Bird and Small Mammal Communities of Sagebrush-Dominated Mountain Meadows: An Examination of Meadow Characteristics as Part of a Hierarchical, Multi-Level Study of the Wasatch-Cache National Forest

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BIRD AND SMALL MAMMAL COMMUNITIES OF SAGEBRUSH-DOMINATED MOUNTAIN MEADOWS: AN EXAMINATION OF MEADOW CHARACTERISTICS AS PART OF A HIERARCHICAL, MULTI-LEVEL STUDY OF THE WASATCH-CACHE NATIONAL FOREST

by

Elizabeth Johnson

A thesis submitted in partial fulfillment of the requirements for the degree of

MASTER OF SCIENCE

in

Wildlife Biology

Approved:

UTAH STATE UNIVERSITY
Logan, Utah

2005
ABSTRACT

Bird and Small Mammal Communities of Sagebrush-Dominated Mountain Meadows: An Examination of Meadow Characteristics as Part of a Hierarchical, Multi-Level Study of the Wasatch-Cache National Forest

by

Elizabeth J. Johnson, Master of Science
Utah State University, 2005

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Department: Forest, Range, and Wildlife Sciences

Sagebrush shrubsteppe ecosystems have increasingly garnered attention as an endangered ecosystem. Ninety nine percent of all sagebrush ecosystems are thought to have been impacted by humans, and over 50% of grassland and shrubsteppe species are believed to be in decline. Most of the research on sagebrush ecosystems has been conducted at lower elevations and in large expanses of sagebrush. A considerable amount of sagebrush is found at higher elevations, often in meadows found within a forest matrix. The role of this high-elevation habitat is poorly understood. We conducted bird, small mammal, vegetation, and soil surveys in sagebrush-dominated mountain meadows within the Wasatch-Cache National Forest in northeastern Utah. Meadows ranged from 0.6 to 782 hectares in size and included an impressive list of associated plant species. We detected two sagebrush-obligate species and numerous shrubsteppe-associated species.
Each species appears to respond to different habitat characteristics, but all species that showed a significant relationship with meadow size were more likely to occur in larger meadows. Many species showed no relationship with size, suggesting that while larger meadows were preferred by some species, small meadows could also play an important role as habitat. While sagebrush-dominated mountain meadows were important for some species, we also failed to detect a number of species of interest. In particular, Sage Thrasher, Sage Sparrow, and pygmy rabbit were not found within the study area. North American Breeding Bird data suggests that Sage Thrashers can be found nearby. It is likely that these birds are only found in large expanses, and none of our meadows were large enough to support them. Sagebrush-dominated mountain meadows appear to be an important supplement to large expanses of sagebrush shrubsteppe habitat, but are not substitutable for all species.

(83 pages)
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Elizabeth J. Johnson
# CONTENTS

<table>
<thead>
<tr>
<th>ABSTRACT</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>ii</td>
</tr>
<tr>
<td>ACKNOWLEDGEMENTS</td>
<td>iv</td>
</tr>
<tr>
<td>LIST OF TABLES</td>
<td>vii</td>
</tr>
<tr>
<td>LIST OF FIGURES</td>
<td>viii</td>
</tr>
</tbody>
</table>

## CHAPTER

1. **LITERATURE REVIEW**

   - Introduction                                                              | 1   |
   - Sagebrush Ecosystems                                                     | 3   |
   - High-elevation Sagebrushes                                               | 3   |
   - Land ownership                                                           | 5   |
   - Geographical location and extent                                          | 5   |
   - Mountain Meadows                                                         | 6   |
   - Creation and maintenance of forest openings                              | 7   |
   - Climate                                                                  | 10  |
   - Fire                                                                     | 11  |
   - Land-use history                                                         | 12  |
   - Primary producers                                                        | 14  |
   - Consumers                                                                | 14  |
   - Purpose                                                                  | 15  |
   - Style                                                                    | 16  |
   - Literature Cited                                                         | 17  |

2. **HIGH-ELEVATION SAGEBRUSH MEADOWS: SUBSTITUTE FOR OR SUPPLEMENT TO LOW-ELEVATION SAGEBRUSH HABITAT**

   - Abstract                                                                  | 25  |
   - Introduction                                                              | 26  |
   - Methods                                                                   | 28  |
   - Study area and site selection                                             | 28  |
Bird sampling ................................................................. 31
Small mammal sampling .................................................. 33
Meadow and habitat characteristics sampling ..................... 34
Data treatment ................................................................. 35
Statistical analyses ............................................................ 36

Results .................................................................................. 37
Discussion .............................................................................. 40
Literature Cited ....................................................................... 42

3. A COMPARISON OF DOUBLE-OBSERVER AND DISTANCE
   METHODS FOR MODELING BIRD DETECTABILITY ............... 49
   Abstract ........................................................................ 49
   Introduction .................................................................... 49
   Methods ......................................................................... 52
   Results ........................................................................... 54
   Discussion ....................................................................... 55
   Literature Cited ............................................................... 61

4. CONCLUSIONS ................................................................. 68
   The Contribution of High-Elevation Sagebrush Shrubsteppe .... 68
   Survey Effort and Density Calculations .............................. 69
   Recommendations for Research and Management .............. 71
   Literature Cited ............................................................... 72

APPENDIX .............................................................................. 75
LIST OF TABLES

<table>
<thead>
<tr>
<th>Table</th>
<th>Description</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>2.1</td>
<td>Bird species detected within study area</td>
<td>47</td>
</tr>
<tr>
<td>2.2</td>
<td>Mammal species trapped within study area</td>
<td>48</td>
</tr>
<tr>
<td>3.1</td>
<td>Probabilities of detection for bird species using Program Distance ($p_1$) and Program Dobserv ($p_2$) with standard errors for each</td>
<td>63</td>
</tr>
<tr>
<td>4.1</td>
<td>List of birds identified within the study area during point transects</td>
<td>74</td>
</tr>
</tbody>
</table>
**LIST OF FIGURES**

<table>
<thead>
<tr>
<th>Figure</th>
<th>Description</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>3.1</td>
<td>Green-tailed Towhee detection probabilities. Bars are based on actual</td>
<td>64</td>
</tr>
<tr>
<td></td>
<td>detections, line indicates predicted values</td>
<td></td>
</tr>
<tr>
<td>3.2</td>
<td>Green-tailed Towhee detection probabilities when data is not</td>
<td>65</td>
</tr>
<tr>
<td></td>
<td>converted into intervals. Note the increased probabilities of detection</td>
<td></td>
</tr>
<tr>
<td></td>
<td>around 15, 25, 35, 40, 50, 60 and 70-meter detection distances</td>
<td></td>
</tr>
<tr>
<td>3.3</td>
<td>Chipping Sparrow detection probabilities modeled with a uniform key</td>
<td>66</td>
</tr>
<tr>
<td></td>
<td>function and cosine series expansion</td>
<td></td>
</tr>
<tr>
<td>3.4</td>
<td>Chipping Sparrow detection probabilities modeled using a hazard rate key</td>
<td>67</td>
</tr>
<tr>
<td></td>
<td>function</td>
<td></td>
</tr>
</tbody>
</table>
CHAPTER 1
LITERATURE REVIEW

Introduction

Throughout the Intermountain West, sagebrush is one of the most dominant plants. Sagebrush ecosystems are believed historically to have dominated approximately 63 million hectares (156 million acres) in the Western United States (West & Young 2000) and by some estimates may have covered as much as 270 million acres (Blaisdell et al. 1982). Big sagebrush ecosystems occur from ~1,600 feet to 11,500 feet in elevation (Blaisdell et al. 1982), and in a variety of climatic conditions ranging from warm deserts to alpine tundra.

Although widespread, sagebrush ecosystems have been seriously impacted over much of their range (Baker et al. 1976; Davis 1982; Paige & Ritter 1999; Vander Haegen et al. 2000; Bauer et al. 2002; Knick et al. 2003) by human use. A desire for better livestock forage has led to efforts to reduce sagebrush abundance and increase graminoid and herb abundance, including disking, chaining, plowing, herbicide use, fall sheep grazing, prescribed burning, and reseeding with introduced grass species (Davis 1982; Vander Haegen et al. 2000; Bauer et al. 2002). Sagebrush also has been removed so that land can be converted to production agricultural use (Paige & Ritter 1999). Invasion by introduced plant species has changed sagebrush community ecosystems. *Bromus tectorum* (cheat grass), in particular, has changed the ecology of huge tracts of land, but *Taeniatherum caput-medusae* (medusahead), *Centaurea solstitialis* (yellow star thistle), *Centaurea ssp.* (knapweed), *Sisymbrium altissimum*
(tumble mustard), and *Halogeton glomeratus* (halogeton) also have had a continued impact (Paige & Ritter 1999). It is currently estimated that approximately 10% of sagebrush steppe has been converted to agriculture, 25% has become dominated by introduced annual grasses such as *Bromus tectorum*, and over 99% has been impacted by livestock grazing (West 1996). As a result, sagebrush-dominated communities are now considered one of the most endangered ecosystems in North America (Knick 1999; Anderson & Inouye 2001; Knick et al. 2003). About 70 mammal species and 100 bird species are found in sagebrush habitats. Over 50% of western grassland and shrubland bird species are in decline, and some of these species live only in sagebrush (Paige & Ritter 1999).

While the majority of studies of sagebrush habitats have focused on low elevations, high-elevation sagebrush habitats, particularly sagebrush-dominated mountain meadows, also have been changed by both historic and current anthropogenic activities. However, because of greater precipitation, relative inaccessibility, prolonged winter conditions, and spatial patchiness, high elevation areas may have been less affected by human activities than the sagebrush steppe found at lower elevations and consequently may currently function as a refugium for sagebrush obligate or near-obligate species. Indeed, high elevation areas appear to have greater biological diversity than low-elevation sagebrush (Medin et al. 2000) and as a consequence, a higher conservation value. Careful management and conservation of currently intact high-elevation sagebrush steppe, or sagebrush-dominated mountain meadows or parks, may be a cost-effective supplement to the restoration of lower sagebrush steppe.
Sagebrush Ecosystems

Sagebrush ecosystems are typically classed as Great Basin sagebrush and sagebrush steppe (West & Young 2000). In sagebrush steppe, sagebrush is co-dominant with perennial bunchgrasses and there is an herb-dominated phase (West & Young 2000). Great Basin sagebrush areas are more arid and sagebrush is the sole dominant, typically comprising 70% or more of the relative plant cover (West & Young 2000). Almost all is publicly owned (West 1983a). Sagebrush steppe occurs in the northern portion of the Intermountain region and at higher elevations in southern areas whereas Great Basin sagebrush occurs to the south (West & Young 2000).

About one third of sagebrush steppe is privately owned; two thirds are on public lands (West 1983b). Sagebrush steppe and Great Basin sagebrush both cover huge contiguous areas, but there are also smaller pockets of sagebrush found within other matrices. For instance, at high elevations forests are dominant, but sagebrush-dominated meadows are found within these forests.

High-elevation Sagebrush Systems

Small pockets of high-elevation sagebrush systems have been somewhat difficult for researchers to deal with. They are sagebrush-dominated systems, and because of that are often lumped with low-elevation sagebrush steppe. They exist, however, under extremely different conditions. Alternatively, sagebrush meadows are sometimes lumped with other types of mountain meadows. As a consequence, those who have studied either meadows, or alternatively sagebrush systems (e.g., Peet 2000), have in effect left high elevation sagebrush ecosystems unstudied. High-
elevation sagebrush-dominated mountain meadows typically fall into the sagebrush steppe category but are often more diverse and productive (Jensen et al. 1988). A variety of herbs, graminoids, and other shrubs are typically abundant with sagebrush being only co-dominant. Associated shrubs can include Cercocarpus montanus (mountain mahogany), Amelanchier alnifolia (serviceberry), Prunus spp., Purshia tridentata (bitterbrush), Ceanothus, and Symphoricarpos spp. (snowberry). These meadows may be surrounded by a variety of forest types including juniper-pinyon woodlands, ponderosa pine forests, lodgepole pine forests, spruce-fir forests, and aspen forests (Knight 1994). They typically comprise but a small part of a diverse landscape mosaic.

A variety of wildlife is associated with mountain meadows. Medin et al. (2000) conducted a survey in each main habitat type along an altitudinal gradient in Nevada and found more birds in mountain big sagebrush than any other habitat type. Also, a variety of game animals used these areas for forage (Eckert et al. 1973). However, much of the available literature on animal species composition in sagebrush steppe is based on research done in low-elevation sagebrush, and the contribution of high-elevation sagebrush to biodiversity is unclear.

The big sagebrush subspecies most commonly found at high elevations are Artemisia tridentata ssp. vasyana (mountain big sagebrush, Vasey sagebrush, or subalpine big sagebrush) and A. t. spiciformis (spiked big sagebrush or subalpine big sagebrush). A. t. spiciformis typically has been considered a form of vasyana and only recently recognized as a separate species (Winward 2004) or subspecies (Johnson 2000), though much of the data available for A. t. vasyana does not
differentiate it from *spiciformis*. The majority of data available on all big
gazebrushes is either generalized to *A. tridentata* without regard to subspecies or
generalized to mountain meadows.

**Land ownership**

BLM manages the majority of sagebrush habitat (Bureau of Land
Management 2002), but much of the high-elevation sagebrush is under Forest Service
management. The Forest Service manages the majority of the forested areas in the
Intermountain West, so sagebrush meadows within the forest matrix are likely to be
under Forest Service management as well. Of the sagebrush that the Forest Service
manages, the majority is mountain big sagebrush (Winward 1999).

**Geographical location and extent**

*A. t. vaseyana* occurs from British Columbia south through Utah to California,
New Mexico and Arizona. To the east, it is found in Colorado, Montana, and
Wyoming, and has been identified as far east as Nebraska and the Dakotas. It occurs
in a wide variety of geographical provinces: the Basin and Range, Rocky Mountains,
elevations of ~1,400 to 3,000 meters (~4,600 to 10,000 feet) and in a variety of
ecological conditions. Much of the literature on *A. t. vaseyana*, therefore, may not be
relevant to all *A. t. vaseyana* communities, especially those in mountain meadows. *A.
t. spiciformis* occurs only at the highest elevations, typically 2,700 to 3,000 meters
(~8,800 to 10,000 feet) or higher (Bureau of Land Management 2002). Although
more recent estimates are unavailable, in 1960 Beetle estimated that there were 7,000
mi$^2$ of $A. \ t. \ vaseyana$ and only 10 mi$^2$ of $A. \ t. \ spiciformis$ (then $A. \ t. \ vaseyana$ form $spiciformis$). Montana, Washington, California, and Nevada are estimated to only have 1 mi$^2$ each of this subspecies. For this study, only sagebrush-dominated mountain meadows in Utah were considered. We identified big sagebrush subspecies on site with $A. \ t. \ vaseyana$ the most common, followed by $A. \ t. \ spiciformis$ then $A. \ t. \ wyomingensis$.

**Mountain Meadows**

Within forested areas there occur treeless openings or meadows varying from several meters to several miles in diameter. For each meadow, the transition between forest and meadow is often abrupt (Daubenmire 1943a). Plant cover typically is similar to the plant cover of the adjacent basal plains. In the central Rockies, parks and meadows on the eastern slope are often dominated by grasses whereas on the west slope sagebrush is a dominant (Daubenmire 1943a). However, different microclimates result in a variety of vegetation types, some quite different from those of lower elevations.

Meadows are extremely variable in terms of community composition, elevation, moisture, substrate, soil, elevation, topography, and nutrient availability (Hunt et al. 1988; Knight 1994; Povirk et al. 2001). Meadows have been classed using a variety of schemes, and there does not appear to be any one commonly used system. In the Greater Yellowstone Ecosystem, Debinski et al. (1999) recognized six different meadow types. *Artemisia tridentata* was identified in five and comprised $>5\%$ of the total cover and considered by Debinski to be a dominant in three. Within
Grand Teton National Park, Sabinske and Knight (1978) identified four distinct sagebrush communities. Knight (1994) recognized four subalpine meadow types in Wyoming, only one of which was sagebrush meadow. In our study, all meadows with a sagebrush component of at least 5% were included.

**Creation and maintenance of forest openings**

There has been considerable speculation as to what causes forest openings. A variety of possible reasons have been suggested, including water-saturated soils which cause anoxia, vertisols (shrinking clay soils), cold air accumulating in sinks, slope, wind exposure, avalanches, fire, logging, ungulate grazing, pocket gopher activity, and other disturbances (Daubenmire 1943b; Burke et al. 1989a; Knight 1994). Additionally, parks and meadows often are associated with poorly aerated soils, dry soils, steep slopes, high wind and sun exposure, or heavy snow accumulation (Daubenmire 1981).

Lynch (1998) evaluated three hypotheses for the origin of parks dominated by sagebrush, grasses, and forbs. The “permanent site hypothesis” suggests that parks are the result of different physical characteristics of the site, and therefore the opening is stable. The “remnant hypothesis” suggests that parks and meadows are remnants of once widespread vegetation. Climate change has allowed trees to slowly move in and spread. Finally, the “replacement hypothesis” suggests that parks replaced forest in response to disturbance and climate change. Lynch looked at fossil pollen and found that the dominant plants had indeed changed over time, supporting the “replacement hypothesis.” Once forest is removed, meadow vegetation can prevail if climatic
conditions do not favor tree species recruitment. However, other explanations are possible. Indeed, some meadows have stable boundaries, while others show evidence of invasion by trees. Although often associated with anthropogenic activities, such as fire regime change, climate change, and grazing (Dunwiddie 1977; Knight 1994), natural processes are involved. Also, soil conditions and slope interact with climate to determine water availability, temperature averages and extremes, and snow redistribution (Billings 1969). According to Povirk et al. (2001) "edaphic properties may be the primary control on mountain meadow existence and perseverance." In particular, soil texture can be a critical factor (Daubenmire 1943a; Peet 2000); parks and meadows often are found in valley bottoms with fine textured (alluvial and colluvial) soils. Excessive soil moisture or dryness can prevent tree establishment or favor shrubs, herbs, or grasses. Highly fertile soils often support shrubs, which may inhibit tree regeneration (Knight 1994; Peet 2000). Even for this, however, there is no agreement. Schimpf et al. (1980) found the soils of four zones within spruce-fir forest, one of which was meadow, and one of which was spruce-fir, to be physically and chemically similar and therefore conclude that soil was not a cause of meadow existence in their study area. They agreed with Lynch (1998) that meadows may have become established during a colder period and that the vegetation itself prevents tree establishment.

Regardless of the origin of meadows and parks, current environmental factors contribute to their perpetuation. The vegetation patterns themselves influence snow distribution patterns. Creation of a park or meadow through fire or other disturbance changes snowdrift patterns, energy balances, and local hydrologic budgets such that
these openings may become self-perpetuating (Billings 1969). Support for this idea come from data that show that alternating strips of ribbon forest and meadow strips are correlated well with winter snow accumulations. Drifting snow that lasts until late summer can prevent tree establishment or germination, shorten the growing season, encourage parasitic molds, break or bend trees, and increase pocket gopher densities (Knight 1994).

Snow depth, snowpack release date, summer moisture availability, and the surrounding forest type all play a role in the determination of the vegetation within a mountain meadow (Weaver 1974; Burke et al. 1989a; Povirk et al. 2001). In areas of extremely deep snow accumulation (>2.8 meters), sagebrush is either unable to germinate or is killed by snow molds (Allen et al. 1987; Sturges 1989). _A. t. vaseyana_, therefore, is most successful in areas with intermediate amounts of snowfall; _A. t. wyomingensis_ or other vegetation replaces it when average snowpack is too thin (Sturges 1989). Cold air or frost pockets can also determine which species are able to germinate.

Not all subalpine meadows are stable or have fixed species composition lists, however. Tree invasion is causing the loss of many mountain big sagebrush meadows systems (Miller & Eddleman 2000). Trees invade meadows during prolonged droughts and snow-free periods (Peet 2000). Meadow losses also are correlated with livestock grazing (Dunwiddie 1977; Peet 2000) and fire suppression (Miller & Eddleman 2000). Conversely, sagebrush invasion in herbaceous meadows has been an issue of concern (Berlow et al. 2002).
Climate

The macroclimate of mountain meadows is the same as that of the surrounding forest. Different soils, topography, aspect, and the effects of the vegetation itself, however, result in remarkably different localized conditions. Wind redistribution of snow in particular, dramatically changes the effective precipitation any one area receives (Burke et al. 1989b). Mountain forests typically are mesic; moisture is not a limiting factor except where soils and/or topography limit moisture available to plants. The average annual temperature decreases by about 6°C for each 1,000 meters of elevation gain, but time of year, cold-air drainage related to topography, mountain size, snow cover duration, and temperature inversion potential can make these averages useless (Knight 1994). In subalpine forest zones the winters are cool and long (6-8 months) and summers are cool and short. Temperature fluctuations can be severe. Altitude alone has a number of effects due to the relatively thinner atmosphere of high elevations. There is more intense insolation during the day, rapid loss of heat from soil surfaces at night, greater evaporation, and a tendency toward suboptimal oxygen concentrations in the soil (Daubenmire 1943b).

Western mountain meadows occur within a variety of forest types, each of which experience different climatic conditions. Lodgepole pine and ponderosa pine forests are found in notably drier climate regimes than aspen or spruce-fir forests (Knight 1994). This may explain the variation of precipitation described in the literature, from a low of about 12 to a high of over 30 inches per year. The timing of precipitation varies depending on region. At the Rocky Mountain Biological Laboratory in Colorado, heavy snowfall through the winter melts in mid-May or early
June and the majority of the rain comes as August afternoon showers (Shaw & Harte 2001). Several study sites have warm summers with frequent thunderstorms and warm, dry spring and fall weather. At these sites, about one quarter of the annual precipitation occurs during the summer growing season (USDA 2003). There is no available information on the average growing season, but one site had 50-56 days each year when neither temperature nor soil moisture limited mountain big sagebrush growth (Jensen et al. 1989). Considering the wide variability among sagebrush ecosystems, there is likely considerable variability in growing season as well.

Unpredictable temperature extremes are common in mountain meadows. In early February at Middle Sink, a high elevation hollow near Logan, UT, the temperature was recorded at -61°F (-51°C) (West pers. commun.; Williams 2004). These extremes are more important than the averages in determining species compositions (West pers. commun.).

Fire

Greater precipitation results in higher productivity and a greater fuel load (Cook et al. 1994). As a result, fire return intervals were historically shorter in these systems than for low-elevation sagebrush systems. For *A. t. vaseyana* the return interval was 12-15 years in some populations (Miller & Rose 1999), whereas the fire return interval for *A. t. wyomingensis* was 60-110 years (Miller & Rose 1999). As a result, these sagebrushes are more adapted to fire. *A tridentata spiciformis* is the only big sagebrush to resprout after being top-killed by fire (Winward 1999). However, the vegetative and nutrient responses of high-elevation big sagebrush to fire are largely
unknown (Cook et al. 1994), and some *A. t. vaseyana* communities may have a fire return interval of as long as 200 years (Miller 2002).

**Land-use history**

Historically, sagebrush ecosystems are thought to have existed in a variety of conditions. This varied from a low density of sagebrush with high densities of grasses and/or forbs to nearly pure stands of sagebrush. Additionally, there was a mosaic of different states (Bureau of Land Management 2002). Vale (1975) reviewed 29 historical journals and diaries that emphasized the dominance of *Artemisia* throughout the Intermountain West, but unfortunately none of the entries he provided appear to refer to high-elevation sagebrush-dominated meadows. Overall, there appears to have been little written about these ecosystems historically.

A primary human use of sagebrush ecosystems has been grazing. Livestock grazing in many mountain meadows probably began about 1890 with heavy grazing from about 1900 through the 1930s (Povirk et al. 2001), but some grazing may have begun even earlier; high-elevation watersheds in the West were notably damaged as early as 1880 (Ellison 1960 as per Monsen & Shaw 2000). Over the last 100-plus years, considerable summer grazing in all meadow types has continued because ranchers typically wanted to rest their own lands in spring when plants were growing and more susceptible to damage, and as a consequence moved their livestock onto public lands (Knight 1994). Early heavy grazing can damage the vegetation of the mountain meadows, especially if soils are still wet from snow melt. Currently, permittees are allowed to graze livestock from mid-summer to early fall (Knight
1994). Alternatively, some meadows are under private ownership and historically have been used as private summer ranches. Second homes or ranchettes, which may have an even greater and more permanent ecological impact, are now being built on some of this private land (West pers. commun.).

Mountains also provide watersheds for cities and human developments. The Forest Reserve Act of 1891 was passed in large part due to concerns about flooding, streamflow, and water supplies. This act, along with the Organic Act of 1897, resulted in many high-elevation areas in the west being set aside as forest reserves. As a result, many areas have been protected from destructive use and managed for erosion control, although multiple use management principles allow managed livestock grazing and timber harvests to continue (Harper 1953). Additionally, erosion control has included the use of introduced species (Monsen & Shaw 2000) but the affects of these species on high-elevation sagebrush ecosystem functions have not been explored fully.

Current human uses include recreation. Sagebrush meadows are popular with snowmobilers and backcountry skiers in the winter and ATV drivers and campers in the summer. Even light recreational use, e.g., foot trails, have been shown to impact animal species diversity and composition (Miller et al. 1998), and increasing use due to population growth, improved technology, and disposable time and income will likely have a greater impact on these areas in the future.
Primary producers

Peet (2000) wrote that “relatively little has been written on the composition of the meadow and park vegetation of the Rockies.” However, communities dominated by *A. t. vaseyana* have significantly greater levels of primary production and species richness and diversity than other sagebrush systems (Jensen et al. 1988), presumably caused by greater soil moisture. In comparison to other montane ecosystems however, sagebrush meadows have a low to intermediate level of primary production. While there is high productivity and plant density in the summer growing season, in winter, all meadow vegetation except sagebrush senesces in at least some areas (Shaw & Harte 2001). *A. t. vaseyana* and *A. t. spiciformis* have two types of leaves: ephemeral leaves that are present only during the growing season, and persistent leaves that last 12-13 months before being shed (Favi & Eversman 2002). Little has been written about the effects and distribution of invasive plant species in sagebrush meadows, e.g. orchardgrass, smooth brome, tarweed. While introduced species are found at high elevations, overall, it appears that they have not been a major problem in these areas.

Consumers

Information on animal species that use sagebrush meadows is surprisingly scarce. In order to get an idea of the species likely to be present, a variety of other known habitat types can be combined. For example, there are many species listed as present in mountain meadows. Some overlap with species listed as found in sagebrush. Many animals typically associated with forests are likely to be found in mountain meadows. With the goal of evaluating use of sagebrush-dominated
mountain meadows in comparison to large expanses of sagebrush shrubsteppe, we compiled a list of sagebrush and shrubsteppe associates from Dobkin and Sauder (2004), Paige and Ritter (1999), and Braun et al. (1976). We removed species not known to occur in Utah, Idaho, or Wyoming. Species that occurred in Idaho or Wyoming but not Utah were included due to the proximity of both states to our study area. Species lists are shown in Tables 2.1 and 2.2 in Chapter 2.

Purpose

This study is part of a hierarchical study examining both landscape and meadow-level constraints on sagebrush obligate and shrubsteppe-associated species. There are several purposes of this study. Our overarching goal was to determine the contribution of high-elevation sagebrush shrubsteppe meadows to sagebrush biodiversity by determining which sagebrush obligate and shrubsteppe associate birds and small mammals used sagebrush-dominated mountain meadows in Utah. We compared our species lists with those from lower elevations. Sagebrush obligates require sagebrush for some portion of their lifecycles. Shrubsteppe associates are species that are heavily or entirely dependent on shrubsteppe ecosystems, which may or may not include a sagebrush component. A secondary objective was to assess the influence of meadow characteristics such as vegetation, soils, size, shape, and edge length on species presence for each of the identified sagebrush obligate and shrubsteppe-associated species. A companion study is examining species responses to higher level meadow organization and assessing the relationship between the species of interest and landscape level variables such as landscape configuration.
To achieve these objectives, 50 points were randomly selected from the Wasatch- Cache National Forest of northeastern Utah; the nearest meadow to each point was selected as the meadow of interest. From these, 34 meadows met our selection criterion. These meadows were surveyed for birds, small mammals, vegetation, and soils. For bird surveys we used point transects, which are similar to point counts, but include a distance measurement from the point to each bird or cluster of birds detected. We also used the double-observer method. Both methods adjust bird numbers to reflect detectability. We present and discuss our bird counting methodology and the value of each of these methods in greater detail in Chapter 3. To measure small mammal diversity, we set up two trap webs within each of 11 of the 34 meadows. Vegetation surveys were done at all points, but soil sampling was conducted only at small mammal trapping sites. Species detected, and habitat variables which correlated with each species, as well as a more detailed description of our methods are presented in Chapter 2.

Style

This thesis is written in the multiple-paper format. Chapters 2 and 3 are each intended for publication in scientific journals, but Chapters 1 and 4 are not. Chapters 1 and 4 were written to introduce and summarize the other papers. The Abstract, Acknowledgments, Contents, Chapter 2, and the Conclusion follow the editorial guidelines of Conservation Biology. Chapter 3 was written using the editorial guidelines of The Auk.
Literature Cited


CHAPTER 2

HIGH-ELEVATION SAGEBRUSH MEADOWS: SUBSTITUTE FOR OR SUPPLEMENT TO LOW-ELEVATION SAGEBRUSH HABITAT? ¹

Abstract

As sagebrush shrubsteppe ecosystems throughout the western U.S. decline, so do associated bird and small mammal species. As a result, understanding the causes and consequences of habitat and species loss and decline increasingly has become a priority. However, much of the research in these systems has been conducted in the large expanses of open sagebrush at lower elevations. The role of sagebrush-dominated high elevation mountain meadows as reservoirs or sources for sagebrush shrubsteppe obligate and associated species is poorly understood. We asked the questions: To what extent do high elevation meadows contribute to the total biodiversity of sagebrush shrubsteppe birds and small mammals? Can high elevation meadows substitute for lower elevation habitat loss? We conducted our study on the Wasatch-Cache National Forest in northeastern Utah, where the majority of sagebrush steppe is located within a forested matrix. The sagebrush-dominated meadows examined in our study ranged from 0.6 to 781 hectares, and were highly variable in vegetation except that all contained a sagebrush component. We surveyed for birds and small mammals for two summers. Some of the known sagebrush-obligate species were detected; others were not. We did record a high number of shrubsteppe-associated species, characterized by a higher percentage of riparian species than of upland species. This may be due in part to the more mesic conditions.

¹ Coauthored by Johnson, Elizabeth J., Tammy L. Wilson, and John A. Bissonette. Department of Forest, Range, and Wildlife Sciences, Utah State University, Logan, UT.
of this type of sagebrush shrubsteppe. However, it is clear that the high-elevation
mountain meadows do not substitute for lower elevation habitat. We found many
fewer sagebrush-obligate species in the meadows at high elevations. Nevertheless,
these habitats appear to play an important role for many species associated with
shrubsteppe.

Introduction

Throughout the Intermountain West, sagebrush (woody *Artemisia* spp.) is one
of the most dominant plant groups, with sagebrush shrubsteppe ecosystems
historically dominating between 63 and 110 million hectares in the western United
States (Blaisdell et al. 1982; West & Young 2000). Sagebrush ecosystems occur from
about 1,600-11,500 feet elevation (Blaisdell et al. 1982), and in a variety of climatic
conditions ranging from warm deserts to alpine tundra. Although widespread, these
ecosystems have been seriously impacted over much of their range and are garnering
increasing interest and concern (Baker et al. 1976; Davis 1982; Paige & Ritter 1999;

The decline in sagebrush shrubsteppe stems from a variety of contributing
factors. Sagebrush management often has been conducted with a desire to improve
agriculture and create better livestock forage (Baker et al. 1976). Efforts to reduce
sagebrush abundances and increase graminoid and herb abundances have included
disking, chaining, plowing, herbicide use, fall sheep grazing, prescribed burning, and
reseeding with introduced grass species (Davis 1982; Vander Haegen et al. 2000;
Bauer et al. 2002). Invasion by a variety of introduced plant species has also affected
sagebrush habitats. *Bromus tectorum* L. (cheat grass), in particular, has changed the ecology of huge tracts of land, but *Taeniatherum caput-medusae* (L.) Nevski (medusahead), *Centaurea solstitialis* L. (yellow star thistle), *Centaurea* L. spp. (knapweeds), *Sisymbrium altissimum* L. (tumble mustard), and *Halogeton glomeratus* (Bieb.) C.A. Mey. (halogeton) also have a continued impact (Paige & Ritter 1999). West (1996) estimated that approximately 10% of sagebrush steppe has been converted to agriculture, 25% has become dominated by introduced annual grasses such as cheat grass, and over 99% has been impacted by livestock grazing. As a result, sagebrush-dominated communities are now considered one of the most endangered ecosystems in North America (Knick 1999; Anderson & Inouye 2001; Knick et al. 2003).

Approximately 100 bird species and 70 mammal species use sagebrush habitats. Over half of all western grassland and shrubland bird species are in decline, including species that live only in sagebrush (Paige & Ritter 1999). For small mammals, fewer data are available, but it appears that many shrubsteppe species have been reduced to small, isolated populations (Dobkin & Sauder 2004). In our study we looked at two groups of sagebrush-associated birds and small mammals: sagebrush obligates and sagebrush associated species. Obligate species require sagebrush habitats for some portion of their lifecycle, while associates are heavily dependent on these ecosystems.

While the majority of the studies of sagebrush habitats have focused on low elevations and on large expanses of sagebrush, high-elevation sagebrush habitats, particularly sagebrush-dominated mountain meadows, also appear to harbor many of
these species (Medin et al. 2000). Although these areas have been impacted by both historic and current anthropogenic activities, greater precipitation, relative inaccessibility, prolonged winter conditions, and patchiness, high-elevation areas appear to be less influenced by human impacts than lower elevation sagebrush steppe. Consequently, high-elevation habitats may function as a refugium or source for sagebrush obligate and shrubsteppe-associated species and in some sense provide a substitute for lower elevation habitats that are declining.

High-elevation and low-elevation sagebrush communities differ qualitatively from one another in slope and aspect, moisture, soil temperature, vegetation structure, landscape area, patch configuration, dominant sagebrush species and subspecies of big sagebrush, as well as adjacent land types and uses. North American Breeding Bird Survey data for routes near our study area suggested that a variety of sagebrush steppe associated species were present in the area (Sauer et al. 2002), particularly in large expanses of habitat, but the extent to which they used smaller meadows was unclear. We expected that the shape, size, and vegetation of high-elevation sagebrush patches would affect bird and small mammal diversity patterns. We also evaluated to what extent that high elevation meadows might substitute for loss of habitat at lower elevations.

**Methods**

**Study area and site selection**

The study area was located in the Bear River Range portion of the Wasatch Mountains in northeastern Utah. There is a mix of public and private lands in these
mountains, with the majority of the land public, and managed by the USDA Forest Service as part of the Wasatch-Cache National Forest. There are, however, a few small parcels of state-owned school trust land and a large state wildlife area (Hardware Ranch) located at the southern end of the study area. All sites selected were on public land.

Selection of the study sites involved several steps. First, 50 random reference points that met specific criteria were selected throughout the study area. The criteria included the following: 1) the surrounding area was predominantly public land; 2) the matrix consisted of forest with sagebrush meadows interspersed; and 3) the reference points were within 2.5 kilometers of a road for accessibility. Thirty-four reference points met the required conditions. Second, the nearest meadow to the reference point was selected provided: 1) it had a sagebrush component; and 2) its overall slope was <45%. We calculated slope using ArcView GIS and a digital elevation model. The presence of sagebrush was determined from aerial photos, but included ground verification as well. Meadows were digitized by hand using Digital Ortho Quarter Quads (DOQQ). Third, sample points within each meadow were randomly selected in proportion to meadow size, and ranged from 2-10 points. All points were at least 250 meters apart, except when the meadow was too small, in which case two points were randomly selected within the meadow. Sample points occurring in water or on roads were discarded. Our resulting sampling density was approximately one point per 6 hectares of meadow.

The 34 meadows were quite variable, and ranged in size from 0.6 to 781 hectares. Elevations ranged from 1,810 to 2,900 meters. Vegetation, geology, soils,
and climate were highly diverse throughout the study. Three species of sagebrush were found to occur in these meadows: *Artemisia nova* A. Nels., *A. arbuscula* Nutt., and *A. tridentata*. *A. tridentata* was further identified to subspecies. Our sites primarily consisted of *A. tridentata*. Nutt. ssp. *vaseyana* (Rydb.) Beetle, but *A. tridentata* Nutt. ssp. *spiciformis* (Osterhout) Kartesz & Gandhi and *A. tridentata* Nutt. ssp. *wyomingensis* Beetle & Young were also found in the study area.

Geologic parent material included Cambrian quartzite, Quaternary surficial deposits, Silurian dolomite, as well as Devonian, Mississippian, Ordovician, Proterozoic, and Eocene age materials (Ramsey 1996). Soils included a variety of combinations of loam, sand, and clay, with sandy loam and clay loam most common.

The Natural Resource Conservation Service (NRCS) operates several Snowpack Telemetry sites (SNOTEL) which are automated to collect snowpack and climate data. SNOTEL sites within the study area include Tony Grove Lake, Temple Fork, Bug Lake, Horse Ridge, Dry Bread Pond, and Monte Cristo. These indicate average annual precipitation for 1971-2000 of between 77.2 cm (30.4 inches) at Bug Lake and 127.8 cm (50.3 inches) at Tony Grove, but in 2003 these two sites received approximately 80% of normal precipitation. Less than average precipitation at these points was recorded for 2000, 2001, and 2002 as well. Data for 2004 were not yet available at the time of the defense (NRCS 2005). The forest matrix in which the meadows were imbedded was also diverse, with meadows surrounded by combinations of aspen *Populus tremuloides* Michx., bigtooth maple *Acer grandidentatum* Nutt., and various conifers, including limber pine *Pinus flexilis* James, Douglas fir *Pseudotsuga menziesii* (Mirbel) Franco subalpine fir *Abies*
lasiocarpa (Hook.) Nutt., Engelmann spruce *Picea engelmannii* Parry ex Engelm., and Rocky Mountain juniper *Juniperus scopulorum* Sarg. In addition to sagebrush, other shrubs also were frequently present. These included mountain snowberry *Symphoricarpos oreophilus*, western chokecherry *Prunus virginiana* L., Antelope bitterbrush *Purshia tridentata* (Pursh) DC, and Woods' rose *Rosa woodsii* Lindl.

**Bird sampling**

We used point transects (variable distance point counts) to determine bird species composition and relative densities. Point transects are bird point counts that include distance estimates to each bird seen or heard, that can then be used to calculate bird density (Buckland et al. 1993). We used laser range finders to determine most bird distances, but shorter distances (less than 10 meters) were estimated or paced. By recording distances, densities and a measure of detectability for each species could be calculated. However when density or relative abundance measurements are used in analysis, the underlying assumption is that higher densities of animals will be found in better habitat. Studies done to confirm this by looking at fitness have found that this assumption often does not hold true, as higher densities do not necessarily correlate with higher quality habitat (Van Horne 1983; Railsback et al. 2003). Additionally, due to the incursion of trees and forested areas into our sampled areas, we were concerned that our density estimates could be biased. Frequently, portions of the surveyed area would extend outside of the meadows. We compensated for this by using a survey effort calculation based on the portion of the surveyed area within the meadow, and observations made outside of the meadow were discarded.
However, we often recorded only a small number of individuals at each point and this could potentially bias our results. For example, two Green-tailed Towhees might be found at each of two points. If one point had a survey effort of 1.0, and the other of 0.75, the density of Green-tailed Towhees at the point with the smaller survey effort would be calculated as higher than that at the larger survey effort. As a result, we used species occurrence data in our analysis. This requires the assumption that species are more likely to be found in better habitat, which is a more reasonable and parsimonious assumption.

We conducted counts for 8 minutes at each point. This ensured that we identified both the maximum number of birds while using time as efficiently as possible (Bibby et al. 2000). Point transects began mid-May and ended after the first week in July. Counts began soon after sunrise, which was approximately 5 a.m. and continued no later than 10:30 a.m. A modified double-observer method was used to ensure greater detection rates and compensate for observer detection differences (Nichols et al. 2000). The primary observer identified all birds seen and heard and estimated distances from the point to each bird. The second observer recorded the primary observer’s observations, took bearings to each bird, and also independently recorded any birds seen or heard that the primary observer had missed. The two observers switched roles at each new point. Birds that flew over the meadow without landing were recorded separately. We conducted additional bird point surveys in the surrounding landscape within a 6-kilometer area surrounding the initially selected random points and used these counts to compile species lists. We did not use these data in any other aspect of the analysis.
Small mammal sampling

The same meadows selected for bird point counts were used to conduct the small mammal trapping. Because of the greater effort required for small mammal sampling, we trapped at 11 of the 34 study sites located within one kilometer of a road. Meadow sizes ranged from 0.7 hectares to 182 hectares. Trapping was conducted at two meadows during the first field season and at nine meadows during the second. For each meadow, we set two trap webs for three consecutive trap nights at two randomly selected sample points. Each web was centered at the sample point, with eight trap lines radiating from each center point. Traps points were established at points located 0, 5, 10, 20, 30, 40, and 50 meters from the center along each of the radiating lines for a total of 98 traps in each web. We used both Victor rat snap traps and Sherman live traps in order to catch as great of a variety of species as possible (Mengak & Guynn 1987; Jones et al. 1996). Two traps were set at each point; one Sherman and one rat snap trap. Peanut butter, oatmeal, birdseed, and chopped raisins were mixed together and used as bait in the snap traps, and maple-coated horse feed was used in the live traps. We set the traps the first morning of each trap week, then checked them each morning and evening for three trap nights, and collected them in the morning of the fourth day. We closed live traps during the day to prevent heat-related deaths, but reset them each evening. Kill traps remained set both day and night. A representative sample of dead specimens from each meadow was collected and donated to the Utah Museum of Natural History as voucher specimens. All
trapping was done in accordance with acceptable practices approved by the Institutional Animal Care and Use Committee at Utah State University.

**Meadow and habitat characteristics sampling**

We measured a variety of meadow and habitat characteristics. We determined meadow size, perimeter, and edge, as well as parent material, using an ArcView Geographic Information System. Edge was calculated by:

$$ E = A(P^{-1}) $$

where $E =$ edge, $A =$ meadow area, and $P =$ meadow perimeter. Parent material was obtained from Utah Geologic Map (Ramsey 1996). We sampled vegetation and soil characteristics in the field. We assessed plant community composition at each point using two perpendicular 10-meter transects centered at each sample point. We created species lists for all vascular species encountered in a five-minute timed survey, measured basal coverage for litter, bare ground, and rock, and estimated canopy coverage of all live shrubs, grasses, and forbs. To accomplish this, nine 10 x 50 centimeter quadrats were systematically established at each sample point, and were used to estimate cover (Daubenmire 1959). We also measured cover for each shrub species, but used line intercept methods (Canfield 1941) rather than Daubenmire methods. Additionally, all shrubs within a one-meter radius of the center point were identified to species and age class (Gatsuk et al. 1980), and in each quadrat the dimensions of the shrub nearest the center point were recorded (Martin et al. 1997). Plant species specimens are located in the Intermountain Herbarium as vouchers. We conducted soil sampling at the small mammal sample points. We used
an auger to determine soil depth, texture, and rock component (Schoeneberger et al. 2002).

**Data treatment**

We compiled a list of sagebrush and shrubsteppe associates from Dobkin and Sauder (2004), Paige and Ritter (1999), and Braun et al. (1976). The list was then distilled by removing species whose known distributions did not include Utah, Idaho, or Wyoming. We included species found in Idaho or Wyoming even if their known distributions did not include Utah because our study area was near the border of both states (Tables 2.1 and 2.2). Each individual species presence became a response variable.

To analyze species presence against meadow size and edge, we conducted basic logistic regressions for each species. Relationships were examined using a single value for each meadow, so that all effects were fixed. These data distributions were examined for normality and outliers, and were log transformed.

We aggregated the soil measurements to create a soil suitability index, with greater depth increasing the value of the index, and higher rock content reducing it. Proportions of shrubs, graminoids, and forbs were calculated by dividing the cover measurement of each by the cover of all green material combined. This was calculated in order to test for responses of relative compositions of plants in addition to total amounts. Prior to analysis, we examined the linearity of the relationship between each explanatory variable and each response variable. We used a mixed model and hence the assumptions of normality were relaxed, however we still
examined the data distributions. Variables were transformed to achieve more uniformly distributed data as well as linear relationships between the response and explanatory variables. Line intercept coverage values for *A. tridentata* and for *S. oreophilus* Gray, and the proportion of *Artemisia* spp. as related to total shrub cover all required square-root transformations. Daubenmire cover estimates for bare ground were log transformed. In some cases, relationships were determined to be quadratic and, as a result, additional variables representing squared components of the polynomial were added.

**Statistical analyses**

To test the correlation between meadow size and edge, we conducted a basic logistic regression using PROC LOGISTIC in SAS. The presence of each species of interest within each meadow was regressed against meadow size and edge. PROC LOGISTIC was also used to analyze species co-occurrences within meadows. For the analysis of species occurrences against explanatory habitat variables, generalized mixed models were used, allowing us to analyze the results using each sample point rather than losing variability by creating a composite value for each meadow. Sample points were selected within meadows and were not independent; therefore, meadows were analyzed as random effects. While a generalized linear model requires independence of all points, by assuming that all variables are fixed, a generalized linear mixed model allowed us to analyze the data while taking into account that points within each meadow are more likely to be related than those in different meadows (Littell et al. 1996). The SAS macro program GLIMMIX was used (Version
02 June 2002; it is available at http://www.sas.com/techsup/download/stats). This macro iteratively calls SAS PROC MIXED until model convergence is achieved. Each model parameter estimate was fitted using the restricted likelihood method as per Wolfinger and O’Connell (1993) (Littell et al. 1996). Only univariate models were used. Models were deemed significant for $P \leq 0.05$.

**Results**

We detected only one of the four sagebrush obligate bird species of interest, the Brewer’s Sparrow. For shrubsteppe associate avian species, we detected 9 out of 21 upland species, and 8 of the 11 riparian species. We encountered 4 shrubsteppe bird species at points other than those conducted in the focal meadows (Table 2.1).

We caught one of two sagebrush obligate mammal species, the sagebrush vole but not pygmy rabbit. For shrubsteppe-associated small mammals, we trapped 1 out of the ten possible upland species and three of six possible riparian species (Table 2.2).

Brewer’s Sparrows, the only avian sagebrush-obligate species detected, were more likely to be present in larger meadows and in meadows with less edge. They were positively correlated with sites high in green cover ($P = 0.032$), and also were more often present at sites with higher densities of shrubs ($P = 0.0112$), in particular of *A. tridentata* ($P = 0.0407$). Their presence was negatively correlated with bare ground ($P = 0.008$).

Of the avian shrubsteppe associates, Green-tailed Towhee (*Pipilo chlorurus*), Vesper Sparrow (*Pooecetes gramineus*), White-crowned Sparrow (*Zonotrichia*
leucophrys), Orange-crowned Warbler (Vermivora celata), and Chipping Sparrow (Spizella passerina) were present at enough sites for us to conduct analyses. Green-tailed Towhee presence increased with big sagebrush cover (P = 0.0005), but decreased as snowberry coverage increased (P = 0.0004). A quadratic relationship existed between Green-tailed Towhee presence and the proportion of sagebrush relative to the total amount of green matter (P = 0.0271). Green-tailed Towhees were least likely to be present when this ratio was very high or very low. They also showed a quadratic relationship in relation to the amount of bare ground, and were more often detected in areas with moderate amounts of bare ground (P = 0.0011).

Vesper Sparrows were also less likely to occur as the amount (P = 0.0002) and proportion of shrubs (P < 0.0001) increased. In particular, we found a negative correlation with S. oreophilus cover (P = 0.0316). We did not find a significant relationship with the amount of sagebrush. Vesper Sparrows were more likely to be present at sites with greater forb cover (P = 0.0269) and at sites with moderate proportions of graminoids (P = 0.02). Vesper Sparrows were positively correlated with meadow size (P = 0.0303) and negatively correlated with edge (P = 0.022) and with elevation (P = 0.0245).

White-crowned Sparrows were more likely to be present at higher elevations (P = 0.0271). They were positively correlated with greater forb coverage (P = 0.0447), and negatively correlated with graminoid coverage (P = 0.0188) and with the proportion of graminoids to other vegetation (P = 0.0048).

Orange-crowned Warblers decreased as elevation increased (P = 0.0022), but no other explanatory variables exhibited significant relationships with this response.
variable. Orange-crowned warblers were only present at 16 points, however, making relationships difficult to discern.

Chipping Sparrow presence showed no significant correlation with any of the vegetative explanatory variables. The only correlation detected was with meadow size. We found a positive correlation between meadow size and Chipping Sparrow detections ($P = 0.0469$).

Although Gray Flycatcher (*Empidonax wrightii*), Prairie Falcon (*Falco mexicanus*), Western Meadowlark (*Sturnella neglecta*), Brewer's Blackbird (*Euphagus cyanocephalus*), Yellow Warbler (*Dendroica petechia*), and Willow Flycatcher (*Empidonax traillii*) were all detected at points within the study meadows, they were not sufficiently abundant to allow statistical analyses.

There were also several species which tended to co-occur. Brewer's Sparrows and White-crowned Sparrows were positively correlated ($P = 0.0383$). Orange-crowned Warblers were positively correlated with both MacGillivray's Warblers ($P = 0.0111$) and with Vesper Sparrows ($P = 0.0111$). MacGillivray's Warblers and Vesper Sparrows were not correlated, however ($P = 0.2642$).

For small mammal sagebrush-obligate species, sagebrush voles *Lemmiscus curtatus* were present at 3 out of 22 points, and at 2 out of 11 meadows. There were too few presences to determine relationships with explanatory variables.

Of the shrubsteppe-associated species, only long-tailed voles (*Microtus longicaudus*) showed significant relationships with any of the explanatory variables. Long-tailed voles were actually less likely to occur at points with high sagebrush
coverage (P = 0.0066), and more likely at points with high amounts of forb coverage (P = 0.0073).

Least chipmunks (*Tamias minimus*) were trapped at all meadows except one, Montane voles (*Microtus montanus*) were only found at two meadows, and western jumping mice (*Zapus princeps*) were trapped at only one meadow. Although Great Basin pocket mice (*Perognathus parvus*) were trapped at 5 out of 11 meadows, no significant relationships with any of the explanatory variables were detected. None of the small mammal species of interest showed any correlation, either positive or negative, with meadow size or edge.

**Discussion**

A variety of sagebrush obligate and shrubsteppe associate vertebrates were identified in the study, although several important sagebrush obligate vertebrates were not detected. We did not detect Greater Sage-Grouse or pygmy rabbits in the study, but our methods were unlikely to have detected them, even if they were present. Sage Thrashers and Sage Sparrows, however, were notably absent. We did detect a high number of shrubsteppe associates, with 21 of the 48 species found within the study area.

All species that demonstrated a correlation with meadow size and edge were more likely to be encountered in larger meadows and in meadows with less edge. None of the species analyzed showed either a negative correlation with meadow size, or a positive correlation with edge. However, different species of interest were correlated with different habitat variables, suggesting that there was no one type of
sagebrush-dominated meadow preferable to all species of interest. Vegetation characteristics that correlated with species presence were different for each species. Only a few species showed evidence of co-occurrence.

All significant correlations with meadow size and edge indicated a similar message: larger meadows with less edge were more likely to contain the species of interest. Although Brewer’s Sparrows were found in meadows as small as 2 hectares, they were much more likely to occur in larger meadows. It is possible that they are supplementing the sagebrush habitat found in one meadow with that found in others nearby. Interestingly, higher proportions of riparian than upland shrubsteppe-associated bird species were detected. This was not the case for small mammals. Our mammal trap sites were rarely located near riparian areas, and never in them, and mammal detections were in meadows without any known surface water other than the occasional small, muddy stock pond.

Our fieldwork was conducted during the summers of 2003 and 2004, during an ongoing drought which has affected much of the West. Bird species, in particular, may have been affected by this. The more mesic high-elevation shrubsteppe may have experienced different use patterns as a result. It would be of interest to determine in future studies whether species are more abundant at these higher, more mesic areas in or following drought years. It would also be of interest to determine the nest success rates of various species in these sagebrush meadows as compared with those in the larger sagebrush expanses. While we know that many species use these meadows, fitness was not measured.
Our study was limited to meadows, and did not include sampling in larger expanses of sagebrush, as these did not qualify as being within a forest matrix and were predominately owned by private interests. Breeding Bird Surveys (Sauer et al. 2002) have identified Sage Thrashers in a nearby area of open sagebrush. Greater Sage-Grouse, not surveyed for with our study design, have also been found in nearby high-elevation areas. However, our focus was on sagebrush-dominated mountain meadows, not all high elevation sagebrush habitats. It is possible that these species require larger, uninterrupted sagebrush expanses, and that multiple meadows in close proximity may not be substitutable (sensu Dunning et al. 1992). Despite the lack of detection of some of these species, these meadows do appear to be important habitat for a number of the shrubsteppe species. They are not, however, substitutable for low elevation habitat. While these meadows are used by a large number of shrubsteppe species, and are important in their own right, their conservation should serve as a supplement to, rather than a substitute for, the conservation of low elevation shrubsteppe habitat.

**Literature Cited**


TABLE 2.1. Bird species detected within study area. * Denotes species which were detected within the study area, but not within one of the study meadows.

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<td>Vireo vicinor</td>
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<tr>
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<td>Chondestes grammacus</td>
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<td>Loggerhead Shrike</td>
<td>Lanius ludovicianus</td>
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<td>Icterus parisorum</td>
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<tr>
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<td>Tympanuchus phasianellus</td>
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<tr>
<td>Vesper Sparrow</td>
<td>Poocetes gramineus</td>
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<tr>
<td>Virginia's Warbler</td>
<td>Vermivora virginiae</td>
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<tr>
<td><strong>Shrubsteppe Associates Riparian Species</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Belted Kingfish</td>
<td>Ceryle alcyon</td>
<td>*</td>
<td>*</td>
<td>*</td>
</tr>
<tr>
<td>Bullock's Oriole</td>
<td>Icterus bullockii</td>
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<td>MacGillivray's Warbler</td>
<td>Opornis tolmiei</td>
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<td>11</td>
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<tr>
<td>Nashville Warbler</td>
<td>Vermivora ruficapilla</td>
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<td></td>
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<tr>
<td>Orange-crowned Warbler</td>
<td>Vermivora celata</td>
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<tr>
<td>Song Sparrow</td>
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<td>*</td>
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<tr>
<td>Swainson's Thrush</td>
<td>Catharus ustulatus</td>
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<td>Veery</td>
<td>Catharus fuscescens</td>
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<td>Willow Flycatcher</td>
<td>Empidonax traillii</td>
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<tr>
<td>Yellow Warbler</td>
<td>Dendroica petechia</td>
<td>8</td>
<td>4</td>
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### TABLE 2.2. Mammal species trapped within study area.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th>Points</th>
<th>Meadows</th>
<th>Individuals</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pygmy Rabbit</td>
<td>Brachylagus idahoensis</td>
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<td>3</td>
<td>2</td>
</tr>
<tr>
<td>Sagebrush Vole</td>
<td>Lemmiscus curtatus</td>
<td></td>
<td>2</td>
<td>1</td>
</tr>
</tbody>
</table>

**Sagebrush Obligates**

| Least chipmunk               | Tamias minimus                   |        | 20      | 10          | 151         |
| Chisel-toothed kangaroo rat  | Dipodomys microps                |        |         |             |             |
| Dark kangaroo mouse          | Microdipodops megacephalus       |        |         |             |             |
| Desert woodrat               | Neotoma lepida                   |        |         |             |             |
| Great Basin pocket mouse     | Perognathus parvus               |        | 10      | 6           | 27          |
| Idaho ground squirrel        | Spermophilus brunneus            |        |         |             |             |
| Little pocket mouse          | Perognathus longimembris        |        |         |             |             |
| Merriam’s shrew              | Sorex merriami                   |        |         |             |             |
| Piute ground squirrel        | Spermophilus canus               |        |         |             |             |
| Preble’s shrew               | Sorex preblei                    |        |         |             |             |

**Shrubsteppe Associates Upland Species**

| Long-tailed vole             | Microtus longicaudus             | 9       | 6       | 15          |
| Montane vole                 | Microtus montanus               | 4       | 3       | 20          |
| Townsend’s pocket gopher     | Thomomys townsendii             | 4       | 3       | 20          |
| Water shrew                  | Sorex palustris                 | 4       | 3       | 20          |
| Western harvest mouse        | Reithrodontomys megalotis       | 2       | 1       | 3           |
| Western jumping mouse        | Zapus princeps                  | 1       | 1       | 3           |

**Shrubsteppe Associates Riparian Species**
A COMPARISON OF DOUBLE-OBSERVER AND DISTANCE METHODS FOR MODELING BIRD DETECTABILITY¹

Abstract

Ornithologists have long relied on bird point counts to evaluate population trends, habitat relationships, and species distributions. A major flaw in using raw count numbers is that the assumption that they accurately represent the total number of birds present is seldom met. Empirical modeling to compensate for differences in detectability across species, space, and time is gaining popularity. We evaluated two of several methods currently in use. We used both the double-observer method and the distance method in our study design and present here a comparison of our results. Using the same data, distance sampling suggested that the detection probabilities of seven different bird species ranged from 0.16 to 0.80. The double-observer method gave a detection value of almost 1 (0.9979) for all seven species, however, an analysis of graphed distance data suggested these values were too high. We conclude that the double-observer method may have a variety of benefits, but under conditions similar to those we faced, does not provide reliable estimates of detectability.

Introduction

Terrestrial bird counts have long been used in studies of habitat relationships, bird distribution, and population trends of birds. The most common method used has been unadjusted bird counts that use comparisons of avian abundance without regard

¹ Coauthored by Johnson, Elizabeth J., Tammy L. Wilson, and John A. Bissonette. Department of Forest, Range, and Wildlife Sciences, Utah State University, Logan, UT.
to issues of detectability. Rosenstock et al. (2002) found that 95% of studies using landbird count data that were published in major journals from 1989 to 1998 used unadjusted index counts as opposed to empirical modeling techniques. This is indeed problematic because unadjusted index counts can be highly misleading (Thompson 2002). Use of index counts requires the assumption that detectability is consistent across years, locations, or both, so that counts at one time or place can be compared to counts at other times or places. This assumption is seldom valid (Burnham 1981, Nichols et al. 2000, Rosenstock et al. 2002). In reality, detectability is most often highly variable and serious problems result. It is possible that the number of birds may remain constant over time but detectability decreases, leaving the impression the number of birds is decreasing. Conversely, the number of birds may in fact have changed, but we would have no way of knowing. For example, if 100 birds are detected in a given area, and ten years later, 100 are again detected, use of index counts would not allow us to determine if bird numbers really stayed the same or changed. Empirical models would allow a conclusion: e.g., in the first year there were 400 birds in the surveyed area, but only 200 in the tenth year, a substantial decrease.

Despite the clear problems, many ornithologists continue to rely on index counts (Rosenstock et al. 2002). There are several suggested methods which investigators have available to them, but all require significant effort in data analysis or collection. Methods currently include double sampling (Bart and Earnst 2002), removal (Farnsworth et al. 2002), double-observer method (Nichols et al. 2000), and distance sampling (Buckland et al. 2001). Double sampling involves conducting traditional counts at all sites followed by more expensive complete counts at a
random subset of the sites. Removal methods involve recording birds in time
intervals, and analyzing the data using probabilities of detection during each interval.
The double-observer method requires two or more technicians to conduct each
survey. Differences between detections are then used to determine detectability of
each species. This can be done using dependent or independent observers.
Independent observers each conduct the count at the same time, but ignore the other
observer. Results are later compared. Dependent-observers require one observer to be
the primary observer and to identify all birds while the other records the observations
of the primary observer while simultaneously noting any bird missed by the primary
observer. Distance sampling involves recording distances to each bird from a line or
point. It is assumed that all birds on the point will be detected, and that the proportion
of birds detected will decrease with distance. There are advantages and disadvantages
to each method, and while it is unlikely that one approach will be found to be the
“best,” ornithologists need to consider pros and cons for each in order to make
informed decisions. Moore et al. (2004) provided a comparison of two of these
methods: removal and modified independent double-observer modeling. We add to
this by comparing dependent double-observer modeling with distance modeling. Both
methods were used simultaneously, allowing us to compare the results from each
using the same data set.

Each approach requires different assumptions. Distance modeling requires
three critical assumptions (Buckland et al. 2001): 1) All birds on the line or point are
detected with certainty; 2) Birds are detected at their initial locations; 3)
Measurements to each bird or cluster of birds are exact. For double-observer methods,
the assumptions are that an observer has the same probability of detecting a species regardless of whether the person is serving in a primary or secondary role, and that the detections made by the primary observer are not influenced by those made by the secondary observer (Nichols et al. 2000).

Methods

Our study was conducted in sagebrush-dominated mountain meadows in the Wasatch- Cache National Forest of northeastern Utah. Thirty-four point locations were selected randomly throughout the study area, with the meadow nearest the point selected as the meadow of interest. Additionally, a 6-kilometer square was centered on the point, and additional meadows within the landscape were also surveyed. A total of 299 sample points within meadows were used in this analysis. Independence of sampling point locations is not an assumption when comparing the two methods.

Prior to the beginning of the first field season, we provided 2 weeks of intensive training for technicians. For the second season, half of our technicians returned, so we provided only 1 week of training. We conducted surveys using 8-minute point transects, also called variable distance bird point counts, and two observers. Each point was sampled once, between sunrise and 1030 Mountain Daylight Time from mid-May through the first week of July in 2003 and 2004. Using laser range finders, we estimated the distance from the point to each bird. Most laser range finders are less accurate at close range, so the distance to birds within 10 meters was estimated or paced. All birds seen or heard were recorded, but observers were instructed to focus on birds within a 100-meter radius.
Our two observers played different roles as either "primary" or "secondary" observer at each successive point. In the dependent double-observer method, the primary observer indicates to the secondary observer all birds detected. The secondary observer records the data, but also records separately any birds he or she detects that were not indicated by the primary observer. Secondary observers were allowed to assist with identification as necessary. Because we also were using the distance method, the primary observer was responsible for providing distance estimates to each bird. For birds missed by the primary observer, the secondary observer would estimate distances after the point transect was concluded. The secondary observer recorded data typically gathered in point counts including species code, number of birds in cluster, time period (0-3, 3-5, or 5-8 minutes), type of observation (aural, visual, or both), and a box was checked if the bird(s) flew overhead but did not land. In addition to these typical data measurements, the secondary observer measured and recorded direction to the bird, which was used for a separate analysis, and whether the bird was located in meadow, tree within the meadow, forest, forest stringer, or riparian area. The primary observer was responsible for providing all information to the secondary observer except the direction measurement. The roles of primary and secondary observer were alternated at each point.

Distance data was analyzed using Program Distance (Thomas et al. 2004). We combined data from both observers and analyzed each species separately. We chose to do distance analyses for meadow species with more than 75 detections. These were American Robins (*Turdus migratorius*), Brewer’s Sparrows (*Spizella breweri*),
Chipping Sparrows (*Spizella passerina*), Dark-eyed Juncos (*Junco hyemalis*), and Green-tailed Towhees (*Pipilo chlorurus*). We also ran analyses for Vesper Sparrows (*Pooecetes gramineus*) and White-crowned Sparrows (*Zonotrichia leucophrys*); even though these had 49 and 40 individuals detected, respectively. These were meadow species of particular interest to our study. Our surveys were conducted in heterogeneous habitats. We surveyed in meadows, but frequently detected forest birds as well. We only analyzed data from species known to associate within meadows. In initial evaluations, we truncated the data at 100 meters and grouped the data into 10-meter intervals (Buckland et al. 2001). Truncation is recommended by Buckland et al. (2001) to delete outliers and facilitate modeling. Grouping of the data into intervals improves robustness when rounding errors in distance measurements has resulted in a disproportionate number of distances which have been rounded to distances such as 5 or 10 or 50 meters, i.e. heaping of the data. We evaluated models using Akaike’s Information Criterion (AIC), and adjusted for small sample bias (AICc). The model with the lowest AICc was selected for each of the seven species. Data analyzed using the double-observer method was also truncated at 100 meters, providing consistency in the data used for each analysis while also defining a reasonable area in which both observers could be expected to make the same detections. Dobserv software (Hines 2000) was used. AICc values were again used in selecting the best model.

**Results**

The best double-observer model ignored both species and observer effects and produced very high probabilities of detection (0.9979) for all species (Table 3.1). If
we had selected the model with a higher AICc value which differentiated between species, our detection values would have ranged from 0.9917 for Vesper Sparrows and 0.9999 for Dark-eyed Juncos. In our distance analysis we ran separate models for each species, resulting in different detection probabilities for each species. Our lowest detectability using distance methods was for Chipping Sparrows, and our highest was for American Robins (Table 3.1).

**Discussion**

The double-observer analysis suggested that we had detected all individuals for all seven species within the 100-meter radius. When we examined the distance results, however, this did not appear correct. The graphs showed that we met the assumption of fewer detections with increasing distance from the point, indicating that we did not detect all birds within the 100-meter radius.

Our results from the double-observer method do not appear to be correct. We have no concrete way to assess whether the fault is inherent in the method or inherent in its implementation. Our secondary observers did not detect many more birds than those detected by the primary observer. We recorded 992 observations of all seven species combined; only 43 were secondary observer detections, an addition of 4.66%. We asked ourselves: Why the difference? At least two possibilities present themselves. First, it is possible that the primary observers were detecting everything that could reasonably be detected during the count, leaving little for the secondary observer. Alternatively or correspondingly, the secondary observers may not have been able to make significant additional detections due to the requirements of
recording the data. If the first possibility holds, then the observers did an excellent job and all birds were indeed observed. With regards to recording duties, one of the assumptions of the double-observer method is an equal probability of detection for both observers. We followed the recommended protocol, but gave each observer one additional task; the primary observer used the laser range finder to measure distance and the secondary observer used a compass to take a bearing to the bird. Since both observers were given an additional task, we argue that the duties of the secondary observer was not a significant factor in observing birds beyond normal protocol. Nichols et al. (2000) have suggested using a third person to act solely as note-taker, allowing each observer to focus solely on species detections. If cost is a limiting factor this may not be economically feasible. We expect that most ecological studies are not well funded.

Our implementation of the double-observer method failed to achieve the expected results, however other studies also have had mixed results, suggesting inherent difficulties with the methodology. Moore et al. (2004) found similar estimates of detectability between removal and double-observer methods, with double-observer detection rates between 0.54 and 0.68. They used independent observers, whereas we used dependent observers. Forcey (2002) compared results from studies that used dependent vs. independent double-observer methods and had similarly high detection values when using the dependent methods. He calculated a mean point specific detection probability of 0.970 with dependent observers, and a lower rate of 0.767 when independent observers were used. Nichols et al. (2000) reported an average detection probability of 0.67 for “difficult” species and 0.86 for
“easy” species while using dependent observers and a count duration of only 3 minutes for their point transects. It is possible that longer count times allowed a primary observer to detect birds that otherwise would have been noticed by the secondary observer alone. Indeed, it was common for the secondary observer to notice birds that the primary observer did not detect until later in the point count. Duration of point counts may be a critical element of the method that needs additional study in an environment where the true number of species is known or can be estimated with some precision.

The costs and benefits appear more straightforward for distance sampling. Distance sampling requires fewer technicians in the field. Range finders are relatively inexpensive (~$250) when compared to the cost of additional technicians, but the amount of time needed to analyze the data is substantial. There are, however, legitimate concerns regarding the assumptions required by distance sampling: 1) Birds at zero distance from the point must always be detected; 2) birds need to be detected at their initial location, without any movement in response to the observer; and 3) distances must be measured accurately. We hope that all birds on the point were detected with certainty, but cannot evaluate this.

The second assumption can be evaluated by looking at data plots. For example, plots of our Green-tailed Towhee distance data suggest that not all birds were detected prior to movement. We saw fewer individuals near the point than would be predicted if birds were not moving away from the observers (Fig. 3.1). We noted this same pattern for Brewer’s Sparrows, but not for the other species.
The third assumption, that the distances to birds are measured accurately, seems easier to accomplish, at least for visual detections near the point. Distances to nearby birds are more likely to be accurate, but as distance from the point increases, there is a tendency for birds to be heard and not seen, making distance measurements more problematic. We did see some heaping in the data, i.e., observers tended to round the data towards multiples of 5 and 10 when making estimates (Fig. 3.2). To compensate, we used the interval option in Program Distance allowing a grouping of observations so that heaping had less of an effect. This tendency to round may be common and was discussed with observers during training, but remained an issue. An additional concern is the lack of a “shoulder” in data patterns from distance methods. A detection probability of 1.0 at the point is an assumption but, it is desirable to have a high probability of detection in the area immediately surrounding the observation point as well. Hence the data pattern should ideally show a “shoulder” (Fig 3.1). We detected two different patterns in our data. For Chipping Sparrows, it appears that detection probabilities dropped rapidly with distance from the point (Fig. 3.3), with no shoulder present. A shoulder, such as that seen for Green-tailed Towhees (Fig. 3.1), gives greater confidence in results obtained. While exact estimates of detectability are likely to be imprecise, Program Distance does provide confidence intervals to reflect uncertainties such as these.

Analysis of the double-observer data was much more straightforward and less subjective than the distance analysis. Program Distance requires more training and experience to use appropriately, and requires greater quantities of data in order to run properly. We found it necessary to aggregate our data into 10-meter intervals and use
a 100-meter truncation. Additionally, while AICc could be used in model selection, other criteria are valid and if used, could result in different models being selected. What this means is that other researchers working with the same data set could generate different results. For example, when we analyzed our Chipping Sparrow distances, the model with the lowest AICc involved a negative exponential key function, but another model, the hazard-rate key function, also provides a good visual fit (Figs. 3.3 and 3.4). It is important to recognize this source of difference. Models can be pooled for greater robustness (Buckland et al. 2001), but managers who want quick and easy results may be disappointed. While the learning curve for Program Distance was steeper and required much more time to analyze, we consider both Program Distance and Dobserv reasonably user-friendly. Program Distance is accompanied by a 219-page on-line user guide (i.e., Thomas et al. 2004) and a 431-page printed text (i.e., Buckland et al. 2001), while Dobserv is summed up in a three short pages (i.e., Hines 2000). Both programs are available for free download on the World Wide Web.

In addition comparing the results of each analysis, there were other important considerations. Even though the double-observer method does not allow detectability corrections of the data, it does offer other benefits. The quality and consistency of the data appeared to be higher as a result of having two observers. In addition to offering a few more detections, the secondary observer contributes in less obvious ways. Analysis showed our data to be quite consistent across observers. During model selection, we ran both Dobserv and Program Distance using observer identity as a covariate. AICc values were not improved in either program. By having two people
conduct point transects together and assist each other in identification, observers
continued to receive training throughout the field season. When pairs are rotated each
person has the opportunity to both teach and learn from the other technicians. Often
bird surveys are conducted by technicians working alone. When this is the case, there
is a good possibility that identification mistakes will not be caught, and bird
misidentifications could continue throughout the season. Clearly, the data quality is
highly dependent on the ability and honesty of each individual. It is likely that a
primary observer is more motivated to detect and correctly identify the birds when
working with a partner than when working alone. This extra motivation, regardless of
detections made by the secondary observer, can be used constructively to generate
higher-quality data. Additionally, having technicians work in pairs rather than alone
is safer from a management perspective. Steep terrain, rattlesnakes and other
dangerous animals, and remote sites without cell phone coverage all imperil safety.

One of the assumptions of bird surveys is that the birds do not respond to
being surveyed (Bibby et al. 2000). This may not be a valid assumption with one
technician, even less so with two. We do not know if having two people at a point has
a greater influence on bird behavior than one person would, but studies showing that
birds respond to a variety of cues including clothing (i.e., Gutzwiller and Marcum
1993), make it impossible to rule out.

In conclusion, both methods have their costs and benefits. For empirical
modeling of the data in order to compensate for detectability, distance sampling
appears superior to the dependent double-observer method. The dependent double-
observer method failed to produce usable results in our study. Using both methods
simultaneously, however, may be an even better solution when financially possible.
The double-observer method could be used as quality control and the distance method
for detectability modeling.

**Literature Cited**

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325 in Estimating Numbers of Terrestrial Birds (S. C. J. Ralph and J. M.

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Forcey, G. M. 2002. An evaluation of double-observer point count techniques and
avian habitat use on the Camp Dawson collective training area, Preston
County, West Virginia. Master's thesis, West Virginia University,


Table 3.1. Probabilities of detection for bird species using Program Distance ($p_1$) and Program Dobserv ($p_2$) with standard errors for each ($SE_1$ and $SE_2$, respectively).

<table>
<thead>
<tr>
<th>Species</th>
<th>Program Distance</th>
<th>Program Dobserv</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>$p_1$</td>
<td>$SE_1$</td>
</tr>
<tr>
<td>American Robin</td>
<td>0.79861</td>
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</tr>
<tr>
<td>Brewer's Sparrow</td>
<td>0.34193</td>
<td>0.01679</td>
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<tr>
<td>Chipping Sparrow</td>
<td>0.15666</td>
<td>0.02139</td>
</tr>
<tr>
<td>Dark-eyed Junco</td>
<td>0.20598</td>
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</tr>
<tr>
<td>Green-tailed Towhee</td>
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<td>0.20621</td>
<td>0.05976</td>
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<tr>
<td>White-crowned Sparrow</td>
<td>0.33929</td>
<td>0.06398</td>
</tr>
</tbody>
</table>

Although the "best" model, using AICc criterion provides a single detection probability of for all species as 0.9979 with a standard error of 0.0006, we show the results of each species individually to allow comparison between models.
Figure 3.1. Green-tailed Towhee detection probabilities. Bars are based on actual detections, line indicates predicted values.
Figure 3.2. Green-tailed Towhee detection probabilities when data is not converted into intervals. Note the increased probabilities of detection around 15, 25, 35, 40, 50, 60 and 70-meter detection distances.
Figure 3.3. Chipping Sparrow detection probabilities modeled with a uniform key function and cosine series expansion.
Figure 3.4. Chipping Sparrow detection probabilities modeled using a hazard rate key function.
Our goal in conducting this study was to determine the contribution of high-elevation sagebrush shrubsteppe meadows to bird and small mammal biodiversity. In particular, we were interested in sagebrush obligate and shrubsteppe-associated species, and undertook to determine the extent to which they use these higher-elevation habitats. Additionally, we wanted to determine which habitat variables were associated with presences and densities of each species. In determining our methods, we also undertook to empirically model the densities of certain birds and small mammals rather than use simple, and potentially misleading, count indices. In doing so, we were able to evaluate and compare various methods in which densities could be determined.

Our results indicate that high-elevation sagebrush-dominated mountain meadows in northeastern Utah are important habitat for a number of sagebrush obligates and shrubsteppe associates, but that they are not used by all of the species of interest. Relationships between species presence and meadow characteristics were quite variable. The presence of several bird species was positively correlated with meadow size, but for many species no correlation, either positive or negative, was detected. We found correlations of presence with a variety of vegetational characteristics, but each species showed different relationships so that no single type of meadow structure could be identified as "best."
Of the four avian sagebrush-obligate species only Brewer’s Sparrow was detected (25%), and of the two sagebrush obligate small mammals, sagebrush voles were caught; pygmy rabbits were not (50%). Of the shrubsteppe associates, 17 of the 32 avian species (53.1%) were detected, but only 4 of the 16 small mammal species (25%).

It appears that although mountain meadows are important habitat for a number of small vertebrate species, they are not fully complimentary. It is clear that high-elevation sagebrush-dominated mountain meadows cannot substitute for loss of lower elevation sagebrush shrubsteppe habitat, at least in regard to birds and small mammals.

**Survey Effort and Density Calculations**

It had been our original intention to analyze the data using densities of birds and small mammals rather than presences. However, issues with the reliability of the density data presented problems. Each point was located randomly within a meadow, which meant that for some points a portion of the survey area would lay outside of the meadow. To compensate, survey efforts for each point were calculated. When birds were recorded, their distance, bearing, and location (meadow, forest, tree within meadow, etc) also was recorded, and used to map the location of each bird. Some species; e.g., Vesper Sparrows and Brewer’s Sparrows, were always found within the meadow, whereas many other meadow-associated species, e.g., Green-tailed towhees and White-Crowned Sparrows, were found along the forest edge and into the forest approximately 20 meters. We used ArcView GIS to determine what proportion of our
100-meter point radii were in the meadow and meadow buffer. We used this proportion as an estimate of our survey effort. We then used Program Distance to calculate bird densities at each point. When we compared the data distributions with meadow metrics, they appeared highly unusual and suggested a bias in the data. Meadow size was log transformed, and the data analyzed using PROC GLM in SAS. Survey effort and meadow size were positively correlated with an $R^2$ value of 0.79 ($P < 0.0001$). Smaller meadows were more likely to have points with smaller survey effort values because randomly selected points in a small meadow are more likely to be near a meadow edge than those in a large meadow. With relatively few birds at each point, survey efforts of less than one systematically increased density estimates for these points. For example, if there was one Brewer’s Sparrow at a point with a survey effort of 1.0, and one Brewer’s Sparrow at a point with a survey effort of 0.75, the point with the smaller survey effort would have been calculated to have a higher density of birds, clearly an erroneous result. Plots of the data showed Brewer’s Sparrows densities at their highest in the smallest meadows, dropping rapidly, and then gradually becoming denser again in the largest meadows. Similar plots were obtained for all other species examined. Because these density estimates were untrustworthy, we chose to analyze the data using species occurrences. This provided an additional benefit of fewer assumptions. It is frequently assumed that higher densities indicate better quality habitat, but this may be misguided (Van Horne 1983, Railsbeck et al. 2003). It is a safer assumption that a species is more likely to be present in high quality habitat than in low quality habitat.
more species with higher priority rank values will have a higher CV index than a meadow with more birds, if those birds have a low priority rank value, e.g., European Starlings. CV indices typically are calculated using relative abundance or density estimates. As the reliability of these was in question, any values calculated with them would also be questionable. Regardless, a preliminary analysis using relative abundance and relative density values showed no significant relationships between either CV indices or richness with any of the meadow variables that we measured.

**Recommendations for Research and Management**

We determined that a variety of sagebrush obligate, shrubsteppe associate, and forest birds used the habitat provided by sagebrush meadows with a montane forest matrix (Table 4.1). We also determined that while several species were found to be more likely to occur in large meadows, there were no meadow characteristics that appeared significantly related for all species. Some species, e.g., long-tailed vole *Microtus longicaudus*, were negatively correlated with a high sagebrush component, whereas others, such as Brewer’s sparrows and green-tailed towhees were positively correlated with increasing sagebrush. A variety of meadow sizes and vegetative characteristics appeared to be important when managing for multiple shrubsteppe species.

Although sagebrush meadows were used by a variety of species, we were unable to measure fitness for any of these species. Additionally, source-sink dynamics were not addressed. Populations of animals in meadows are likely to function as part of a larger metapopulation. Metapopulations may be composed of
Although sagebrush meadows were used by a variety of species, we were unable to measure fitness for any of these species. Additionally, source-sink dynamics were not addressed. Populations of animals in meadows are likely to function as part of a larger metapopulation. Metapopulations may be composed of source populations which produce a surplus of individuals such that some must emigrate elsewhere, and sink populations which require regular immigration from other populations (Pulliam 1988). Due to the presence of many shrubsteppe species, we believe that this habitat is important to shrubsteppe birds and small mammals, but we did not determine whether these meadows serve as population sources or sinks for each species. It is also possible that meadow populations are quite isolated. A genetic analysis of select species to determine the amount of genetic exchange between individuals found in meadows and those found in the larger expanses would be interesting and might provide a better understanding of the role of meadow habitat in the larger metapopulation dynamics. If sagebrush-dominated meadows function as a source to other populations, it will be especially important to manage these areas so that gene flow between populations can continue.

**Literature Cited**


Table 4.1. List of birds identified within the study area during point transects.

<table>
<thead>
<tr>
<th>Common Name</th>
<th>Scientific Name</th>
<th>Common Name</th>
<th>Scientific Name</th>
</tr>
</thead>
<tbody>
<tr>
<td>American Kestrel</td>
<td><em>Falco sparverius</em></td>
<td>Mourning Dove</td>
<td><em>Zenaida macroura</em></td>
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<td>American Robin</td>
<td><em>Turdus migratorius</em></td>
<td>Northern Flicker</td>
<td><em>Colaptes auratus</em></td>
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<tr>
<td>Belted Kingfish</td>
<td><em>Ceryle alcyon</em></td>
<td>Northern Harrier</td>
<td><em>Circus cyaneus</em></td>
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<td>Black-billed Magpie</td>
<td><em>Pica hudsonia</em></td>
<td>Olive-sided Flycatcher</td>
<td><em>Contopus cooperi</em></td>
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<td>Black-capped Chickadee</td>
<td><em>Poecile atricapillus</em></td>
<td>Orange-crowned Warbler</td>
<td><em>Vermivora celata</em></td>
</tr>
<tr>
<td>Black-chinned Hummingbird</td>
<td><em>Archilochus alexandri</em></td>
<td>Pine Siskin</td>
<td><em>Carduelis pinus</em></td>
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<td>Black-headed Grosbeak</td>
<td><em>Pheucticus melanoccephalus</em></td>
<td>Plumbeous Vireo</td>
<td><em>Vireo plumbeus</em></td>
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<td>Blue Grouse</td>
<td><em>Dendragapus obscurus</em></td>
<td>Prairie Falcon</td>
<td><em>Falco mexicanus</em></td>
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<td>Blue-gray Gnatcatcher</td>
<td><em>Polioptila caerulea</em></td>
<td>Red-breasted Nuthatch</td>
<td><em>Sitta canadensis</em></td>
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<td>Brewer's Blackbird</td>
<td><em>Euphagus cyanoccephalus</em></td>
<td>Red-naped Sapsucker</td>
<td><em>Sphyrapicus nuchalis</em></td>
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<td>Brewer's Sparrow</td>
<td><em>Spizella breweri</em></td>
<td>Red-tailed Hawk</td>
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<td>Broad-tailed Hummingbird</td>
<td><em>Selasphorus platycercus</em></td>
<td>Rock Wren</td>
<td><em>Salpinx obsoletus</em></td>
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<td>Ruby-crowned Kinglet</td>
<td><em>Regulus calendula</em></td>
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<td>California Gull</td>
<td><em>Larus californicus</em></td>
<td>Sandhill Crane</td>
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<td>Cassin's Finch</td>
<td><em>Carpodacus cassinii</em></td>
<td>Song Sparrow</td>
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<td>Chipping Sparrow</td>
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<td>Spotted Towhee</td>
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<td><em>Catharus ustulatus</em></td>
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<td>Common Nighthawk</td>
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<td>Townsend's Solitaire</td>
<td><em>Myioborus townsendi</em></td>
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<td>Turkey Vulture</td>
<td><em>Cathartes aura</em></td>
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<td>Vesper Sparrow</td>
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<td><em>Junco hyemalis</em></td>
<td>Violet-green Swallow</td>
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<td><em>Picoides pubescens</em></td>
<td>Warbling Vireo</td>
<td><em>Vireo gilvus</em></td>
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<td>Western Bluebird</td>
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<td>Western Kingbird</td>
<td><em>Tyrannus verticalis</em></td>
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<td><em>Dumetella carolinensis</em></td>
<td>Western Meadowlark</td>
<td><em>Sturnella neglecta</em></td>
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<td>Western Tanager</td>
<td><em>Piranga ludoviciana</em></td>
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<td>Green-tailed Towhee</td>
<td><em>Pipilo chlorurus</em></td>
<td>Western Wood-Peewee</td>
<td><em>Contopus sordidus</em></td>
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<td>Hairy Woodpecker</td>
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<td>White-crowned Sparrow</td>
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<td><em>Catharus guttatus</em></td>
<td>White-throated Swift</td>
<td><em>Aeronautes saxatalis</em></td>
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<td><em>Carpodacus mexicanus</em></td>
<td>Williamson's Sapsucker</td>
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<td>House Wren</td>
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<td>Willow Flycatcher</td>
<td><em>Empidonax traillii</em></td>
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<td>Lazuli Bunting</td>
<td><em>Passerina amoena</em></td>
<td>Wilson's Warbler</td>
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<td>MacGillivray's Warbler</td>
<td><em>Oporornis tolmiie</em></td>
<td>Yellow Warbler</td>
<td><em>Dendroica petechia</em></td>
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<tr>
<td>Mountain Bluebird</td>
<td><em>Sialia currucoides</em></td>
<td>Yellow-rumped Warbler</td>
<td><em>Dendroica coronata</em></td>
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<tr>
<td>Mountain Chickadee</td>
<td><em>Poecile gambeli</em></td>
<td></td>
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</tr>
</tbody>
</table>
APPENDIX
Dear Tammy Wilson:

I am in the process of preparing my thesis in the Department of Forest Range, and Wildlife Sciences at Utah State University. I hope to complete in Spring 2005.

I am requesting your permission to include the attached material as shown. I will include acknowledgments to your work as shown. You will be cited as a co-author on the title page of each chapter of which you are a co-author. Please advise me of any changes you require.

Please indicate your approval of this request by signing in the space provided, attaching any other form or instruction necessary to confirm permission. If you have any questions, please call me at the number above.

Thank you for your cooperation,
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Signed ____________________________

Date 12 April 2005

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