ha_2	0	0.44492	burnedge100ha	0	0.56631	burnedge100ha	0	0.42474
	burnedge100ha	0.9	0.27683	as100ha + as100ha_2	4.9	0.04943	burnedge100ha + Season	1	0.25405
	sh100ha_2	3.4	0.07961	hb100ha	5	0.0457	burnedge100ha*Season	1.3	0.21725
	ab100ha + ab100ha_2	3.5	0.07915	burnedge400ha	5.1	0.04477	as100ha*Season + as100ha_2*Season	4.3	0.04889
	ab100ha	4.2	0.05418	hb400ha	5.1	0.04339	as100ha + as100ha_2	5.7	0.02441

**Table 3.3.** Top vegetation and treatment sub-models (< 2AIC or top 2) representing winter sage-grouse habitat selection at Stratton Sagebrush Ecological Research Site.

Model Definitions	AIC	df	dAIC	weight
Vegetation sub-model				
Sage1m + sh25ha + Blacksage100ha + Herb100ha + Sage100ha + ab400ha + burnedge25ha	9	939.9	0	0.14233
Sage1m + sh25ha + Blacksage100ha + Herb100ha + Sage100ha + ab400ha	8	940	0.1	0.13425
Sage1m + sh25ha + Blacksage100ha + Herb100ha + Sage100ha + ab400ha + burnedge25ha + Blacksage1ha	10	941.2	1.3	0.07278
Sage1m + sh25ha + Blacksage100ha + Herb100ha + Sage100ha + ab400ha + burnedge25ha + Blacksage1ha + Bigsage100ha	10	941.6	1.7	0.06051
Treatment sub-model				
Burn + Black + tb + LekDist + Burn*tb + Burn* LekDist + Black* LekDist + LekDist *tb + Burn* LekDist *tb	11	948	0	0.39694
Burn + Black + tb + lekdist + Burn*lekdist + Black*lekdist	8	950	2.1	0.14212

df = degrees of freedom = # of free parameters + 2, one each for intercept and dispersion parameter (theta)

dAIC = increase in AIC value compared to top model

weight = comparative support for each model amongst entire model set

**Table 3.4.** Top vegetation and treatment sub-models (< 2AIC) representing summer sage-grouse habitat selection at Stratton Sagebrush Ecological Research Site.

Parameters	AIC	df	dAIC	weight
Vegetation sub-model				
as30m + as30m_2 + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha	1054.7	8	0	0.07007
as30m + as30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb6ha	1054.9	8	0.3	0.0608
ab30m + ab30m_2 + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha	1055.2	8	0.5	0.05393
ab30m + ab30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb6ha	1055.2	8	0.6	0.05222
sh30m + sh30m_2 + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha	1055.3	8	0.6	0.05169
Sage1m + Sage1m_2 + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha	1055.4	7	0.8	0.04784
Sage1m + burnedge6ha + Sage25ha + Sage400ha + burnedge400ha	1055.4	7	0.8	0.04784
as30m + as30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb100ha	1055.5	9	0.9	0.04518
ab30m + ab30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb100ha	1055.7	9	1.1	0.04077
sh30m + sh30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb100ha	1055.8	9	1.2	0.03894
Burn1m + burnedge6ha + Sage25ha + Sage100ha + Herb100ha	1055.9	8	1.2	0.0382
sh30m + sh30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb6ha	1055.9	8	1.3	0.03749
Burn1m + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha	1056	7	1.3	0.03628
Sage1m + Sage1m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb6ha	1056	7	1.4	0.03524
Sage1m + burnedge6ha + Sage25ha + Sage100ha + Herb6ha	1056	7	1.4	0.03524
Sage1m + Sage10m_2 + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha + Herb400ha	1056.4	8	1.7	0.02958
Sage1m + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha + Herb400ha	1056.4	8	1.7	0.02958
as30m + as30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb25ha	1056.5	9	1.8	0.02792
Treatment sub-model				
Hab_burn + Graz_mid	1069.1	4	0	0.08804
Hab_burn + Graz_mid + tb	1069.7	5	0.6	0.0646
Hab_burn + Graz_mid + Graz_early	1070.5	5	1.5	0.04259
Hab_burn + Graz_mid + Hab_black	1070.8	5	1.7	0.03691
Hab_burn + Graz_mid + tb + Hab_burn*tb	1070.9	6	1.8	0.03567

df = degrees of freedom = # of free parameters + 2, one each for intercept and dispersion parameter (theta)

dAIC = increase in AIC value compared to top model

weight = comparative support for each model amongst entire model set

**Table 3.5.** Top vegetation and treatment sub-models (< 2AIC) representing year-round sage-grouse habitat selection at Stratton Sagebrush Ecological Research Site.

Parameters	AIC	df	dAIC	weight
Vegetation sub-model				
Sage1m + burnedge6ha + Sage25ha + Sage400ha + burnedge400ha	1055	7	0.8	0.04784
as30m + as30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb100ha	1056	9	0.9	0.04518
ab30m + ab30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb100ha	1056	9	1.1	0.04077
sh30m + sh30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb100ha	1056	9	1.2	0.03894
Burn1m + burnedge6ha + Sage25ha + Sage100ha + Herb100ha	1056	8	1.2	0.0382
sh30m + sh30m_2 + burnedge6ha + Sage25ha + Sage100ha + Herb6ha	1056	8	1.3	0.03749
Burn1m + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha	1056	7	1.3	0.03628
Treatment sub-model				
Hab_burn + Hab_black + Graz_mid + Graz_late + tb + SeasonS + Hab_burn*Graz_mid + Hab_burn*Graz_late + Hab_black*Graz_mid + Hab_black*Graz_late + Hab_burn*SeasonS + Hab_black*SeasonS +	2021	15	0	0.17517
Hab_burn + Hab_black + Graz_mid + Graz_late + tb + SeasonS + Hab_burn*Graz_mid + Hab_burn*Graz_late + Hab_black*Graz_mid + Hab_black*Graz_late + Hab_burn*SeasonS + Hab_black*SeasonS	2022	14	0.7	0.12647
Hab_burn + Hab_black + Graz_mid + Graz_late + SeasonS + Hab_burn*Graz_mid + Hab_burn*Graz_late + Hab_black*Graz_mid + Hab_black*Graz_late + Hab_burn*SeasonS + Hab_black*SeasonS	2022	13	1	0.10804
Hab_burn + Hab_black + Graz_mid + Graz_late + tb + SeasonS + Hab_burn*Graz_mid + Hab_burn*Graz_late + Hab_black*Graz_mid + Hab_black*Graz_late + Hab_burn*SeasonS	2022	13	1	0.10433
Hab_burn + Hab_black + Graz_mid + Graz_late + SeasonS + Hab_burn*Graz_mid + Hab_burn*Graz_late + Hab_black*Graz_mid + Hab_black*Graz_late + Hab_burn*SeasonS	2022	12	1.2	0.09517
Hab_burn + Hab_black + Graz_mid + Graz_late + tb + SeasonS + Hab_burn*Graz_mid + Hab_burn*Graz_late + Hab_black*Graz_mid + Hab_black*Graz_late + Hab_burn*SeasonS + Graz_late*tb	2023	14	1.4	0.08517

df = degrees of freedom

dAIC = increase in AIC value compared to top model

weight = comparative support for each model

**Table 3.6.** Coefficients, coefficient confidence intervals and incident rate ratios (exponentiated coefficients) from the top relative winter habitat selection model based on sage-grouse pellet counts at Stratton Sagebrush Ecological Research Site 2006 – 2009.

	$\beta$	95% Confidence Intervals		IRR
		Lower	Upper	
(Intercept)	50.67	38.44	62.9	-
Sage1m	0.02563	0.01681	0.03445	1.02596
burnedge25ha	-0.00247	-0.00348	-0.00146	0.99753
sh25ha	-0.1709	-0.23299	-0.10881	0.84291
Blacksage100ha	-0.9233	-1.1516	-0.695	0.39721
Herb100ha	-0.8497	-1.2342	-0.4652	0.42754
ab400ha	0.8143	0.6141	1.0145	2.25759
Lek5km	0.0089	0.00402	0.01378	1.00894
LekDist	0.00006	-0.00014	0.00026	1.00006
cti400ha	-0.00923	-0.01469	-0.00377	0.99082
Sage1m*LekDist	0.00058	0.00039	0.00077	1.00058

**Table 3.7.** Top competing (< 2AIC) winter sage-grouse habitat selection models representing relative use at Stratton Sagebrush Ecological Research Site 2006 - 2009.

Parameters	AIC	df	dAIC	weight
Sage1m + sh25ha + Blacksage100ha + ab400ha + LekDist + Sage1m*LekDist + Herb100ha + burnedge25ha + Lek5km + cti400ha	12	941	0	0.04692
Sage1m + sh25ha + Blacksage100ha + ab400ha + LekDist + Sage1m*LekDist + Herb100ha + burnedge25ha + Lek5km	11	942	0.3	0.04104
Sage1m + sh25ha + Blacksage100ha + ab400ha + LekDist + Sage1m*LekDist + Herb100ha + burnedge25ha	10	942	0.4	0.03908
Sage1m + sh25ha + Blacksage100ha + ab400ha + LekDist + Sage1m*LekDist + Herb100ha + burnedge25ha + cti400ha	11	942	1	0.02828
Sage1m + sh25ha + Blacksage100ha + ab400ha + LekDist + Sage1m*LekDist + Herb100ha + burnedge25ha + Lek5km + BLACK	12	943	1.5	0.02264
Sage1m + sh25ha + Blacksage100ha + ab400ha + LekDist + Sage1m*LekDist + Herb100ha	9	943	1.5	0.022
Sage1m + sh25ha + Blacksage100ha + ab400ha + LekDist + Sage1m*LekDist + Herb100ha + Lek5km	10	943	1.5	0.02199
Sage1m + sh25ha + Blacksage100ha + ab400ha + LekDist + Sage1m*LekDist + Herb100ha + burnedge25ha + Lek5km + cti400ha + Sage1m*Lek5km	13	943	1.6	0.02067
Sage1m + sh25ha + Blacksage100ha + ab400ha + LekDist + Sage1m*LekDist + Herb100ha + burnedge25ha + Lek5km + cti400ha + BLACK	13	943	1.8	0.01864
Sage1m + sh25ha + Blacksage100ha + ab400ha + LekDist + Sage1m*LekDist + burnedge25ha + Lek5km + cti400ha	11	943	1.9	0.01788

df = degrees of freedom

dAIC = increase in AIC value compared to top model

weight = comparative support for each model



**Table 3.8.** Coefficients, coefficient confidence intervals and incident rate ratios (exponentiated coefficients) from the top relative summer habitat selection models based on sage-grouse pellet counts at Stratton Sagebrush Ecological Research Site 2007 – 2009.

	$\beta$	95% Confidence Intervals		IRR
		Lower	Upper	
(Intercept)	-0.9383	-9.1763	7.2997	-
sh30m	0.3834	0.32247	0.44433	1.467265
sh30m_2	-0.01302	-0.01575	-0.01029	0.987064
burnedge6ha	-0.02036	-0.02478	-0.01594	0.979846
Burnedge400ha	0.00009	0.00006	0.00012	1.000091
Sage25ha	0.2049	0.1599	0.2499	1.227402
Sage100ha	-0.2246	-0.3454	-0.1038	0.798836
cti6ha	0.00425	0.00193	0.00657	1.004262
MaxLek10km	0.00776	0.00438	0.01114	1.00779

**Table 3.9.** Top competing (< 2AIC) summer sage-grouse habitat selection models representing relative use at Stratton Sagebrush Ecological Research Site 2006 - 2009.

Parameters	df	AIC	dAIC	weight
as30m + as30m <sup>2</sup> + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha + cti6ha + Maxlek10km	10	1051.8	0	0.19795
as30m + as30m <sup>2</sup> + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha + Maxlek10km	9	1052.4	0.5	0.15182
as30m + as30m <sup>2</sup> + burnedge6ha + Sage25ha + burnedge400ha + Maxlek10km	9	1053.7	1.9	0.07809
as30m + as30m <sup>2</sup> + burnedge6ha + Sage25ha + Sage100ha + burnedge400ha + cti6ha + Maxlek10km + Pasture_Mid	11	1053.8	1.9	0.07471

df = degrees of freedom

dAIC = increase in AIC value compared to top model

weight = comparative support for each model

**Table 3.10.** Coefficients, coefficient confidence intervals and incident rate ratios (exponentiated coefficients) from the top relative year-round habitat selection model based on sage-grouse pellet counts at Stratton from 2007 – 2009.

	$\beta$	95% Confidence Intervals		IRR
		Lower	Upper	
(Intercept)	31.23667	16.26157	46.21177	-
Sage1m	-0.27354	-0.43791	-0.10917	0.76069
Winter	-1.33288	-1.90112	-0.76464	0.26372
hb400ha	0.25511	0.16816	0.34206	1.2906
burnedge25ha	-0.00202	-0.00276	-0.00128	0.99798
sh400ha	-0.32319	-0.41505	-0.23133	0.72384
Herb25ha	-0.16876	-0.22678	-0.11074	0.84471
cti6ha	0.00846	0.00626	0.01066	1.0085
Blacksage100ha	-0.6985	-1.01011	-0.38689	0.49733
Lek10km	0.0128	0.00478	0.02082	1.01288
tri400ha	0.00957	0.0063	0.01284	1.00961
Sage1m*Winter	0.01915	0.01276	0.02554	1.01934
burnedge25ha*Winter	-0.00246	-0.00364	-0.00128	0.99755
Sage1m*Blacksage100ha	0.00627	0.00283	0.00971	1.00629

**Table 3.11.** Top competing (< 2AIC) summer sage-grouse habitat selection models representing relative use at Stratton Sagebrush Ecological Research Site 2006 - 2009.

Parameters	AIC	df	dAIC	weight
Sage1m + burnedge25ha + hb400ha + sh400ha + Herb25ha + cti6ha + tri400ha + Lek10km + Blacksage100ha + Sage1m*Winter + burnedge25ha*Winter + Sage1m*Blacksage100ha	1997.7	15	0	0.07762
Sage1m + burnedge25ha + hb400ha + sh400ha + Herb25ha + cti6ha + tri400ha + Blacksage100ha + Sage1m*Winter + burnedge25ha*Winter + Sage1m*Blacksage100ha	1998.1	14	0.3	0.06661
Sage1m + burnedge25ha + hb400ha + sh400ha + Herb25ha + cti6ha + tri400ha + Lek10km + Blacksage100ha + Sage1m*Winter + burnedge25ha*Winter	1998.1	14	0.4	0.06443
Sage1m + burnedge25ha + hb400ha + sh400ha + Herb25ha + cti6ha + tri400ha + Lek10km + Sage1m*Winter + burnedge25ha*Winter	1998.2	13	0.4	0.06313
Sage1m + burnedge25ha + hb400ha + sh400ha + Herb25ha + cti6ha + tri400ha + Lek10km + Blacksage100ha + Sage1m*Winter + burnedge25ha*Winter + Sage1m*Blacksage100ha + Lek10km*Blacksage100ha	1998.2	16	0.5	0.06058
Sage1m + burnedge25ha + hb400ha + sh400ha + Herb25ha + cti6ha + tri400ha + Blacksage100ha + Sage1m*Winter + burnedge25ha*Winter	1998.3	13	0.5	0.06036
Sage1m + burnedge25ha + hb400ha + sh400ha + Herb25ha + cti6ha + tri400ha + Lek10km + Blacksage100ha + Sage1m*Winter + burnedge25ha*Winter + Lek10km*Blacksage100ha	1998.8	15	1	0.04609
Sage1m + burnedge25ha + hb400ha + sh400ha + Herb25ha + cti6ha + tri400ha + Sage1m*Winter + burnedge25ha*Winter	1998.9	12	1.1	0.04431

df = degrees of freedom

dAIC = increase in AIC value compared to top model

weight = comparative support for each model

## **CHAPTER 4 : IMPACT OF PRESCRIBED FIRE AND GRAZING TIMING TREATMENTS IN HIGH-ELEVATION SAGEBRUSH ON SONGBIRD HABITAT SUITABILITY**

### **4.1 INTRODUCTION**

Prior to European colonization fire was one of the dominant disturbances within the sagebrush ecosystem (Miller and Rose 1999). Sagebrush steppe occupies much of the northern portion of the Intermountain West from eastern Washington across western Wyoming, northwest Colorado down to northern Nevada and includes segments of Utah and northern Arizona (Knick et al. 2003). Prevailing opinions about historic fire regimes in sagebrush steppe depict high variability in the frequency and severity of fire events which likely produced a diverse mosaic of sagebrush and grassland dominated habitat patches containing multiple successional community stages (Young et al. 1979). Fire suppression activities, along with extensive overgrazing by livestock and invasion events by exotic annual grasses after the late 1800's, initiated substantial changes in plant community composition and structure across sagebrush rangelands (Young and Sparks 1985). Notably, some shrub dominated landscapes have been overtaken completely by annual grasses inducing increased fire frequency (Knick and Rotenberry 1997). New fire rotation intervals in these cheatgrass (*Bromus tectorum*) infested habitats are often too short to allow sagebrush to re-establish dominance in the vegetation community (Whisenant 1990). In other areas, intense grazing by domestic livestock occurred from

the late 1880's through the mid-1900's contributing to drastic reductions in the accumulation of fine-fuels (Young and Sparks 1985). Reduced fuel loads, in conjunction with fire suppression, may have caused a lengthening of fire-return intervals (in areas not already invaded by cheatgrass) resulting in substantially reduced herbaceous cover, increased sagebrush cover, and conifer encroachment (Miller and Rose 1999, Miller and Eddleman 2000, Miller et al. 2000). However, the ability to accurately determine pre-settlement fire return intervals in sagebrush systems using standard estimation methods is often limited by the scarcity of large trees bearing fire scar evidence (Miller and Tausch 2001). As a result, theories about realistic, historic fire return intervals in sagebrush systems remain contentious. Mean composite fire intervals (CFI) calculated from fire-scar evidence located along perimeters of sagebrush habitats and sagebrush recovery rates indicate that historical fire rotations in mountain big sagebrush vegetation communities may have been as short as 35 – 100 years (Whisenant 1990, Welch and Criddle 2003). Others have provided evidentiary support that 70 – 200 years may be a more realistic estimate (Baker 2006).

In the modern era prescribed fire has become a tool used by land managers to reduce sagebrush cover. The earliest historic goals of sagebrush removal programs were to increase herbaceous forage availability for livestock grazing by reducing competition for resources with shrubs (Braun et al. 1976). In recent years the objective of many sagebrush removal programs has shifted to returning these systems to a healthy balance of shrub, forb, and grass components (Arno and Gruell 1983, Miller and Rose 1999). Even more recently, prescribed fire has been advocated as a method for enhancing habitat

quality for greater sage-grouse (*Centrocercus urophasianus*) by increasing availability of important forb species utilized during brood-rearing (Wambolt et al. 2002).

Many mammal and avian species such as pronghorn antelope (*Antilocapra americana*), pygmy rabbit (*Sylvilagus idahoensis*), greater sage-grouse, Brewer's sparrow (*Spizella breweri*), sage sparrow (*Amphispiza belli*) and sage thrasher (*Oreoscoptes montanus*) are dependent on sagebrush habitat for at least part of their life history requirements. Although a substantial amount of recent research has focused on potential fire effects on sage-grouse populations (Martin 1990, Pyle and Crawford 1996, Connelly et al. 2000, McDowell 2000, Nelle et al. 2000), relatively few investigations have documented the effect of this management initiative on migratory songbirds. The small body of published research on this topic has demonstrated positive, negative, neutral and mixed effects on habitat use and fitness components, depending on sagebrush obligate/associated avian species (Rotenberry and Wiens 1978, Castrale 1982, Bock and Bock 1987, Peterson and Best 1999, Holmes 2007).

Land-use practices within the historic geographic range of sagebrush include conversion to agriculture, livestock grazing, natural resource development, and recreation. Changes in land use post-European settlement have resulted in the direct loss of nearly 50% of historic sagebrush habitats (Knick et al. 2003). Livestock grazing continues to be a primary land-use across publicly managed sagebrush habitats. Overgrazing in the early part of the twentieth century altered plant community composition across extensive expanses of sagebrush (McArthur and Plummer 1978). However, information remains incomplete on the impact of grazing timing on these plant

communities, particularly as it translates into effects on sagebrush associated wildlife. Limited literature is available documenting negative responses to early spring grazing for sage-grouse (Gregg et al. 1994, DeLong et al. 1995). This response is interpreted from increased nest predation resulting from removal of residual herbaceous cover which diminishes nest concealment (Beck and Mitchell 2000). Although harmful effects of over-grazing have been documented for songbirds (Bradford et al. 1996), some authors contend that songbird communities in sagebrush habitats are resilient to moderate grazing regimes (Wiens and Dyer 1975, Bradford et al. 1996). However, no information is available on songbird responses to the timing of summer livestock grazing at light to moderate intensity levels.

I present results from an experimental study investigating the effects of prescribed fire and grazing timing treatments on songbird communities within a high-elevation sagebrush ecosystem. My aim was to assess shrub community characteristics resulting from treatments and relate shrub species composition and cover to differences in songbird densities, largely focusing on sagebrush obligate species. I performed a multi-scale analysis of songbird counts for three breeding seasons following prescribed fire and grazing timing treatments. Results presented here are limited to short-term effects (2 – 4 years post-treatment) but provide additional information that may help inform land managers concerned with the impacts of sagebrush removal initiatives on songbird populations within high-elevation sagebrush habitats.

I hypothesized that the density of sagebrush-, and other shrub-obligate nesting bird species would be lower in burn areas immediately following prescribed fire treatments



compared to unburned habitat due to removal of appropriate nesting substrate. I also expected that densities of ground nesting species such as vesper sparrow (*Pooecetes gramineus*) would be low in burn treatments the initial post fire year(s), due to complete removal of ground cover. However, as herbaceous vegetation components began to re-establish, I expected the densities of ground nesting species to increase with time since burn. Densities of shrub nesting birds were also likely to increase with time since burn in fire treatment areas exhibiting a mosaic of burned and unburned big sagebrush that retained an adequate proximal amount of nesting substrate. Because grazing treatments at my study site were initiated after territory establishment each year (late June), I did not expect songbird densities to be directly impacted by the presence of livestock. Therefore, any differences in densities not accounted for by differences in shrub community measurements (burn and unburned) across grazing pastures may be attributable to grazing impacts on unmeasured vegetation components. With this in mind, I expect densities of both shrub and ground nesting species to be lower with later season grazing where reductions in residual herbaceous cover could result in reduced nest site concealment, particularly for ground nesting species. I also hypothesized that prescribed fire would reduce nest survival for shrub nesting species due to lack of shrubs for nest concealment, and that removal of residual herbaceous cover following late summer grazing would negatively affect reproductive success for both shrub and ground nesting species.

## **4.2 STUDY AREA**

The Stratton Sagebrush Ecological Research Site (Stratton) is located 29 km west of Saratoga in south-central Wyoming. Stratton contains 2,250 ha of high-elevation sagebrush habitat and is managed by the Bureau of Land Management (BLM). The average elevation at the site is 2,400 m with terrain characterized by gently rolling hills with 100 m relief between ridge tops and valley bottoms. A riparian drainage (Beaver Creek) runs west to east through a valley bottom bisecting the study area. The area averages 500 mm of precipitation annually with two-thirds resulting from winter snowfall.

Side- and toe-slopes are dominated by shrub communities of mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) with pockets of antelope bitterbrush (*Purshia tridentata*). Vegetation communities on high-slope and ridge tops are dominated by black sagebrush (*A. nova*) and include Wyoming big sagebrush (*A. t.* ssp. *wyomingensis*) as a secondary shrub. Both shrub communities have well-developed herbaceous understory vegetation but herbaceous cover is generally lower within black sagebrush patches.

## **4.3 METHODS**

### **4.3.1 Experimental Treatments and Data Collection**

Separate prescribed fires were implemented by the BLM in each of three grazing units at Stratton in October, 2005. The burns comprised approximately 10% of the study area (295 ha) but were principally restricted to shrub communities where mountain big

sagebrush was the dominant shrub species. Each of the fires resulted in high sagebrush mortality (~85% sagebrush removal) but patchiness of the burns varied considerably between pastures. The burn in the central pasture had a larger, more contiguous pattern compared to burns in the other two pastures (Figure 4.1). The timing of livestock grazing within pastures was manipulated post-fire to incorporate early summer (east pasture), mid-summer (middle pasture), and late summer (west pasture) grazing treatments over four subsequent growing seasons (2006 – 2009). Early summer grazing occurred from late June to early July, mid-summer grazing occurred from late July to early August, and late summer grazing occurred from late August to early September. Actual dates of grazing events within each pasture varied by up to a week, depending on year, with *ca* two week treatment duration. Cattle stocking rates within each pasture were calculated using BLM allotment tabulations of historic production and expected forage conditions and should thus be consistent with standard management practices across most publicly managed sagebrush habitats.

In order to relate differences in songbird distributions to fire and grazing treatments I conducted songbird surveys during the breeding seasons of 2007 - 2009 between 15 May and 25 June. Bird survey line-transect sampling was conducted along 45- 500 m transects. Transects were placed across the study site using a stratified random design. Within the three pastures, five transects were placed in habitat dominated by big sagebrush, black sagebrush and burn sites, respectively, for a total of 15 transects per pasture. Surveys began at least 10 minutes after sunrise and were completed by 10:30 am. Each transect was surveyed by two separate observers for a total of three to five

surveys each year. During each survey, the observer recorded species, distance, and direction for all birds detected parallel to the transect line using a rangefinder and compass. For each survey, observers also recorded covariates likely to affect detectability, including site, observer, observer experience level, survey start time, duration of survey, Beaufort Wind Scale rating (World Meteorological Organization 1970), Julian date, and minimum daily temperature. Surveys were postponed when precipitation exceeded a light drizzle, wind speeds exceeded 19km/hr, or air temperature (cold or hot) noticeably impacted bird activity.

I also examined nest success by conducting targeted nest searches within a 120 by 500 meter area centered on avian survey transects. My primary nest search method was a rope-drag technique (Earl 1950), which consisted of two observers slowly walking parallel to bird survey transects while gently brushing a 30 m rope along the top of the vegetation. Nests were detected when adult birds flushed off the nest as the rope touched the nesting substrate or surrounding vegetation. One-third of the survey transects were searched using this method in 2007 (stratified across treatments), and each survey transect was searched once in 2008 and 2009. The timing of successive nest searches was randomly stratified across treatments, so that each treatment had equal within-season search effort each survey. Nest searches were performed over a period of 63 days (21 May – July 24), 65 days (21 May – July 26), and 37 days (17 May – June 24) in 2007, 2008 and 2009, respectively. Once nests were found I attempted to monitor them on a two to three day interval until the fate of the nest (fledge or fail) could be determined. Data collected during each nest check included nest phase (nest building, incubation or

nestling), number of eggs or nestlings present, estimated age of nestlings (where applicable), and nest fate determined when the nest was not longer active (fledge, fail or unknown).

#### **4.3.2 Variable Development**

Habitat covariates considered included shrub cover and height measured in the field, and habitat variables derived from geographic information system (GIS) data layers. Field measurements considered were line-intercept measurements of shrub cover (including mountain big sagebrush, Wyoming big sagebrush, black sagebrush, and bitterbrush) collected along four 30-meter shrub transects. Line-intercept transects were perpendicularly located along each of the bird survey transects at the 100, 200, 300 and 400 meter marks. I also collected four focal height measurements for each shrub species within 1-m of each line-intercept transect at the 0, 10, 20 and 30 meter marks. I derived GIS habitat variables across the study area from a supervised habitat classification of 1-m resolution color infrared (CIR) National Aerial Photography Program (NAPP) imagery collected in 2001. For a detailed description of the methods used to create these products, see Chapter 3. Data layers considered in my analysis that were derived from this product included percent habitat within 0.25, 1, 6, 25, 100 and 400 hectare neighborhoods. Categories considered were mean percent habitat dominated by big sagebrush, tall shrubs (big sagebrush and bitterbrush), herbaceous habitat (burns or grass dominated habitat), or black sagebrush, all of which were summarized within a 30 meter rounded buffer centered on each bird survey transect. Because sagebrush obligate bird species might be more responsive to the prevalence of big sagebrush habitat components

within a given scale rather than the mean value, I also extracted a majority statistic for percent big sagebrush habitat occurring within the buffer. The majority value within the buffer might be greater or lower than the mean in cases where the range of values within the buffer varies greatly and the standard deviation of the mean is large. Therefore, the majority statistic may better represent prevalence of big sagebrush habitat within certain spatial scales depending on the distribution of habitat patches. For instance, a few very low values may produce a mean that is biased low even though the majority of values within the buffer area are large.

In addition to habitat classification variables, I also considered zonal statistics at the above neighborhood scales derived from a 30-m resolution shrub and herbaceous cover estimation product developed by the United States Geological Survey (Homer et al. in review), as well as compound topographic (CTI) and terrain ruggedness (TRI) indices derived from a 1-m resolution digital elevation model (DEM). Because my focus was on shrub obligate nesting birds, I also calculated the average distance within the buffer to the nearest big sagebrush or tall shrubs (big sagebrush or bitterbrush dominated habitat). My aim in generating this variable was to investigate differential responses within burn treatments associated with distance to appropriate nesting substrate for non-ground nesting species in proximity to survey transects. Distance to tall shrub might be low in unburned habitats with small amounts of big sagebrush but large amounts of bitterbrush (Figure 4.1). Conversely, where the burn pattern of a prescribed fire is increasingly patchy, average intermediate scale big sagebrush habitat might be near zero, but because many small patches were retained the average distance to intact tall shrub patches might

be low. Additional variables included in the modeling process were time since burn (tb), a categorical representation of grazing pasture (early, mid-summer, or late), and an annual precipitation covariate determined to have the most explanatory power for each species based on evaluation of precipitation metrics across different time extents (i.e., February to March). Seasonal precipitation data (fall, winter, spring) was collected at an onsite precipitation gauge and monthly totals were obtained from the National Climatic Data Center (NCDC) for a weather station located nearby in Saratoga, Wyoming. Definitions and scales for all considered covariates are described in Table 4.1.

### **4.3.3 Statistical Analysis**

Wildlife survey counts are readily modeled using a generalized linear model (GLM) with a Poisson or negative binomial error distribution to estimate relative abundance in cases where animals have the same probability of detection across locations (Brand and George 2001). When survey effort and/or detectability varies between plots, a log link function can be used to appropriately account for unequal effort and detectability using an offset term (Buckland et al. 2009). Following Buckland et al. (2009) I used the multiple covariate distance-sampling engine within Program Distance Release 6.0 (Thomas et al. 2010) to model annual detection probabilities by bird species for each survey transect using the candidate set of detectability covariates described earlier. Following visual inspection of detection histograms, counts for each species were right-truncated to eliminate obvious outliers occurring at large distances (Buckland et al. 2001). By pooling detections across all three sample years I had adequate sample sizes to estimate separate global detection functions for four species; Brewer's sparrow, vesper

sparrow, sage thrasher, and green-tailed towhee. I investigated both hazard-rate and cosine detection functions. Selection of the final detection function model for each species was based on an information theoretic approach (Burnham and Anderson 2002) using Akaike's Information Criterion (AIC). I applied each species' detection function to all unique sites to estimate detectabilities and densities, post-stratified by sample year. This allowed me to calculate separate, transect specific, offset values for each survey year. Following the methodology described in Buckland et al. (2009), the offset used in subsequent count regression modeling was calculated as

$$-\log_e \left( \frac{\hat{D}_k}{n_k} \right) \quad (4.1)$$

where  $n_k$  = the number of animals detected (raw count) on transect  $k$  and  $\hat{D}_k$  = the density of animals as estimated by the detection function model (see Buckland et al. 2009). I then estimated the density of each species by modeling the count of individuals detected at each transect as a function of effort, habitat, treatment, and precipitation covariates (Buckland et al. 2009, Russell et al. 2010). In cases with zero detections at a survey location within a particular year, the offset value was adjusted to the mean offset for the entire study site.

When modeling counts, I restricted my analyses to counts that occurred within the effective strip width (ESW) as determined by the distance sampling analysis. Due to the longitudinal panel structure of my data, analyses of relative bird density were conducted using either "xtnbreg" or "xtpoisson" (negative binomial or poisson regression, respectively) commands in Stata 10.0 (StataCorp 2007). Within each of the separate bird



species analyses, all univariates were tested for overdispersion in the response variable using “xtpoisson”. If overdispersion ( $\alpha$  significantly different than 0) for a majority of univariates remained unaccounted for by modeling the longitudinal structure of the data, then I used “xtnbreg” to appropriately account for overdispersion. If overdispersion was not apparent ( $\alpha$  not significantly different than 0) then a Poisson error distribution was retained as the modeling structure.

I initially evaluated the univariate performance of all variables, but allowed only the most explanatory variable (lowest AIC value), and additional variables that were not highly correlated ( $r > |0.7|$ ) across spatial scales to move forward into the final model selection phase. I then compared all subsets of candidate variables. In cases where I used negative binomial regression for final model selection, I also tested the final model using a Poisson model to determine if the inclusion of multiple covariates adequately accounted for the overdispersion observed in the response variable seen in univariate analysis. I then compared AIC values for both error distributions in cases where  $\alpha$  was no longer significantly different than zero for the Poisson model.

I examined nest success using a cox proportion hazards model (Cox 1972) in Stata 10.0 (StataCorp 2007). The hazard model assumes a multiplicative relationship between the underlying hazard rate and a log-linear function of included covariates without making assumptions about the shape of the hazard function (StatSoft, Inc. 2010). Nests with known failure dates were partitioned using Breslow estimation (Breslow 1974) to break ties in nest failure time. The proportional hazard model implies that

covariate effects on survival do not change as a function of time (unless time dependent covariates are specified). Therefore, I tested the proportional hazards assumption using scaled Schoenfeld (1982) residuals as a function of time. Models that violated this assumption were considered to be improperly specified and removed from consideration. Variables examined in nest survival analysis included; estimated nest age, dominant habitat at nest location (big sagebrush, tall shrub, black sagebrush, or herbaceous), proportions of big sagebrush, sagebrush (all species), or herbaceous habitat within 0.25 and 6 ha of nest site, distance to big sagebrush, sagebrush (all species), tall shrub, or shrub (all species) habitat, estimated shrub, big sagebrush, or total estimated shrub cover at nest site (value within 30 m pixel) and within 1 ha of nest site, Julian date, year, and grazing pasture. Separate analyses were conducted for Brewer's sparrow, vesper sparrow, all shrub nesting species combined, and all ground nesting species combined. For each of the group analyses I examined effects of covariates on incubation and nestling phases separately. Nest with unknown fate were right censored back to the last check date where nest status was known.

## **4.4 RESULTS**

### **4.4.1 Distance Analysis**

After removing data points beyond the ESW 2,692 Brewer's sparrow detections, 2,684 vesper sparrow detections, 581 green-tailed towhee detections, and 424 sage thrasher detections remained. A hazard-rate model had the best fit to detections for Brewer's sparrow, sage thrasher, and green-tailed towhee, but a half-normal model

function was best for vesper sparrow (Figure 4.2). Probability of detection for each species varied by observer, and generally decreased with observer experience. In fact, observer was the only covariate retained in the final detectability model for green-tailed towhee (Table 4.2). Detection probability increased with greater minimum daily temperature for Brewer's sparrow and vesper sparrow, and increased with wind speed for Brewer's sparrow and sage thrasher. Detection probability for vesper sparrow decreased with Julian date (late in the breeding season), but increased with Julian date for sage thrasher (Table 4.2).

#### **4.4.2 Count Based Models**

For some bird species, I discovered that either time since burn (tb) or the top ranked precipitation variable had the best univariate performance (lowest AIC). Unfortunately, because my study was limited to three years, precipitation variables were highly correlated with time since burn (tb). For this reason I examined two separate all combination subsets of candidate variables for each species. One subset included time since burn, while the other included precipitation. Thus, both precipitation and time since burn were allowed to compete in final model selection but were not permitted within the same model.

Following selection of the best univariate variable and exclusion of all highly correlated variables, the candidate model set for Brewer's sparrow included average distance to tall shrub habitat (talldist), average height of mountain big sagebrush (vasht), the prevalence of big sagebrush habitat (majority statistic) within a 6 ha neighborhood

(majbs6ha), estimated sagebrush cover (all species) within a 1 ha neighborhood (as1ha), average pixel value (30 m) of herbaceous cover (hb), average compound topographic index within 400 ha (cti400ha), proportion of habitat classified as black sagebrush within 0.25 ha (black.25ha), grazing pasture (early, mid-summer, late), and either time since burn (tb) or April through June precipitation (amj). Distance to tall shrub (talldist) had the greatest explanatory power (lowest AIC) of any single univariate. All models with substantial empirical support (within 2 AIC; Burnham and Anderson 2002) included distance to tall shrub habitat, average height of mountain big sagebrush, proportion big sagebrush habitat, estimated herbaceous cover, time since burn, and late summer grazing (Table 4.3). Three top ranked models (not including the top model) also contained positive, but weak, effects for one of the following variables; proportion of black sagebrush habitat within 0.25 ha, landscape scale compound topographic index, or estimated sagebrush cover within 1 ha. The top AIC-ranked Brewer's sparrow model had 2.4 times the support of the next best-ranked model (Table 4.3). Although the AIC weight of the top model was only 0.34, all other top ranked models contained all variables present in the top model plus one additional variable. The top model indicated that bird density declined by 8% with each 10 meter increase in distance from tall shrubs (Table 4.4). There was also positive selection for average height of mountain big sagebrush (1.1% per cm mountain big sagebrush height), herbaceous cover (4.2% per % herbaceous cover), intermediate scale prevalence of sagebrush habitat (0.67% per majority statistic for big sagebrush habitat), and time since burn (7.9% per year). Exponentiated coefficients for the top model also suggest a grazing pasture effect

resulting in a 44% greater density in the late grazing pasture compared to early and mid-summer.

Variables used in development of candidate models for sage thrasher included average distance to big sagebrush habitat (bsd<sub>dist</sub>), average maximum shrub height measured along line-intercept transects (max<sub>ht</sub>), grazing pasture (early, mid-summer, late), a majority statistic for big sagebrush habitat within 6 ha (maj<sub>bs6ha</sub>), estimated sagebrush cover within 400 ha (as<sub>400ha</sub>), estimated herbaceous cover within 6 ha (hb<sub>6ha</sub>), average compound topographic index within 400 ha (cti<sub>400ha</sub>), and either time since burn (tb) or total precipitation from December to February (dec<sub>feb</sub>). Despite being the most predictive univariate variable, distance to big sagebrush habitat was only present in two of the top five sage thrasher count models (Table 4.3). Interestingly, increased density with greater winter precipitation appeared in all the top ranked models (within 2 AIC). Other variables appearing in a majority of top ranked models ( $\geq 5$ ) indicated thrasher densities increased with greater maximum shrub height, higher prevalence of intermediate scale big sagebrush habitat, and larger field measurements for tall shrub cover (combined measure of mountain big sagebrush and bitterbrush). Additional variables that each appeared in one of the top models included positive effects for landscape scale sagebrush cover (as<sub>400ha</sub>) and patch scale herbaceous cover (hb<sub>6ha</sub>). Overall support for the top sage thrasher model was only 1.4 times greater than other top models and was much weaker than the top model for the other avian species investigated. Despite this limitation, coefficients for covariates in the top model were similar in all models where they were represented (Table 4.5). These coefficients reflected a 0.9%

increase in thrasher density for each additional cm of maximum shrub height, a 0.4% increase per 1% increase in prevalence of big sagebrush habitat within 6 ha, a 0.4% increase with each increase in landscape scale compound topographic index, a 2% increase per inch of winter precipitation, and a 0.3% decrease in density for each 10 m increase in distance from big sagebrush habitat. The apparent pasture effect found in the top Brewer's sparrow model was absent from the top sage thrasher model. However, weak pasture effects (confidence intervals overlapping zero) did occur in four of the top ranked models and indicated lower densities in the mid-summer grazing pasture compared to the late grazing pasture.

Univariate analysis suggested that time since burn was the best predictor of green-tailed towhee density, with a 45% increase in density each subsequent year after the prescribed fire. Covariates permitted to compete along with time since burn, or an alternate candidate set containing precipitation (April through June) were field measurements for combined mountain big sagebrush/bitterbrush cover (*purvascov*), average estimated sagebrush cover (both big and black) within 6 ha (*as6ha*), average compound topographic index within 0.25 ha (*cti.25ha*), grazing pasture, and a quadratic effect of big sagebrush habitat prevalence within 6 ha (*majbs6ha*). All five of the top ranked models for this species (Table 4.3) contained positive coefficients for local scale tall shrub cover (big sage and bitterbrush), and intermediate scale sagebrush cover. The quadratic term for intermediate scale big sagebrush habitat appeared in four of the top five models (Table 4.6), including the top model. Coefficients for the quadratic effect of big sagebrush habitat in the top model produce a parabolic curve indicating selection for

intermediate levels of big sagebrush habitat. Four of the top models also displayed the positive effect of time since burn, including the top model, with density increasing each subsequent year after prescribed fire. Although time since burn appeared to be a better predictor than precipitation for this species, precipitation did appear in the second highest ranked model (Table 4.3). Additional parameters each appearing in one of the top models (but not the top model) were positive effects of local scale compound topographic index (0.25 ha), and greater density in early (42%) and mid-summer (35%) pastures compared to the late grazing pasture. While models containing compound topographic index and grazing pasture effects were within 2 AIC of the top model, confidence intervals for these additional parameters overlapped zero in the models where they were represented, indicating weak effects.

The top univariate for vesper sparrow was not a habitat variable but instead a negative response to increased winter precipitation. Candidate variables available for vesper sparrow model development included total shrub cover measured along line-intercept transects (totalcov), average proportion of big sagebrush habitat within 6 ha (bs6ha), estimated shrub cover within 400 ha (sh400ha), estimated herbaceous cover within 1 ha (hb1ha), and either total precipitation between the months of October and February (octfeb) or time since burn. There were two top ranked models for this species (< 2 AIC; Table 4.3). Both of these models contained negative effects for increasing winter precipitation (top model: 2% decrease in density per inch of winter snowfall), local shrub cover (2.7% decrease per 1% increase in cover), and landscape scale shrub cover (6% decrease per 1% increase in cover). Densities decreased in response to

measures of shrub cover, but there was a positive effect for intermediate scale big sagebrush habitat (0.6% increase per 1% increase in proportional habitat). Both of the top models also indicated that density varied by grazing pasture. In the top model the mid-summer grazing pasture had 12% lower bird density than pastures subjected to either early or late grazing strategies (Table 4.7). The only variable included in the second highest ranked model that did not appear in the top model was a positive effect of herbaceous cover within 1 ha.

#### **4.4.3 Nest Success**

My nest searching effort resulted in the detection of 46, 104, and 200 nests in 2006, 2007, and 2009, respectively. Only a few nests were found during the nest building phase (21 in total). Therefore, I only retained these nests for analysis if they were observed to move into the incubation phase (8 of 21). Thus, inference on nest survival is based on incubation through fledging. Most of the nests used for this analysis belonged to Brewer's (21, 51, and 76 nests in 2006, 2007, and 2009 respectively) or vesper (9, 36, and 98) sparrows. Combined totals across all three years for nests belonging to other species were two for Brewer's blackbird (*Euphagus cyanocephalus*), 14 for green-tailed towhee, 11 for sage thrasher, 16 for horned lark (*Eremophila alpestris*), and one for lark sparrow (*Chondestes grammacus*). Nests were found in equal proportions across the three grazing pastures in 2008 and 2009. However, because only two transects per unique treatment (habitat X grazing) were searched in 2007, the number of nests detected in the early grazing pasture was three times greater than both the mid-summer and late summer grazing treatment pastures for that sampling year. I found 2 –



3.3 times more nests (ground and shrub nests) in unburned big sagebrush than in burn treatments across the three years of data collection.

With only a few exceptions, the variables I investigated had weak effects on nest survival for Brewer's sparrow, vesper sparrow, shrub nesters, or ground nesters. However, for all shrub nesting species combined, the risk of nest failure was strongly influenced by nest age, decreasing with age during the incubation phase (19% per day, CI = 5 – 31%). However, when survival of Brewer's sparrow nests during incubation was tested the assumption of proportional hazard across time was violated. While nest age did not have a strong effect on survival of ground nests, the hazard during the incubation phase was greater than the nestling phase for both vesper sparrow (80%, CI = 41 – 93%) and ground nesting species combined (51%, CI = 8 - 83%). None of the habitat or treatment covariates investigated appeared to affect nest survival when considering individual species or substrate nesting group during either the incubation or nestling phases.

#### **4.5 DISCUSSION**

As I hypothesized, the density of all four species examined in my study, including the ground nesting species (vesper sparrow), displayed negative relationships to removal of big sagebrush habitat. Both of the sagebrush-obligate species, sage thrasher and Brewer's sparrow, had dramatically lower densities with removal of big sagebrush habitat. These species had 3.9, and 6.7% lower density, respectively, with each 10% loss in big sagebrush habitat at an intermediate spatial scale (6 ha neighborhood). In areas

where the burn pattern resulted in a larger, more contiguous, burn this translated into near complete avoidance by both of these species. While the incidence rate ratio for this effect is larger for Brewer's sparrow, the average density of this species at my study site across all survey years was much greater than sage thrasher (0.82 per ha versus 0.05). Because sage thrasher occurred at dramatically lower densities, a given percent reduction in density translates into a larger impact on the study site-wide population of this species than a similar percent reduction in Brewer sparrow. Thus, the response to intermediate scale loss of big sagebrush can be characterized as more pronounced for sage thrasher. Additionally, the strong negative response of Brewer's sparrow to increased distance to nearest tall shrub habitat (7.8% per 10 m) suggests that the extent of sagebrush removal may be somewhat mediated for this species when an adequate number of small shrub islands are retained, providing suitable nesting substrate, although the consequences of these removals on fitness (survival and reproduction) could be negative. Conversely, patches of remaining habitat left after burns were insufficient to maintain higher sage thrasher densities, which decreased 3.2% for every 10 m increase in distance from big sagebrush habitat. Impacts of big sagebrush removal (proportion of habitat) and distance to nearest unburned big sagebrush indicate proximity of unburned habitat is important to Brewer's sparrow, while retention of a larger proportion of unburned habitat is more important for sage thrashers. This difference in response is likely attributable to the much larger average territory size requirements of sage thrasher (0.52 ha for Brewer's Sparrow versus 1.14 ha for sage thrasher; Reynolds 1981).

Local scale shrub cover was an important univariate predictor for all species examined in my analyses. Green-tailed towhees typically select nesting sites with structurally dense vegetation (Jehle et al. 2006). Indeed, the top AIC-ranked model predicted a 5% increase in towhee density with each one percent increase in tall shrub cover. Conversely, vesper sparrow density was negatively associated with both local scale total shrub cover (declined by nearly 2% with each 1 % increase in percent cover) and landscape scale shrub cover (declined by 6% with each 1 % increase in cover) despite being positively associated with greater intermediate scale big sagebrush habitat (increasing by nearly 1% with each 1 % increase in habitat dominated by big sagebrush). As a ground nesting species, vesper sparrows have been shown to prefer areas with lower shrub cover (King 1968). However, this species is known to select habitat where scattered tall shrubs provide elevated singing perches for territorial males (Castral 1983). This explains why big sagebrush removal at intermediate scale might result in reduced habitat selection by this species despite preference for lower shrub cover.

Similar to the green-tailed towhee, univariate analysis for both sage thrasher and Brewer's sparrow indicated a positive association with local scale tall shrub cover. However, field measurements for cover and height were highly correlated  $|r > 0.80|$ . Based on my univariate selection process average maximum shrub height and mountain big sagebrush height were much better predictors for sage thrasher and Brewer's sparrow, respectively, with ~ 1% increase in density for each 1 cm increase in height. Thus, even though local scale shrub cover measurements were not allowed to compete in the model selection process, tall shrub cover is likely important for both of these species. This

would agree with Castrale's (1982) findings where sage thrasher territories in north central Utah were only found in sagebrush removal treatments where habitat contained taller shrub remnants. Adequate shrub height has also been indicated as an important factor in green-tailed towhee nest site selection (Jehle et al. 2006).

My results indicate that both local and intermediate scale disturbances can have large consequences in habitat selection for sagebrush and shrub obligate songbirds. In a recent review paper, Knick et al. (2005) inferred from the available literature that densities of sagebrush obligate bird species were largely unaffected by sagebrush removal initiatives that resulted in a < 50% reduction in sagebrush cover. My results disagree with these findings. Instead, my findings agree with other investigations that have documented dramatically reduced densities for sagebrush obligates when only a small number of intact continuous sagebrush patches are retained within the landscape (Peterson and Best 1987, Knick and Rotenberry 1999) or when remaining patches are of inadequate size to support a breeding pair (Holmes 2007). Other authors have described direct reductions in habitat quality for sage thrashers and Brewer's sparrows, even with small prescribed fires (Castrale 1982, Kerley and Anderson 1995).

Across the study site, both Brewer's sparrow and green-tailed towhee densities increased with time since the burn. On average, Brewer's sparrow increased by 7.6% and green-tailed towhee increased by 45% each year after the burn. Proportional increases in Brewer's sparrow and towhee densities are not directly comparable because over the entire study period towhee densities were substantially lower than those of Brewer's sparrow (0.1 – 0.21/ha versus 0.76 – 0.9/ha) . Despite this, the time since burn effect

appears to indicate that the negative effects of sagebrush removal at my study site diminished somewhat with time for both of these species. This effect could have resulted from recovery of local scale herbaceous cover across burn transects over the study period. In the first post-fire spring season (which coincides with the timing of initial territory selection) areas within the burn perimeter were virtually devoid of vegetation. Additionally, these areas had very little vegetative re-growth during the second spring season. However, in the last two years of the study (3 & 4 years post burn) there was a noticeable increase in spring herbaceous cover (Erickson, pers. obs.), particularly that of tall annual forbs such as lupine (*Lupinus spp.*) and thistle (*Cirsium spp.*), and several Brewer's sparrow nests were found in lupine during the later half of the breeding season both of these years. My predictive covariate for herbaceous cover was a static estimate derived from imagery acquired the first summer after the fire (Homer et al. in review). Because herbaceous cover can vary significantly from year to year, responding to both precipitation and removal of residual vegetation by grazers, the predictive capability of this variable may have been limited in comparison with measures of sagebrush cover, which generally display negligible change the first several years after a fire (see Chapter 2, on vegetation). Some of this variation may have been captured by the time since burn and precipitation covariates but much of this variation likely remained unexplained by my models.

Lack of a time variant measure of herbaceous cover might also have contributed to the inclusion of precipitation variables in some of the top models. Winter precipitation was a strong predictor in top AIC-ranked models for two of the investigated bird species

(vesper sparrow, sage thrasher). Spring precipitation increased over the study period, and hence, with time since burn, impacting herbaceous production in both burned and unburned habitats. The high correlation between time since burn and both winter (negative) and spring (positive) precipitation during the study period precluded my ability to examine the additive effects of these two variables on bird density responses. The negative effect of winter snowfall on a ground nesting species, such as vesper sparrow, might be expected. Greater than average winter snow accumulation could force individual birds to select territories at lower elevations where snow melt occurs earlier. However, variation in precipitation is also a strong driver of annual fluctuations in herbaceous and shrub cover (Rickard 1985, Bates 2004) which might explain increases in sage thrasher density with greater winter precipitation (December through February) in combination with landscape scale compound topographic index (400 ha). Increases in winter moisture in more mesic big sagebrush areas (larger CTI values) generally results in greater leaf production (greater shrub cover), which could make these sites more attractive for shrub nesting species during territory establishment and nest site selection.

In addition to quantifying responses in bird densities to shrub removal, my experimental design attempted to examine differences in the effects of concurrent grazing timing treatments. I attempted to derive habitat variables that might help account for inherent differences in habitat and burn patterns across grazing treatments. However, the possibility remains that the pasture effect observed for two of the species investigated may have been explained if additional habitat variables describing differences in habitat characteristics between pastures had been considered. As evidenced in Figure 4.1,

habitat characteristics within pastures varied greatly, not only as a result of variable burn pattern and extent, but also within unburned sagebrush habitats. I had hoped that including a variety of habitat metrics at multiple scales would properly account for these differences and allow me to make inference concerning population density differences across pastures attributable to grazing effects. No pasture effects were evident in the top models for sage thrasher or green-tailed towhees. However, a pasture effect was observed for both Brewer's and vesper sparrow. The top Brewer's sparrow model indicated a 44% greater density in the late grazing pasture compared to early and mid-summer (Table 4.4.3). Contrary to my expectations, removal of residual cover with late grazing did not reduce habitat selection. It is possible that the low grazing levels (light to moderate intensity) achieved by my study were insufficient to elicit lower nest site concealment that might be expected with high grazing intensity. In contrast, substantial amounts of apical meristem removal during the early growing season may lead a more compact growth forms for some perennial graminoid species (Blaisdell and Pechanec 1949, Hyder and Sneva 1963, Hyder 1972), particularly in semi-arid environments such as my study site, where low summer precipitation limits regrowth potential. This type of altered growth may reduce habitat suitability by decreasing effective nest site concealment. This is a possible explanation for Brewer's sparrow reduced preference for the earlier grazing pastures.

I found 12% lower vesper sparrow density within the mid-summer grazing pasture (Table 4.4.6) compared to the other two grazing treatment pastures. Mid-summer is when many warm season grasses and forbs at high elevation are investing substantial

resources into seed production. Heavy defoliation during the flowering stage can result in an 89% reduction in flowering stems for species such as bluebunch wheatgrass (*Agropyron spicatum*, Pitt 1986). This could reduce recruitment rates and lengthen recovery time for herbaceous components after a prescribed fire. If this pressure remains constant over multiple years, it may take much longer for herbaceous cover to reach cover levels suitable for effective nest site concealment and foraging activities within burn treatments. However, a possible complication to the interpretation of mid-summer grazing is the larger, more contiguous burn pattern in that pasture compared to the other pastures at the study site. If seed production was indeed impacted by grazing it might result in slower recovery of the burn treatment within this pasture. Levels of unconsumed seed production may not have been large enough to quickly propagate to the interior of this particular burn. Despite potential explanations for differential densities across grazing pastures for both Brewer's sparrow and vesper sparrow, the disproportionate burn patterns across my grazing treatments somewhat confounds interpretability of grazing timing treatments.

Despite limitations in interpretability of the grazing portion of my study, I have clearly demonstrated that sagebrush removal negatively affected passerine habitat selection at my study site. However, caution has been advised when inferring habitat quality from population densities alone without addressing impacts on fitness components (Van Horne 1983). Negative treatment impacts on nest success sometimes require a short-term lag time before effects become apparent in habitat selection, particularly for species with high site fidelity (Haas 1998). In my study, habitat and



treatment covariates had no effect on nest survival for Brewer's sparrow and vesper sparrow or for combined substrate nesting groups (shrub or ground nesting species). Even though nest success was not directly affected by measured variables, habitat selection was dramatically impacted. My results show lower densities in burn treatments dependant on the magnitude of big sagebrush habitat removal.

The absence of fitness consequences, despite differences in densities between different habitat patches, can be explained by a species distribution theory proposed by Fretwell and Lucas (1969). According to the "ideal free distribution" theory, individuals within a population will sort themselves across habitats in a manner that maximizes the number of rewards available to each individual. The realized suitability (or number of rewards per individual) of habitat patch  $i$  ( $S_i$ ) is a function of its "basic" suitability ( $B_i$ ), as determined by the quality/quantity of habitat components meeting the requirements of the species' niche, minus some function ( $f_i$ ) that reduces habitat suitability as the density ( $d_i$ ) increases;

$$S_i = B_i - f_i(d_i), i = 1, 2, \dots, N. \quad (4.2)$$

Due to within patch differences in habitat components available to wildlife populations it can be assumed that no two habitats will have equal "basic" levels of suitability. As a result, when two habitat patches are compared, one patch will have greater "basic" suitability than another (e.g.,  $B_1 > B_2$ ). However, individuals within the population may select areas with lower "basic" habitat suitability,  $B_2$ , when habitats with greater "basic" suitability,  $B_1$ , already have a large number of established individuals or breeding

territories. Thus, the addition of one more individual/territory within habitat 2 confers a realized habitat suitability ( $S_2$ ) that is now equal to habitat 1 ( $S_1$ ), due to density dependant regulation of habitat suitability. If this theory of population distribution holds true, then realized habitat suitability within the two patches will naturally balance at population densities that produce an equilibrium of realized habitat suitability ( $S_1 = S_2$ ), despite greater density in the habitat with greater “basic” suitability ( $d_1 > d_2$ ). Territorial behavior by birds with established territories has been proposed as the mechanism that provides a previously un-established individual (e.g., a late migrant) with a density assessment cue (or “density index”; Fretwell and Lucus 1969). This cue provides a newly arriving individual with information about current density levels and allows it to select a habitat where chances of breeding success are greater based on current population density dependant conditions (Kluyver and Tinbergen 1953). If individuals in my study are sorting themselves according to the “ideal free distribution”, reproductive success rates should, on average, be equal across habitats with disparate “basic” suitability despite the strong differences in habitat selection resulting in unequal densities. Because nest success was unaffected by either fire or grazing treatments at my study site, densities of individual bird species may adequately reflect differences in “basic” habitat quality across treatments, as evidence by strong selection patterns.

Ideally, I would have conducted a true Before-After Control-Impact (BACI) designed experiment (Green 1979) to assess bird densities and nest success prior to implementing treatments. This would have allowed me to account for inherent differences in response variables between grazing pastures and further elucidate the

differences noted between grazing pastures, given that grazing pressure was relatively low in my grazing treatments. Data collected in burn areas prior to burn treatments would also have allowed me to directly assess before/after changes in avian density without the need to infer effects from comparisons between burned and unburned habitats. Despite this limitation in my experimental design, I believe my comparison of burns and unburned sagebrush across grazing treatments provides useful information that can inform management decisions.

#### **4.6 MANAGEMENT IMPLICATIONS**

The results of my experiment imply that the timing of grazing in high elevation sagebrush systems may have an impact on some components of the avian community. Brewer's sparrow displayed stronger selection for habitats within the late summer grazing pasture; even after many of the differences in shrub habitat composition across pastures were examined. Conversely, vesper sparrow displayed moderate avoidance of habitats subjected to mid-summer grazing. If these are indeed real grazing effects and not the product of un-modeled differences in shrub habitat distribution or complexity between pastures, then I would expect the addition of an annual measure of herbaceous cover and height, included as continuous variables, might provide clearer indications of grazing effects compared to the simple categorical representations of grazing treatment examined here. Indeed, range condition, as measured in percent cover of climax species has been found to influence habitat selection for avian species in sagebrush ecosystems (Vander Haegen et al. 2000). Unfortunately, the collection and processing time required for traditional herbaceous cover assessments, such as vegetation clipping, are often quite

time consuming, and adequate sample sizes for this type of analysis can be cost prohibitive. Additionally, estimates produced by less experienced observers can vary substantially (i.e., ocular assessment) introducing considerable bias (Stohlgren et al. 1998). Newer techniques using object based image analysis of on site digital photos using GIS and image analysis software will likely allow future investigative efforts to include more meaningful, time variant herbaceous cover estimates in analysis of grazing effects on herbaceous cover (see Luscier et al. 2006). Indeed, I suggest that this or some other type of higher precision analysis of herbaceous characteristics (i.e., cover) may be required if meaningful results are to be discovered related to grazing effects on passerine distributions and nest success in sagebrush habitats.

Publicly managed sagebrush habitats are used extensively as rangeland. Yet, to date, very little research has been conducted on the effects of grazing timing on sagebrush-associated songbirds, particularly related to long-term effects. Therefore, I believe we still lack an adequate understanding of both the short and long-term implications of grazing timing strategies for songbirds that require sagebrush habitats. However, based on the results of my investigation, I suggest that grazing prior to herbaceous seed set should be deferred within burn treatments in high elevation mountain big sagebrush communities until herbaceous components have had a chance to reestablish. On the other hand, late August grazing appeared to have little effect on songbird densities at my study site. This implies that late summer/early fall grazing by livestock may be compatible with retention of larger songbird breeding populations of other high quality habitat components are present.

Bird species examined in my analyses included both shrub/sagebrush and ground nesting species. While each of these species displayed distinct distributional relationships to shrub community composition and structure, each was negatively associated with reductions in big sagebrush habitat within a 6 ha neighborhood. Other research efforts suggest that negative effects of prescribed fire on sagebrush-associated bird species can be minimized by implementing a mosaic patterned burn that retains a large amount of sagebrush cover (Peterson and Best 1987, Miller and Rose 1999). In my study, Brewer's sparrows occurred within burn treatments that retained small scattered patches of big sagebrush habitat. However, all species investigated, including the ground nesting vesper sparrow, showed dramatic declines with big sagebrush removal at the 6 ha scale, which is smaller than sagebrush removal scales investigated by many other research efforts (widely ranging from 12.5 - 2462 ha: Peterson and Best 1987, Noson et al. 2006, Holmes 2007, Bates et al. 2009). Effects of sagebrush removal at these small spatial scales on sagebrush obligate and associated species observed in my study suggest limited benefit of prescribed burns for ground nesting species. In contrast, wide application of prescribed fire may be detrimental to the long-term viability of sagebrush obligate songbird populations. Despite similar nest success rates for shrub nesting species in burned and unburned big sagebrush habitats, densities for Brewer's sparrow were extremely low in burns compared to unburned big sagebrush (annual averages = 0.6 – 0.7/ha versus 1.5 – 1.7/ha). As a result, given an equal number of fledglings per nesting effort, sagebrush habitats subjected to prescribed fire treatments are producing 2.1 – 2.8 fewer offspring than unburned big sagebrush habitats. This could have incredible

implications related to range-wide declines in Brewer's sparrow populations, as reported in recent Breeding Bird Survey (BBS) data (2.1% decline/year between 1966 - 2007; Sauer et al. 2008). Based on my results, I suggest that prescribed fire in big sagebrush vegetation communities may be a large contributor to these declines, as prescribed fire treatments were implemented annually on 251 – 764 square kilometers of sagebrush habitat managed by the BLM from 1997 – 2002 (Connelly et al. 2004).

Prescribed fire in sagebrush habitats has been used in an attempt to enhance sage-grouse habitat, increase production of herbaceous biomass and restore proposed historic fire regimes (Bunting et al. 1987, Pyle and Crawford 1996, Crawford et al. 2006). The effectiveness of prescribed fire as a tool for sage-grouse habitat enhancement has recently been called in to question (Connelly et al. 2004). While some studies have demonstrated short term increases in forbs that provide an essential food source during brood-rearing (Pyle and Crawford 1996) other studies related to habitat selection have shown a near complete avoidance of these habitats by sage-grouse (see Dahlgren et al. 2006, and Chapter 3 of this thesis). Additional studies have reported inconclusive or contradictory evidence on the effects of fire on other components of sage-grouse habitat such as lek site attendance and winter habitat use (Ricker 1970, Fisher et al. 1996, Nelle et al. 2000). Historic fire return intervals in sagebrush habitats may have been dramatically underestimated and were likely 70 – 200 years or more in mountain big sagebrush ecosystems (Baker 2006). Thus, opinions prompting the frequent use of fire as a management tool to restore historic fire regimes in these systems may need to be re-evaluated. Many research efforts on the effects of prescribed fire on herbaceous biomass

have determined that when forage enhancement does occur, any increases in livestock forage are generally short-lived. In fact, herbaceous production often reverts to pre-burn levels within three to five years after the fire event (Bunting 1985), but often the duration of increased production is even shorter (i.e. 1 year or less; see Chapter 2). Even when increased livestock forage is achieved, burns remain devoid of appreciable sagebrush cover for at least an additional 30 years depending on the system (Harniss and Murray 1973, Wambolt et al. 2001). Future research should examine how smaller burns that leave a patchy mosaic of sagebrush 'islands' might affect sagebrush obligate bird species, and whether beneficial habitat manipulations for livestock grazing that do not result in strong negative effects for songbirds can be achieved.

Until further research can adequately establish the validity of the proposed benefits of prescribed fire within sagebrush habitats its use as an effective management tool will remain in question. I propose that long-term monitoring of vegetation and wildlife responses should be initiated using a BACI design (Green 1979) within an adaptive management framework (Walters 1986, Aldridge et al. 2004), if further understanding is to be gained. If beneficial short-term effects of prescribed fire can indeed be more clearly documented, land managers should still weigh the long-term negative impacts on sagebrush-associated passerines when planning prescribed fire projects. In rangeland systems where increased herbaceous production after prescribed fire has been documented and advocated for livestock benefit, the costs to the ecosystem may still outweigh the short-term economic benefits. Implementing a prescribed fire that retains an adequate amount of cover and number of patches for sagebrush obligate birds

is difficult to properly implement and, more than likely, cost prohibitive. Based on my results and those of other recent studies (Nelle et al. 2000, Holmes 2007) I suggest that land managers proceed with caution when considering the use of prescribed fire for habitat enhancement in high elevation sagebrush habitats. At the very least, prescribed fires should be limited in management units where conservation of sagebrush-obligate songbirds is a concern.

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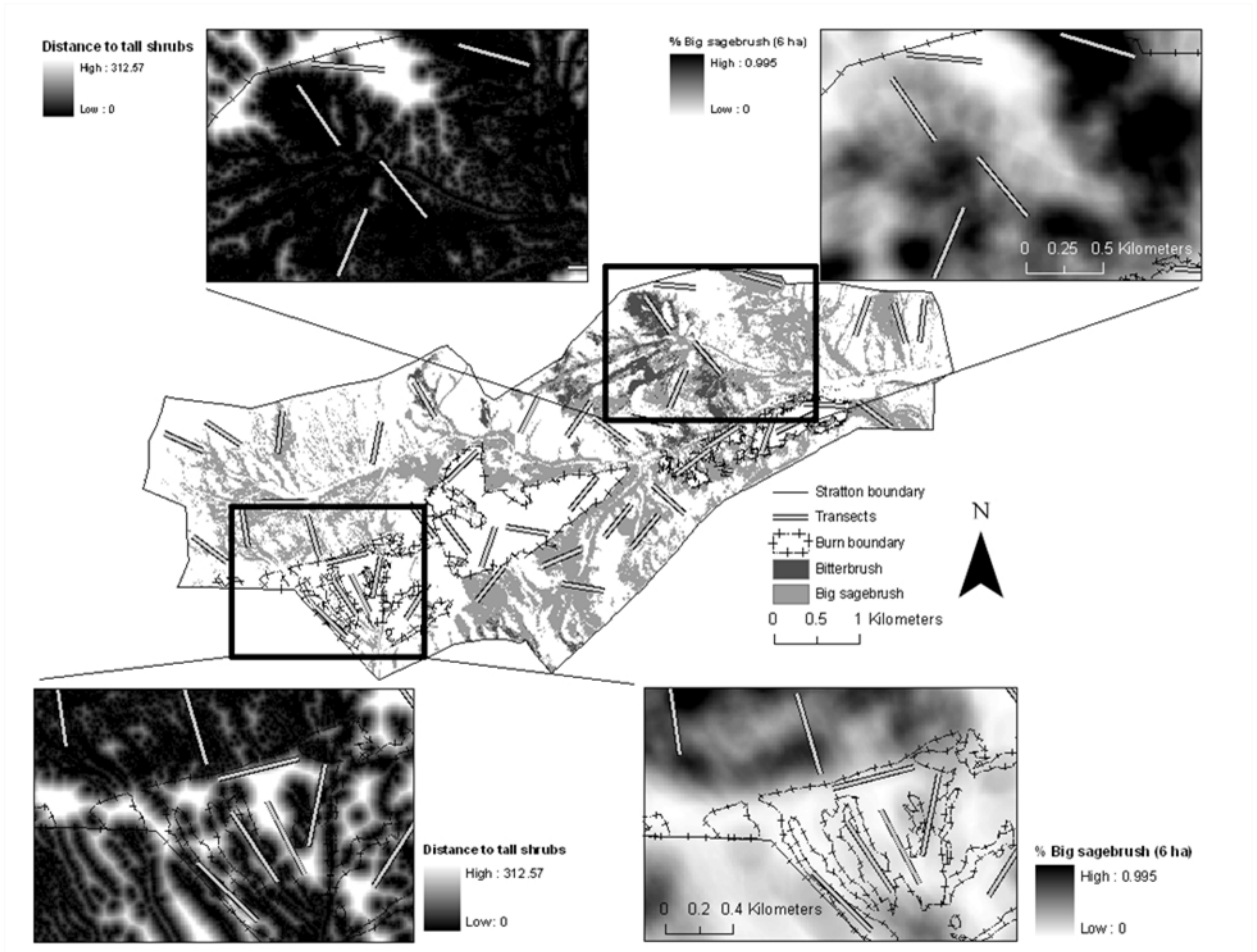
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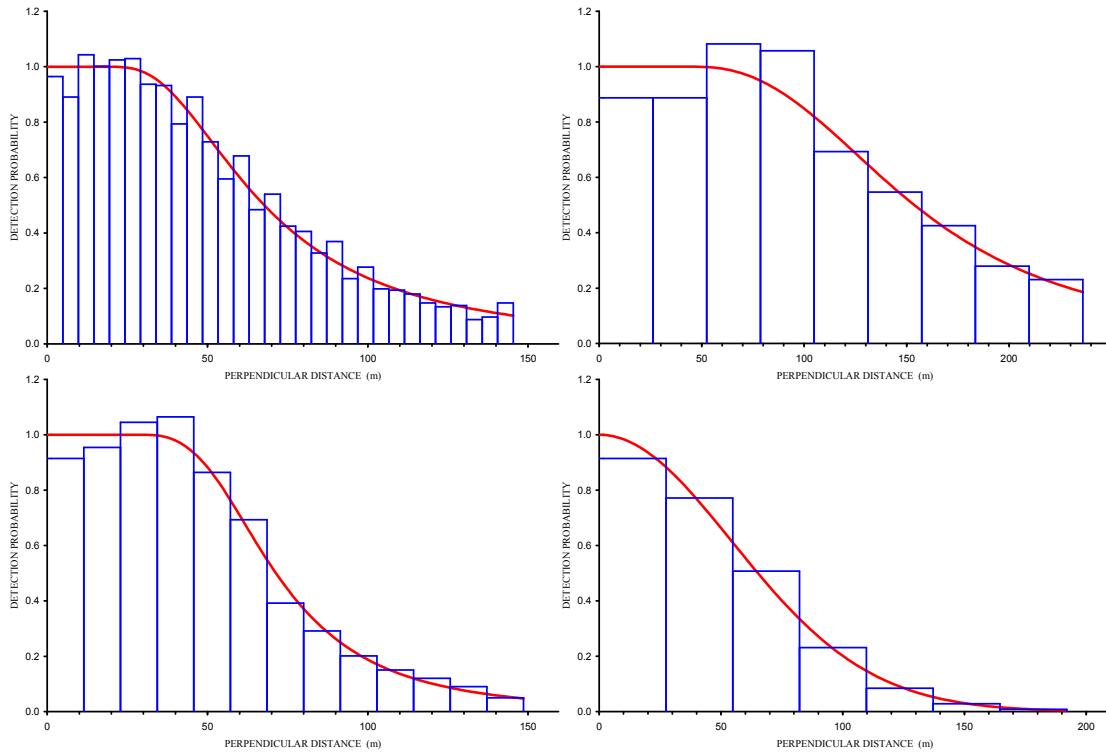
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**Figure 4.1.** Distribution of tall shrub habitat at Stratton Sagebrush Ecological Research Site (Stratton). Insets portray examples of derived variables for distance to tall shrub habitat and average percent habitat dominated by big sagebrush vegetation within unburned sagebrush habitats (top) and a mosaic patterned prescribed burn (bottom). Variables portrayed in insets include straight-line distance to tall shrub habitat (left) and percent habitat with 6 ha classified as big sagebrush (right). These variables were generated from a 1-m resolution supervised classification of color infrared National Aerial Photography Program (NAPP) imagery.



**Figure 4.2.** Estimated detection functions for Brewer’s sparrow pooled across observer, wind speed, and minimum temperature (top left), sage thrasher pooled across observer, Julian date, and wind speed (top right), green-tailed towhee pooled across observer experience (bottom left) and vesper sparrow pooled across Observer, Julian date and minimum temperature (bottom right). All detection curves represent data pooled over three years of surveys (2007 – 2009).



**Table 4.1.** Definitions for variables representing field measurements, treatment categories, and derived GIS characteristics measured within a 30 m buffer centered on bird survey transects and at individual nest locations at Stratton. Species column indicates individual species (BRSP = Brewer’s sparrow, SATH = sage thrasher, GTTO = green-tailed towhee, and VESP = vesper sparrow) for which each variable, at most predictive scale, was allowed to compete in count based model selection.

Variable Name (Scale Names)	Variable type	Variable definition	Species
Pasture	Treatment	Categorical representation of grazing timing treatment (early, mid-, or late summer)	All
tb	Treatment	Time since burn treatment	All
octfeb	Field measurement	Sum of annual precipitation from the months of October through February	VESP
decfeb	Field measurement	Sum of annual precipitation from the months of December through February	SATH
amj	Field measurement	Sum of annual precipitation from the months of April through June	GTTO
purvascov	Field measurement	Average line-intercept measurement for mountain big sagebrush and antelope bitterbrush across four 30 meter transects per survey location	SATH, GTTO
totalcov	Field measurement	Average line-intercept measurement for all shrub species across four 30 meter transects per survey location	VESP
bigstage (1 m, 0.25, 1, 6, 25, 100, 400 ha)	*GIS: Habitat classification	Proportion of habitat classified as dominated by big sagebrush vegetation within 30 meters of survey transects	VESP
majbs (1 m, 0.25, 1, 6, 25, 100, 400 ha)	*GIS: Habitat classification	Zonal majority statistic representing proportion of big sagebrush habitat within 30 meters of survey transects	SATH, BRSP, GTTO
bsdists	*GIS: Habitat classification	Straight line distance to nearest habitat classified as dominated by big sagebrush vegetation	SATH
talldists	*GIS: Habitat classification	Straight line distance to nearest habitat classified as dominated by tall shrub vegetation	BRSP
as (30 m, 0.25, 1, 6, 25, 100, 400 ha)	**GIS: Cover estimate	Estimated sagebrush cover (all sagebrush species)	SATH, BRSP, GTTO
sh (30 m, 0.25, 1, 6, 25, 100, 400 ha)	**GIS: Cover estimate	Estimated shrub cover (all shrub species)	VESP
hb (30 m, 0.25, 1, 6, 25, 100, 400 ha)	**GIS: Cover estimate	Estimated herbaceous cover	SATH, BRSP, VESP
cti (1 m, 0.25, 1, 6, 25, 100, 400 ha)	GIS	Compound topographic index	SATH, BRSP, GTTO

\*Variable derived from a supervised classification of 1 meter resolution NAPP imagery.

\*\*Variable derived from a habitat cover estimation produced by the USGS (Homer et al. in review)

Note: Available scales for GIS neighborhood variables = cell value (1 m for habitat product or 30 m for cover product), 0.25, 1, 6, 25, 100, and 400 ha

**Table 4.2.** Covariates and model performance scores (dAIC = increase in AIC) for the top three detection function models for Brewer’s sparrow, vesper sparrow, green-tailed towhee and sage thrasher. Potential covariates included observer, observer experience (categorical), Beaufort wind speed, minimum daily temperature, Julian date, start time and survey duration.

Species	Key function	Continuous covariates	Factor covariates	dAIC
Brewer's sparrow	Hazard-rate	Wind (Beaufort), minimum temperature	Observer	0
	Hazard-rate	Wind (Beaufort), minimum temperature, julian date	Observer	1.39
	Hazard-rate	Wind (Beaufort), minimum temperature, survey start time	Observer	1.91
Vesper sparrow	Half-normal	Minimum temerature, julian date	Observer	0
	Half-normal	Minimum temerature, julian date, wind (Beaufort)	Observer	0.55
	Half-normal	Minimum temperature, julian date, start time	Observer	1.51
Green-tailed towhee	Hazard-rate	None	Observer experience (OE)	0
	Hazard-rate	Julian date	OE	1.57
	Hazard-rate	Start time	OE	1.64
Sage thrasher	Hazard-rate	Julian date, wind (Beaufort)	Observer	0
	Hazard-rate	Julian date	Observer	0.86
	Hazard-rate	Julian date, start time	Observer	2.38

**Table 4.3.** Top count based models (< 2 AIC) relating bird densities (Brewer’s sparrow, sage thrasher, green-tailed towhee, and vesper sparrow) to experimental treatments at Stratton. \*dAIC = increase in AIC over top performing model and  $w_i$  = AIC weight

Species	Model	dAIC	wi
Brewer's sparrow	talldist + vasht + majbs6ha + tb + hb + Pasture(late)	0	0.427314
	talldist + vasht + majbs6ha + tb + hb + Pasture(late) + black0.25ha	1.7371	0.179284
	talldist + vasht + majbs6ha + tb + hb + Pasture(late) + cti400ha	1.9207	0.163558
	talldist + vasht + majbs6ha + tb + hb + Pasture(late) + as1ha	1.9664	0.159863
sage thrasher	decfeb + majbs6ha + cti400ha + maxht + bsdist	0	0.181935
	decfeb + majbs6ha + cti400ha + maxht	0.6899	0.128857
	decfeb + cti400ha + maxht + bsdist	0.6943	0.128574
	decfeb + majbs6ha + maxht + Pasture(early) + Pasture(mid-summer)	0.9529	0.112979
	decfeb + majbs6ha + cti400ha + maxht + purvascov + Pasture(early) + (mid-summer)	0.9729	0.111855
	decfeb + majbs6ha + cti400ha + purvascov + Pasture(early) + (mid-summer)	1.4769	0.086938
	decfeb + majbs6ha + cti400ha + bsdist + purvascov + as400ha	1.5313	0.084606
	decfeb + majbs6ha + cti400ha + bsdist + purvascov + hb6ha	1.5828	0.082455
green-tailed towhee	decfeb + majbs6ha + bsdist + purvascov + Pasture(early) + Pasture(mid-summer)	1.5987	0.081802
	purvascov + as6ha + tb + majbs6ha + majbs6ha_2	0	0.323329
	purvascov + as6ha + amj + majbs6ha + majbs6ha_2	1.4278	0.175528
	purvascov + as6ha + tb + majbs6ha + majbs6ha_2 + Pasture(early) + Pasture(mid-summer)	1.5265	0.167076
	purvascov + as6ha + tb + majbs6ha + majbs6ha_2 + cti0.25ha	1.7412	0.150069
purvascov + as6ha + tb	1.7567	0.148911	
vesper sparrow	totalcov + octfeb + Pasture(mid-summer) + bigsage6ha + sh400ha	0	0.513862
	totalcov + octfeb + Pasture(mid-summer) + bigsage6ha + sh400ha + hb1ha	1.9852	0.190443

**Table 4.4.** Random-effects negative binomial regression coefficient ( $\beta$ ) estimates for best model of Brewer’s sparrow counts following prescribed fire and grazing timing treatments at Stratton Sagebrush Ecological Research Site with estimated confidence limits (LCL – UCL) and incident rate ratios (IRR).

Parameter	LCL	$\beta$	UCL	IRR
talldist	-0.0107578	-0.0078292	-0.0049005	0.9922014
vasht	0.0051421	0.010687	0.0162318	1.010744
hb30m	0.0266025	0.0412803	0.0559581	1.042144
majbs6ha	0.0030006	0.0066868	0.0103729	1.006709
tb	0.1810513	0.3650618	0.5490723	1.079381
Pasture (late)	0.0242027	0.0763874	0.1285721	1.440603
-----				
ln(r)	3.499214	4.666823	5.834433	-
ln(s)	2.228096	3.26119	4.294284	-

Note: Parameters  $r$  and  $s$  represent random-effects dispersion which varies randomly by group following a Beta distribution.

**Table 4.5.** Poisson regression coefficient ( $\beta$ ) estimates for best model of sage thrasher counts following prescribed fire and grazing timing treatments at Stratton Sagebrush Ecological Research Site with estimated confidence limits (LCL – UCL) and incident rate ratios (IRR).

Parameter	LCL	$\beta$	UCL	IRR
bsdists	-0.0069316	-0.003121	0.0069316	0.9968838
maxht	0.0015169	0.0090897	0.0166625	1.009131
majbs6ha	-0.0007273	0.0038841	0.0084954	1.003892
cti400ha	0.0006555	0.0039874	0.0073193	1.003995
decfeb	0.005264	0.0164492	0.0276345	1.016585
-----				
alpha	0.0009846	0.0203968	0.4225202	-

Note: The parameter alpha represents random-effects dispersion following a Gamma distribution.

**Table 4.6.** Coefficient ( $\beta$ ) estimates for best model of green-tailed towhee counts following prescribed fire and grazing timing treatments at Stratton Sagebrush Ecological Research Site with estimated confidence limits (LCL – UCL) and incident rate ratios (IRR).

Parameter	LCL	$\beta$	UCL	IRR
purvascov	0.0205028	0.0489567	0.0774106	1.050175
as6ha	0.0063886	0.0730393	0.1396899	1.075773
tb	0.2520056	0.3726638	0.4933219	1.451596
majbs6ha	0.0058738	0.0349105	0.0639473	1.035527
majbs6ha^2	0.0005563	0.0002863	0.0000162	0.9997138
-----				
ln(r)	3.499214	4.666823	5.834433	-
ln(s)	2.228096	3.26119	4.294284	-

Note: Parameters  $r$  and  $s$  represent random-effects dispersion which varies randomly by group following a Beta distribution.

**Table 4.7.** Negative binomial coefficient ( $\beta$ ) estimates for best model of vesper sparrow counts following prescribed fire and grazing timing treatments at Stratton Sagebrush Ecological Research Site with estimated confidence limits (LCL – UCL) and incident rate ratios (IRR).

Parameter	LCL	$\beta$	UCL	IRR
octfeb	-0.0261728	-0.0198484	-0.0135239	0.9803473
Pasture (mid)	-0.2287298	-0.1244292	-0.0201286	0.8830008
sh400ha	-0.0943536	-0.0618432	-0.0293328	0.9400303
bigsage6ha	0.0064987	0.0085626	0.0106266	1.008599
totalcov	-0.0268708	-0.0183568	-0.0098428	0.9818107
-----				
ln( $r$ )	3.499214	4.666823	5.834433	-
ln( $s$ )	2.228096	3.26119	4.294284	-

Note: Parameters  $r$  and  $s$  represent random-effects dispersion which varies randomly by group following a Beta distribution.

## CHAPTER 5 : SYTHESIS

### 5.1 CONCLUSIONS

In the preceding thesis, I presented results of research assessing the effects of prescribed fire and grazing timing strategies on select ecosystem components within a high-elevation sagebrush system. Direct responses in herbaceous productivity were measured using dry weight biomass of graminoid and forb functional groups (see Chapter 2). I assessed relative seasonal habitat use (summer and winter) by greater sage-grouse (*Centrocercus urophasianus*) across treatments based the accumulation of fecal pellets (see Chapter 3). I also examined differential responses to treatments in migratory songbird densities and reproductive success (see Chapter 4). In each of the analyses relating impacts to avian habitat selection, responses were not only contrasted across categorical treatments but also using continuous variables representing habitat composition and burn configuration. The goal of my research was to provide insight into how current land management practices across high-elevation sagebrush rangelands impact wildlife species that depend on those habitats and offer recommendations on the appropriateness of certain management initiatives in the light of conservation concerns for sage-grouse and other sagebrush obligate species.

Livestock grazing has a long and contentious history within sagebrush rangelands (Clawson 1983), particularly on publicly administered lands. Prior to enactment of legislation intended to improve the health of public rangelands (Taylor Grazing Act 1934,



Federal Land Policy and Management Act 1976, Public Rangelands Improvement Act 1978), unchecked overgrazing across many semi-arid rangelands contributed to substantial degradation of those systems (McArthur and Plummer 1978). Following implementation of revised regulations, stocking rates were significantly reduced and many rangelands demonstrated obvious improvements through the latter half of the 20<sup>th</sup> century (Crawford et al. 2004). Despite changes in grazing practices, grazing remains a principle land-use across public sagebrush rangelands, occurring on > 53 million hectares.

Sagebrush control treatments have been closely associated with livestock grazing on both public and private rangeland. Prior to the early 1980's, large expanses of sagebrush were subjected to mechanical shrub removal or herbicide defoliation treatments in an attempt to increase herbaceous production by removing shrubs from the pool of resource competitors (Tanaka and Workman 1988). After the use of 2,4 D was banned on public lands, prescribed fire became the favored method of sagebrush control (Braun 1987). Prescribed fire has been documented to produce short-term increases in both forage yield (White and Currie 1983) and quality (Duvall and Whitaker 1964) within ecosystems historically dominated by herbaceous vegetation. One mechanism for this effect is the release of nitrogen, calcium, and other organic nutrients from previous years' residual biomass which makes these nutrients available for incorporation into new vegetative growth (Wan et al. 2001). However, increase in forage production after fire events is highly dependent on moisture conditions in the growing seasons that immediately follow a fire event. When fire is followed by drought, any short-term benefit is typically lost (Wright 1974). Increased forage production has been reported

post-fire within sagebrush rangelands (Harniss and Murray 1973, Davies et al. 2007) but anticipated enhancement has often been absent following woody vegetation control measures across much of the semi-arid sagebrush steppe (Daubenmire 1975, Fraas et al. 1992, Wambolt et al. 2001).

Herbaceous production was significantly lower in burn treatment plots compared to unburned big sagebrush control plots the first growing season following a fall prescribed fire at my study site. In the second growing season, herbaceous production did increase, but was only significantly greater than unburned habitats in one of three grazing pastures. By the third post-fire growing season differences in forage production between treatments were no longer discernable. Fluctuations in annual herbaceous vegetation production across semi-arid sagebrush systems often display highly variability and are principally determined by precipitation, soil characteristics, and pre-burn grass/forb composition (Ries and Fisser 1979). In my analysis of herbaceous response to fire treatments, I attempted to control for variation in herbaceous production related to precipitation. Therefore, a first year reduction in biomass production and a second year increase in only one of three pastures suggest that sagebrush removal via prescribed fire resulted in a short-term net loss in forage production at my study site. Grazing is typically deferred for one to two growing seasons after fire across most publically managed rangelands (BLM 2007). However, grazing at my study site was continued the growing season immediately following prescribed fire treatments. It is possible that results may have been different with a short-term grazing deferment. This does not negate the implications of my results, particularly since research by other investigators

has indicated that the timing of post-fire grazing has a much larger impact on herbaceous recovery and production than duration of rest (Bates et al. 2009, Augustine et al. 2010). The only positive response in herbaceous production observed at my study occurred during the second post-fire growing season. This single year increase was only observed in the pasture subjected to early summer grazing and increased production was negligible or non-existent in pastures subjected to mid- or late summer grazing prescriptions. It has been suggested that limiting grazing to the vegetative growth phase for perennial bunchgrasses gives these plants the opportunity to re-grow from existing tillers after grazing, while grazing during the reproductive stages can result in limited re-growth response (Bates et al. 2009). This may in turn affect seed pools and recruitment in subsequent growing seasons.

Targeted outcomes of prescribed fire initiatives across sagebrush habitats, in addition to increasing forage biomass for livestock (Laycock 1979), include halting the encroachment of conifers (Miller and Rose 1999), and enhancing habitat characteristics for greater sage-grouse and other native wildlife species (Martin 1990, Pyle and Crawford 1996, McDowell 2000). However, the utility of burning as a tool for enhancing sage-grouse habitat has recently been called into question (Robertson 1991, Nelle et al. 2000, Byrne 2002). Based on fecal pellet accumulation, used as a relative measure of habitat selection, sage-grouse at my study site avoided burn treatments during the summer, the season when beneficial increases in forb production are theorized to provide habitat enhancement. Further, removal of sagebrush cover had an even more pronounced negative impact on winter habitat selection. Sage-grouse selected

intermediate levels of localized shrub cover in summer. Cover estimates in burns were much lower than unburned habitats, but the burns were variable in average distance to unburned big sagebrush habitat, and patchy burns that retained substantial nearby big sagebrush had less negative impact on sage-grouse selection than large, contiguous burns with limited proximal shrub cover. Thus, prescribed fire in my study area did not appear to enhance sage-grouse habitat. If range managers do undertake a prescribed fire treatment, great care should be taken to implement a patchy burn configuration in areas where sage-grouse populations exist. Similarly, during winter, near complete avoidance of burned habitat was largely explained by the lack of sagebrush habitat at a very fine scale (1 meter pixels averaged over a 3 ha area). Selection for sagebrush habitat at such a small spatial scale suggests that prescribed fire should be precluded in areas where retaining winter habitat for sage-grouse is a conservation priority.

Research by others has suggested that sage-grouse avoid habitats with a large livestock presence (Lupis et al. 2006). However, the coarse temporal scale of my data precluded inference concerning the direct impacts of short-duration (2 weeks) livestock presence on sage-grouse. Rather, I attempted to examine season-long (winter and summer) differences in habitat use across grazing timing treatments in both burned and unburned big sagebrush habitats. While pasture effects were evident in an examination of categorical treatment variables, differences in vegetation characteristics between pastures, largely related to burn treatments, proved to have much greater explanatory power related to sage-grouse habitat selection. Thus, the timing of low to moderate

intensity summer grazing (early, mid-, or late summer) appeared to have minimal impact on either summer or winter habitat use by sage-grouse.

The impact of habitat alterations on greater sage-grouse has been the subject of an increasing number of investigations over the past 20 years (e.g. Martin 1990, Pyle and Crawford 1996, Connelly et al. 2000, Nelle et al. 2000, Aldridge and Boyce 2007). Due to the recent decision to list the species as a candidate for Endangered Species Act protection (U.S. Department of Interior 50 CFR Part 17 March 2010), greater sage-grouse response to public land management will likely continue to be closely scrutinized. However, many other wildlife species also depend on sagebrush ecosystems for part or all of their life history requirements. Comparatively little research has been conducted on the effects of prescribed fire and grazing regimes on migratory songbirds such as Brewer's sparrow (*Spizella breweri*), sage sparrow (*amphispiza belli*), and sage thrasher (*Oreoscoptes montanus*), which depend heavily on sagebrush habitats during the breeding season. Results of research that is available on songbird responses to sagebrush management initiatives vary widely and include positive, negative, neutral and mixed impacts on habitat selection and reproductive fitness depending on species or functional guild (Rotenberry and Wiens 1978, Castral 1982, Bock and Bock 1987, Peterson and Best 1999, Holmes 2007). Yet, range-wide population declines have been reported for many passerines considered to be sagebrush obligates (Brewer's sparrow: 2.1% per year between 1966 – 2007, sage thrasher 1.1% per year between 1980 – 2007, sage sparrow: - 6.1% per year between 1966 – 1980, Sauer et al. 2008).

Nest success rates for songbirds were unaffected by any of the treatment metrics I examined. Based on mechanisms determining animal distribution across habitat patches containing different amounts of resources described in the Ideal Free Distribution theory (Fretwell and Lucas 1969), this suggests that habitat selection assessed by bird densities can provide a relative measure of habitat suitability after treatments at my study site. I observed dramatically lower densities of shrub nesting birds in prescribed fire units compared to unburned big sagebrush. Densities of Brewer's sparrow were positively associated with patch scale (6 ha) proportions of habitat dominated by big sagebrush vegetation, mountain big sagebrush height, and local scale herbaceous cover, but densities for this species were negatively associated with distance to tall shrub habitat. Together these factors indicate that Brewer's sparrows are selecting habitats containing larger patches of highly productive (herbaceous understory) shrub habitat and strongly avoid of large burns that leave little intact big sagebrush in close proximity. Sage thrashers also selected for larger patch scale big sagebrush habitat (although confidence intervals overlapped zero indicating a weak effect), greater average shrub height and landscape scale (400 ha) compound topographic index, and avoided areas further from big sagebrush habitat. Similar responses displayed by these two sagebrush obligate species indicate a low tolerance for burning of big sagebrush for both of these species. These habitat preferences suggest that future prescribed fires should attempt to produce a patchy mosaic burn configuration and avoid burning the most mesic (productive) portions of available big sagebrush in cases where conservation of these species is a management concern. These management recommendations for future burn

prescriptions are similar to those stated in this thesis for sage-grouse, but again, any burning will have negative impacts on all of these species.

Timing of summer grazing appeared to have no effect on sage thrasher habitat selection. However, Brewer's sparrow densities were 44% greater in the late grazing pasture compared to both early and mid-summer grazing pastures. This result was contrary to my expectation that removal of residual grass and forb cover with grazing later in the growing season would result in lower nest site concealment and reduce habitat selection. It is possible that grazing intensity levels across unburned habitats were too low to significantly impact herbaceous vegetation components that influence nest site selection. A more likely explanation for greater selection within the late grazing pasture is un-modeled differences in vegetation composition or configuration. Interpreting interaction effects for continuous variables is often difficult. Therefore, I did not explicitly examine these interactions in my analyses. However, there were distinct differences in vegetation patterns between pastures and it is very likely that the interactions of these compositional and structural attributes of habitats impact habitat quality for Brewer's sparrow.

The results that I have presented herein suggest that the timing of low to moderate intensity summer grazing (earliest treatment began in late June) has little impact on post-fire herbaceous production, seasonal habitat use by sage-grouse (winter or summer), or breeding season habitat selection and fitness for migratory songbirds. Conversely, I have demonstrated that prescribed fire in a high-elevation sagebrush system may provide limited benefits related to enhancement of herbaceous production. Potential economic

benefits from a limited single season gain in livestock forage production were largely outweighed by the large loss of forage the year immediately after the fire. The negative ecological impacts that removing big sagebrush vegetation has on both sage-grouse and migratory songbirds further questions the appropriateness of prescribed fire treatments within high-elevation sagebrush systems.

## **5.2 MANAGEMENT IMPLICATIONS**

In the past, management goals across sagebrush rangelands largely considered economic benefits related to enhancing livestock forage and maintaining a balance of vegetation components. More recently, it has been recognized that wildlife species also have value (although difficult to quantify) and that impacts on wildlife species should be considered when making management decisions. While long-term overgrazing in sagebrush systems previously led to ecosystem degradation, short-duration grazing at light to moderate intensity levels during the summer appears to have negligible effect on avian species. However, limited increases in forage biomass observed in this, and other studies, suggests that negative ecological impacts on avian species resulting from prescribed fire in sagebrush ecosystems overwhelmingly outweigh any minor economic benefits related to livestock forage. If prescribed fire is retained as a management tool within publicly administered sagebrush systems, then burn pattern configurations that limit negative impacts on big sagebrush dependant wildlife species should be implemented.



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