

Utah State University

DigitalCommons@USU

---

All Graduate Theses and Dissertations

Graduate Studies

---

8-2023

## Anthropogenic Factors Affecting Common Raven Occurrence and Depredation of Artificial Nests Within Greater Sage-Grouse Habitat in Southern Utah

Zoë S. Moffett  
*Utah State University*

Follow this and additional works at: <https://digitalcommons.usu.edu/etd>



Part of the [Poultry or Avian Science Commons](#)

---

### Recommended Citation

Moffett, Zoë S., "Anthropogenic Factors Affecting Common Raven Occurrence and Depredation of Artificial Nests Within Greater Sage-Grouse Habitat in Southern Utah" (2023). *All Graduate Theses and Dissertations*. 8827.

<https://digitalcommons.usu.edu/etd/8827>

This Thesis is brought to you for free and open access by the Graduate Studies at DigitalCommons@USU. It has been accepted for inclusion in All Graduate Theses and Dissertations by an authorized administrator of DigitalCommons@USU. For more information, please contact [digitalcommons@usu.edu](mailto:digitalcommons@usu.edu).



ANTHROPOGENIC FACTORS AFFECTING COMMON RAVEN OCCURRENCE  
AND DEPREDATION OF ARTIFICIAL NESTS WITHIN GREATER SAGE-  
GROUSE HABITAT IN SOUTHERN UTAH, USA

By

Zoë S. Moffett

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

of

MASTER OF SCIENCE

in

Wildlife Biology

Approved:

---

S. Nicki Frey, Ph.D.  
Major Professor

---

Michael Conover, Ph.D.  
Committee Member

---

Eric Gese, Ph.D.  
Committee Member

---

D. Richard Cutler, Ph.D.  
Vice Provost for Graduate  
Studies

UTAH STATE UNIVERSITY  
Logan, Utah

2023

Copyright © Zoë Moffett 2023

All Rights Reserved

## ABSTRACT

Anthropogenic Factors Affecting Common Raven Occurrence and Depredation of  
Artificial Nests within Greater Sage-grouse Habitat in Southern Utah

by

Zoë S. Moffett, Master of Science

Utah State University, 2023

Major Professor: Dr. S. Nicole Frey  
Department: Wildland Resources

As generalist opportunistic predators, common ravens (*Corvus corax*; hereafter raven) have been found to pose a serious threat to many sensitive species in the Intermountain West. For example, ravens have become one of the primary nest predators of the greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse). Raven populations have been growing throughout the distribution of sage-grouse, primarily because of anthropogenic subsidies that provide food, water, and habitat to the ravens. This is of particular concern within the southern most extent of the sage-grouse distribution where their populations are small and fragmented. The purpose of my research was to explore the potential risks that ravens pose to sage-grouse in southern Utah, given the presence of various human activities that surround and permeate sage-grouse habitat. In Chapter 2, I used point-count survey data to determine the three most significant predictors for raven occurrence within and near sagebrush vegetative communities: the calendar date of the survey, proximity to agricultural practices, and the presence of livestock. In Chapter 3, I conducted an artificial sage-grouse nest experiment

within different sagebrush vegetative treatments and found that nests placed within sagebrush that had been treated for pinyon-juniper removal were more likely to be depredated than nearby intact (i.e. untreated, undisturbed) sagebrush. I also found that ravens were more likely to depredate nests under shrubs that were in close proximity to a nest that the ravens had also found. My research highlights some of the more significant factors that may increase the threat that local ravens pose to the sage-grouse populations in southern Utah. By understanding the impacts of livestock, agriculture, and habitat treatments on the depredation rate of sage-grouse nests by ravens, land managers may better plan mitigation and conservation strategies.

(108 pages)

## PUBLIC ABSTRACT

Anthropogenic Factors Affecting Common Raven Occurrence and Depredation of  
Artificial Nests within Greater Sage-grouse Habitat in Southern Utah

Zoë S. Moffett

Certain species of wildlife are more generalist and adaptive than others. These species often flourish when supported by human activities that provide additional food and habitat for them. The common raven (*Corvus corax*; hereafter raven) is one such species; their populations have risen and spread throughout the Intermountain West. As generalist scavengers and predators, ravens have been found to pose a severe threat to several threatened or sensitive species, including the greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse). The purpose of my research was to determine the specific anthropogenic and habitat factors that may increase the threat that ravens pose as sage-grouse nest predators in southern Utah. To do this, I conducted surveys for ravens in the winter and spring months (February-June) in 2021 and 2022. I also completed artificial nest experiments during the same years by placing chicken eggs underneath sagebrush with the intent of determining what habitat factors led to more instances of nest depredation by ravens. Through my surveys, I found that raven presence was significantly affected by the calendar date of the survey, how far the survey was from agricultural practices, and whether or not livestock was present during the survey. During the artificial nest experiment, I found that ravens were more likely to find and depredate nests that were within habitat that had been treated for pinyon-juniper removal compared to intact sagebrush. I also found that the nests were more likely to be depredated if they

were placed underneath a shrub that was close to a nest that a raven had found and depredated. My research will help to inform sage-grouse management decisions that pertain to sagebrush vegetation treatments and raven mitigation strategies.

## CONTENTS

	Page
Abstract.....	iii
Public Abstract.....	v
List Of Tables .....	ix
List Of Figures .....	x
Chapter I Introduction and Literature Review .....	1
Common Raven Ecology and Natural History .....	1
Habitat and Range.....	1
Diet.....	2
Movement .....	2
Greater Sage-Grouse Ecology and Natural History .....	3
Habitat Management.....	4
Populations.....	7
Nest Depredation .....	8
Methods to Survey Ravens in Relation to Habitat.....	10
Methods for Artificial Nest Experiments.....	11
Literature Cited .....	13
Chapter II Livestock Production and Agricultural Practices Impact Common Raven Occurrence Within Greater Sage-Grouse Habitat in Southern Utah, USA .....	23
Abstract .....	23
Introduction.....	24
Study Area .....	27
Methods.....	28
Point-Count Surveys .....	28
Data Analysis .....	29
Results.....	31



Discussion.....	33
Management Implications.....	37
Literature Cited.....	38
Figures.....	45
Tables.....	50
Chapter III Artificial Nest Depredation by the Common Raven Within Various Sagebrush Treatments in Southern Utah, Usa.....	52
Abstract.....	52
Introduction.....	53
Study Area.....	59
Methods.....	60
Artificial Nest Experiment.....	61
Data Analysis.....	63
Results.....	64
Discussion.....	67
Management Implications.....	70
Literature Cited.....	71
Figures.....	81
Tables.....	86
Chapter IV Conclusions.....	91
Appendices.....	95

## LIST OF TABLES

	Page
Table 2.1. Parameter estimates of fixed effects from fitted generalized linear mixed model (family = negative binomial, link = logit) using data from 810 point-count surveys of the common raven ( <i>Corvus corax</i> ), Iron and Beaver counties, Utah, USA; February–June 2021–2022.....	50
Table 2.2. Parameter estimates of fixed effects from fitted generalized linear mixed model (family = binomial, link = logit) where the binary response variable was raven occurrence (present or absent); the data was from 810 point-count surveys of the common raven ( <i>Corvus corax</i> ), Iron and Beaver counties, Utah, USA; February–June 2021–2022.....	51
Table 3.1. Variables that were selected after single variable logistic regression analyses describing depredation of 115 artificial nests ( $p > 0.15$ ), Iron County, Utah, USA, 2021–2022.....	86
Table 3.2. Top performing generalized linear models predicting artificial nest outcome per study site that were within 2 Akaike’s information criterion units (adjusted for small sample sizes; AICc) ( $n = 115$ ), Iron County, Utah, 2021–2022.....	88
Table 3.3. Parameter estimates for the top 3 performing generalized linear models (family = binomial, link = logit) for the Buckskin Valley study site that were within 2 Akaike’s information criterion units (adjusted for small sample sizes; AICc) ( $n = 115$ ), Iron County, Utah, USA, 2021–2022.....	89
Table 3.4. Parameter estimates for the top 2 performing generalized linear models (family = binomial, link = logit) for the Bald Hills study site that were within 2 Akaike’s information criterion units (adjusted for small sample sizes; AICc) ( $n = 115$ ), Iron County, Utah, USA, 2021–2022.....	90
Table A1. Parameter estimates of the fixed effects from a fitted generalized linear mixed model (family = negative binomial, link = logit) where the outliers were removed ( $n = 733$ surveys; no. of groups = 87), Iron and Beaver counties, Utah, USA; February–June 2021–2022.....	97
Table B1. Top logistic regression models for artificial nest outcomes in the Bald Hills study site with year included as a parameter. ....	98

## LIST OF FIGURES

	Page
Figure 2.1. Map of Sage-grouse Management Areas (SGMAs) in the state of Utah, USA.....	45
Figure 2.2. Map of survey points with 1 km buffer around each point.....	46
Figure 2.3. Rate of raven detection (number of times ravens were detected / total surveys) across all surveys from February–June 2021–2022.....	47
Figure 2.4. Expected number of ravens observed at a given survey point according to ordinal date (February-June) (with 95% confidence interval).....	48
Figure 2.5. Expected number of ravens observed at a given survey point (with 95% confidence interval) according to the survey point’s distance to agricultural practices (e.g., cropland) and according to livestock presence.....	49
Figure 3.1. Map of Sage-grouse Management Areas (SGMAs) in the state of Utah, USA.....	81
Figure 3.2. Map of artificial nest outcomes for 2022 within the Bald Hills study site, Iron County, Utah, USA.....	82
Figure 3.3. Map of artificial nest outcomes for 2022 within the Buckskin Valley study site, Iron County, Utah, USA.....	83
Figure 3.4. Trail camera image of a raven depredating an artificial nest in May of 2021, Iron County, Utah, USA.....	84
Figure 3.5. Artificial nest outcomes in the Buckskin Valley depending on distance of artificial nest to the nearest depredated nest.....	85
Figure A1. DHARMA’s residual diagnostic plots for the fitted generalized linear mixed model: number of ravens ~ (1  survey point and year) + livestock present + log(distance to ag) + ordinal date.....	96

## CHAPTER I

### INTRODUCTION AND LITERATURE REVIEW

#### **Common Raven Ecology and Natural History**

Breeding bird surveys indicate that common raven (*Corvus corax*; hereafter raven) populations within the Great Basin have been increasing by approximately 3% every year since 1966; this is slightly higher than the 2.5% average annual increase recorded across the United States (Sauer et al. 2019). Much of this growth in raven populations can be attributed to anthropogenic activities and structures that provide subsidies (nesting habitat, food, and water) for ravens (O’Neil et al. 2018). The most prevalent nesting and roosting subsidies that humans provide are transmission or power lines as they provide ample support and space for large stick nests (Coates et al. 2014, Restani and Lueck 2020). Food subsidies can be found in landfills (Kristan et al. 2004, Peebles and Conover 2017), along paved roads (i.e. roadkill; Kristan et al. 2004, Harju et al. 2018), and throughout agricultural (Engel and Young 1992) and livestock operations (Coates et al. 2016). While many of these subsidies are seemingly inevitable in areas more populated with humans, there is a cause for concern when anthropogenic subsidies support raven populations in areas where they would not normally flourish, such as the semi-arid sagebrush (*Artemisia* spp.) habitats throughout the Great Basin (Coates et al. 2020, Duerr et al. 2021).

*Habitat and Range.* Ravens can be found across the throughout North America, northern Europe, Siberia, central and western China, and even in parts of northern Africa. In the early 1900’s, there was a steep decline in raven populations throughout the eastern

United States; this can partly be attributed to shooting and poisoning efforts (Boarman and Heinrich 2020). However, by the mid 1900's, raven populations began to rise across the country (Harju et al. 2021). As is suggested by their widespread range, ravens are extreme habitat generalists and have high behavioral plasticity (Boarman and Heinrich 1999). Within the western U.S., they are found in open habitat such as shrublands, tundra, and deserts. However, a large reason why ravens flourish in these systems is because of the anthropogenic subsidies provided by urban development, agricultural fields, landfills, and roadways (Webb et al. 2011, Coates et al. 2020, Duerr et al. 2021).

*Diet.* Ravens are generalist omnivores, foragers, and scavengers; they have a wide diet and they have been known to exploit novel resources when found (Boarman and Heinrich 1999). As their populations have increased and spread throughout the west (Harju et al. 2021), there has been a focus on how ravens affect several sensitive species of wildlife. The three primary prey species of concern are the desert tortoise (*Gopherus agassizii*; Kristan and Boarman, 2003), snowy plover (*Charadrius nivosus*; Ellis et al. 2020), and the greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse; Dinkins et al. 2012 and 2014, Lockyer et al 2013, Coates et al. 2020, Coates et al. 2021b). Generalist omnivores like ravens pose a significant threat to these sensitive prey species. Because ravens can rely so heavily on human subsidies, their populations may continue to grow even as their natural prey species' numbers begin to dwindle, thus exasperating the negative effect the ravens can have on these populations (Sinclair et al. 1998, Coates et al. 2021b, Dinkins et al. 2021).

*Movement.* Little is known about raven movement and dispersal throughout the western United States. In general, ravens aren't believed to be migratory; breeding adults

are believed to remain in their range year-round (Boarman and Heinrich 2020). Like most birds, raven movements differ dramatically between the breeding and non-breeding seasons. One study in Wyoming found that during the breeding season, breeding ravens traveled an average of 0.8 km (range 0.5-1.3 km) every day while nonbreeding ravens traveled an average of 10.4 km (range 4.1-26.3 km) every day (Harju et al. 2018). The breeding ravens traveled short distances from their nests to nearby subsidies such as roads or railroads; the nonbreeding ravens were more likely to travel greater distances to point-source subsidies such as landfills. A separate study within the Mojave Desert of southern California tracked raven movements during the nonbreeding season (September to December) and found that ravens traveled on average 41.5 km (range 0.13-206.1 km) each day (Duerr et al. 2021). Duerr et al (2021) found that while ravens rely heavily on urban areas with numerous subsidies (both food and habitat), during the nonbreeding season they spend a lot of time in less developed areas where there are no anthropogenic subsidies.

### **Greater Sage-grouse Ecology and Natural History**

Before European settlement of North America, sage-grouse occupied a range that was almost double what it is today (Schroeder et al. 2004). This decrease in the distribution (and in the overall population) of sage-grouse has in large part been attributed to habitat loss and fragmentation (Connelly and Braun 1997, Schroeder et al. 2004). Much of the habitat loss is fairly easy to qualify with human expansion into sagebrush for agricultural uses (Connelly and Braun 1997). As a sagebrush obligate, sage-grouse are unique in that they require different types of sagebrush seasonally and within different life stages. Their life history requires specific habitat-types for the winter

(non-breeding), lekking, nesting, early brood-rearing, and late brood-rearing seasons (Dahlgren et al. 2016 and 2019). During the nesting season, hens require moderately dense sagebrush with a mixture of forbs and grasses that provide vertical concealment for the hen and her nest (Dahlgren et al. 2016). While mammalian predators rely on olfactory cues for depredating nests, avian predators such as ravens likely depend on their ability to see the movement of the hen on or near the nest (Coates and Delehanty 2008). Therefore, when sagebrush habitat becomes overly dense, resulting in decreased forb and grass cover, or when cheatgrass (*Bromus tectorum*) dominates the native forbs and grasses, sage-grouse hens and their nests have significantly less concealment (Connelly et al. 2000).

*Habitat Management.* There are several habitat management strategies that aim to improve sage-grouse nesting and brood-rearing habitat; these methods are central to sage-grouse conservation efforts within Utah (Utah Division of Wildlife Resources 2019). Recently, the 2 primary management strategies for improving sagebrush habitat within the Intermountain West are pinyon-juniper removal and sagebrush thinning/reseeding (Dahlgren et al. 2006, Frey et al. 2013, Schupp et al. 2015). The former is important as pinyon-juniper forests encroach into sagebrush systems (Miller et al. 2008), and the latter is necessary as sagebrush systems grow too old and dense due to fire suppression (Pyke et al. 2014, Ellsworth et al. 2016). Both management strategies involve a suite of mechanical removal options as well as the option of seeding to increase native grasses, forbs and shrubs (Schupp et al 2015, Cook et al. 2017, Utah Division of Wildlife Resources 2019).

During the last century there has been a dramatic increase in both pinyon pine (*Pinus edulis*, *Pinus monophylla*) and juniper (*Juniperus* spp.; pinyon-juniper) within the Intermountain West. Much of this expansion has been into sagebrush ecosystems where the increase in canopy cover has led to a decrease in shrubs, forbs, and grasses (Miller et al. 2000). It has been demonstrated that sage-grouse tend to avoid stands of pinyon-juniper adjacent to sagebrush ecosystems and that the sage-grouse who avoid these trees have a higher rate of survival (Coates et al. 2017). As land managers have been working to increase and improve sage-grouse habitat, there have been massive efforts to remove stands of pinyon-juniper in order to restore sagebrush ecosystems. One analysis found that within six years (2011-2013 to 2015-2017), 1.6% of pinyon-juniper habitat had been reduced (either from wildfire or by management) across the Great Basin and into the Columbia and Colorado Plateaus (Reinhardt et al. 2020). Reinhardt et al. (2020) estimate this rate of reduction is just about matching the rate of expansion of pinyon-juniper throughout the range. During the six years that were analyzed, about 691 km<sup>2</sup> of pinyon-juniper were reduced within the state of Utah, and the vast majority of that management took place within areas designated as Priority Areas for Conservation (PACs) for sage-grouse (Reinhardt et al. 2020).

There are several treatment options for reducing or removing pinyon-juniper stands within sagebrush ecosystems. The choice of treatments can depend on management goals including what successional phase of encroachment the habitat is in on a scale of 1 to 4 (early, mid, late, and closed), which is based on tree density and the amount of sagebrush understory (Miller et al. 1999). Managers in Utah have used the following treatments to remove pinyon-juniper: mastication (e.g., bull hog), hand-



thinning (e.g., lop and scatter), chaining, and fire (Cook et al. 2017, Utah Division of Wildlife Resources 2019, Reinhardt et al. 2020). Multiple studies have found that sage-grouse will use the mixed sagebrush/forb/grass communities as it grows back post pinyon-juniper removal (Frey et al. 2013, Cook et al. 2017). However, little is known about sage-grouse nest and brood survival rates within sagebrush communities that have been treated for pinyon-juniper removal.

In addition to pinyon-juniper removal, sagebrush stands can sometimes require restoration via sagebrush thinning or removal. This form of restoration becomes necessary as stands of sagebrush become overly old and dense with consequential losses of native perennial grasses and forbs; this is an issue for both livestock managers who need more vegetation for their cattle and for wildlife biologists who want more forbs for breeding sage-grouse (Connelly et al. 2004). These dense stands of sagebrush can also lead to higher intensity wildfires with high connectivity across large sections of habitat (Schupp et al. 2015). Whether it is for livestock foraging, wildlife conservation, or for woody fuels reduction, land managers may decide to use one or more of the following methods to reduce sagebrush cover: prescribed fires, mechanical methods, herbicide applications, or livestock management (Connelly et al. 2004).

While there are many studies that investigate the vegetation structure of sagebrush ecosystems following these treatments (e.g., Davies et al. 2012, Porensky et al. 2020, Pyke et al. 2022, Roth et al. 2022), there has been limited research on how these differing sagebrush treatments impact sage-grouse populations throughout the year. One such study took place on Parker Mountain in Utah where it was found that sage-grouse broods preferentially utilized sagebrush habitat that had been treated with Tebuthiuron, a

defoliant, compared to control plots (Dahlgren et al. 2006). One of the primary conclusions of this study was that sage-grouse use was highest within 10 m of the treatment boundary, indicating that they were still associating with the adjacent sagebrush cover in the control plots. A separate study in northern Utah used an artificial nest experiment to determine differences in nest depredation rates between treated (chained and seeded 25 years prior) and untreated sagebrush stands; they found that fewer nests were depredated in the treated habitat, although the treatment had occurred quite a long time before the study took place (Ritchie et al. 1994). Overall there is a gap in the literature when it comes to sage-grouse activities throughout the year within habitat that has been treated for sagebrush thinning or removal.

*Populations.* Across the Western US, sage-grouse are estimated to occupy about half of their historical range (compared to pre- European colonization) (Schroeder et al. 2004), and from 1966 to 2019, it has been estimated that sage-grouse abundance range-wide has declined by about 81% (Coates et al. 2021a). The primary way that wildlife managers monitor sage-grouse populations is via lek surveys, where observers count the number of males (and sometimes females) present at individual leks (Utah Division of Wildlife Resources 2019). During the breeding season, male sage-grouse gather within flat, open habitats where they perform a courtship display called a “strut”; the strutting display is where the males move around the open area and perform a series of auditory “plops” by inflating and deflating their large esophageal pouches (i.e., throat sacs) (Schroeder et al. 2020). One of the main difficulties with monitoring sage-grouse populations via lek counts is that changes in male counts year to year may not accurately reflect changes in populations. Males may utilize different lekking grounds in different

years, and wildlife managers may not find and observe the smaller leks (Coates et al. 2021a).

It is also difficult to say which limiting factors are the most significant to sage-grouse population growth, as it is likely that this varies across their range. Conover and Roberts (2016) reviewed and summarized some of the literature concerning this question and suggested several factors related to recruitment that may be important. This included increases in nest and brood predators as well as losses of suitable sagebrush habitat that can provide the necessary dietary requirements for sage-grouse chicks (namely forbs and associated insects) (Conover and Roberts 2016).

In Utah, breeding bird surveys show an average annual decrease of 0.7% in sage-grouse populations between 1966-2019. However, between 2000-2019 the state saw an average annual increase of 1.7% (Sauer et al. 2019). While wildlife managers continue to improve and augment sage-grouse habitat, they are continuously monitoring the populations throughout the state. Other than lek counts, rates of nest success are crucial to quantifying the status of a population; without nest survival and recruitment, the population declines over time.

*Nest Depredation.* There are numerous habitat factors that play into nest success (grass and forb height, sagebrush height and cover, presence of taller shrubs, etc.), however the primary cause for nest failure is often nest depredation (Moynahan et al. 2007, Lockyer et al. 2013).

Recorded nest predators of sage-grouse include ravens, black-billed magpies (*Pica hudsonia*), American badgers (*Taxidea taxus*), western gopher snakes (*Pituophis*

*catenifer*), coyotes (*Canis latrans*), elk (*Cervus elaphus*), weasels (*Mustela* spp.), and bobcats (*Lynx rufus*) (Holloran and Anderson 2003, Coates et al. 2008, Lockyer et al. 2013). A review by Conover and Roberts (2017) found that ravens were the most common nest predator, responsible for 35% of depredated nests across 13 different sage-grouse studies. In northwestern Nevada, Lockyer et al. (2013) found that ravens were the most frequent sage-grouse nest predator, depredating about 47% of the total depredated nests. There have also been multiple studies that specifically looked at how the presence of ravens negatively impacts the success rate of nesting sage-grouse (Bui et al. 2010, Gibson et al. 2018, Coates et al. 2020). Each of these studies quantified how raven activity could be attributed to the anthropogenic subsidies discussed above. For instance, Coates et al. (2020) were able to use 10 years of raven surveys in the Great Basin to determine anthropogenic and environmental covariates associated with raven detections; they found that sage-grouse nest survival was negatively impacted where raven density exceeded about 0.40 km<sup>-2</sup>.

Because both sagebrush communities and anthropogenic activities vary greatly throughout the western US, it is difficult to extrapolate the conclusions of one sage-grouse depredation study to a different region. While the above studies took place primarily in Nevada and Wyoming, there has been little to no research on raven activity in sage-grouse habitat in Utah. It is important to conduct similar studies in Utah for several reasons. First, sage-grouse populations throughout Utah have historically been, and continue to be, fairly fragmented due to the nature of the landscape and due to anthropogenic activities (Schroeder et al. 2004, Utah Division of Wildlife Resources 2019). As sage-grouse habitat continues to become more fragmented throughout its entire

range, it is useful to use Utah's sage-grouse populations as indicators for what we may expect to see within other populations. Also, Utah has unique characteristics for sage-grouse nesting and brood-rearing habitats that considerably differ from that of other portions of the species' range (Dahlgren et al. 2016). Sage-grouse populations within southern Utah may be disproportionately affected by raven activities as they are often smaller and more fragmented than those in other portions of their range (Frey et al. 2013, Beers 2023).

The primary objectives of this research are therefore to study and quantify the threats that ravens may pose to sage-grouse populations in southern Utah. There are 2 parts to this project that will help with these objectives. The first part works to investigate raven relative abundance as it relates to different habitat factors (both environmental and anthropogenic). The second part of the project is to quantify the potential for nest depredation by ravens within sagebrush habitat that has undergone various types of management/treatments.

### **Methods to Survey Ravens in Relation to Habitat**

While research has been conducted to survey ravens within sage-grouse habitat in Idaho, Nevada, and Wyoming (Bui et al. 2010, Dinkins et al. 2012, Coates et al. 2016, Peebles and Conover 2017, Harju et al. 2018, and O'Neil et al. 2018, Coates et al. 2020), there has been no reported studies monitoring ravens within these systems in Utah. These projects monitored ravens either via point-count surveys or via transmitter data (telemetry or GPS transmitters). Point-count surveys have been used to estimate raven densities in relation to habitat covariates or in relation to sage-grouse populations (Bui et al. 2010,

Dinkins et al. 2012, O'Neil et al. 2018, Coates et al. 2020); spatial monitoring has been used to study raven habitat selection and movement distances (Peebles and Conover 2017, Harju et al. 2018). Most raven point-count surveys have taken place during the sage-grouse breeding and brood-rearing seasons (March-August) and occurred at survey points identified via stratified random sampling across known sage-grouse habitat. Harju et al. (2018) found that nonbreeding ravens in their study system flew on average 10.4 km from their central roost; this means that it may be important to have studies include survey points that are outside of the sagebrush communities since ravens are able to fly from nontarget habitats into sage-grouse habitats. There have been limited raven survey efforts in the state of Utah, and there is little research into raven habitat use within sagebrush ecosystems during the winter (other than Peebles and Conover 2017).

### **Methods for Artificial Nest Experiments**

One of the most common ways to study questions regarding nest depredation events is to set up “artificial nests” that can mimic the nests of a study species. Within sage-grouse research, artificial nests have been used to compare depredation rates between differing nesting habitats (on a nest shrub scale and macrohabitat scale) and as a means of identifying nest predators (Ritchie et al. 1994, DeLong et al. 1995, Watters et al. 2002, Dinkins et al. 2013). These artificial nest experiments all involved placing chicken, quail, and/or plasticine eggs underneath sagebrush within a slight bowl impression in the dirt; Watters et al. (2002) and Ritchie et al. (1994) both decided to place their nests at 160 m intervals all within 2 km of an active sage-grouse lek. Only one of these projects included trail cameras on or near the artificial nests in order to identify nest predators (Dinkins et al. 2013), while others did not use cameras and used their best

educated guesses to determine nest predators (or they were not interested in identifying predators) (Ritchie et al. 1994, DeLong et al. 1995, Watters et al. 2002). Dinkins et al. 2013 reported a slight trend of camera presence being positively associated with more depredation events, however they had data limitations that meant that they couldn't include this in their overall model. One artificial nest study in France (Bravo et al. 2020) involved placing artificial ground nests throughout a flat, open farmland habitat; as part of this project, the researchers compared nest outcomes depending on whether or not they placed a trail camera on the nest. Bravo et al. (2020) found that the presence of a camera lead to fewer nest depredations however many predators (mostly corvids) would still visit and not depredated the nest; they hypothesized this behavior was due too neophobia.

In general, artificial nest experiments must be conducted with a lot of intention when it comes to the questions that are being asked and to the conclusions that are being drawn (Dinkins et al. 2013). Dinkins et al. (2013) conducted a study where they placed artificial nests within the nest bowls of both successful and unsuccessful sage-grouse nests. By doing this, they removed the difficult variable within artificial nest studies of nest-site selection as determined by human researchers. Dinkins et al. (2013) discuss three primary elements of artificial nests that likely lead to issues when comparing artificial and real sage-grouse nests: nest placement, nest appearance, and nest vulnerability. Related to these factors, the authors also believe that it is difficult to be confident that just because a species of predator depredates an artificial nest, the same species will depredate actual sage-grouse nests within that habitat.

It is important to understand the above limitations of artificial nest experiments when determining the questions and hypotheses for such research. For this project, I will

be focusing on the predatory behaviors of local ravens rather than drawing conclusions about depredation rates for the sage-grouse populations. I will also include cameras on each artificial nest because it is important for us to be certain what species of predator depredates our nests.

### **Literature Cited**

- Beers, A. T. 2023. Local and Regional Landscape Characteristics Driving Habitat Selection by Greater Sage-Grouse Along a Fragmented Range Margin. All Graduate Theses and Dissertations. 8695. <<https://doi.org/10.26076/4f07-36a8>>. Accessed March 6, 2023.
- Boarman, W. I., and B. Heinrich. 1999. Common Raven. In A. Poole and F. Gill, editors. The birds of North America, Number 476. The Birds of North America, Philadelphia, Pennsylvania, USA.
- Boarman, W. I. and B. Heinrich. 2020. Common Raven (*Corvus corax*), version 1.0. In Birds of the World; S. M. Billerman, Editor. Cornell Lab of Ornithology, Ithaca, NY, USA.
- Bui, T. V. D., J. M. Marzluff, and B. Bedrosian. 2010. Common raven activity in relation to land use in western Wyoming: Implications for Greater Sage-Grouse reproductive success. *Condor* 112:65–78.
- Coates, P. S., B. E. Brussee, K. B. Howe, K. B. Gustafson, M. L. Casazza, and D. J. Delehanty. 2016. Landscape Characteristics and livestock presence influence



common ravens: relevance to greater sage-grouse conservation. *Ecosphere* 7(2):e01203.

Coates, P. S., J. W. Connelly, and D. J. Delehanty. 2008. Predators of Greater Sage-grouse nests identified by video monitoring. *Journal of Field Ornithology* 79(4):421-428.

Coates, P. S. and D. J. Delehanty. 2008. Effects of environmental factors on incubation patterns of greater sage-grouse. *The Condor* 110(4):627-638.

Coates, P. S., K. B. Howe, M. L. Casazza, and D. J. Delehanty. 2014. Common raven occurrence in relation to energy transmission line corridors transiting human-altered sagebrush steppe. *Journal of Arid Environments* 111(2014):68-78.

Coates, P. S., S. T. O'Neil, B. E. Brussee, M. A. Ricca, P. J. Jackson, J. B. Dinkins, K. B. Howe, A. M. Moser, L. J. Foster, and D. J. Delehanty. 2020. Broad-scale impacts of an invasive native predator on a sensitive native prey species within the shifting avian community of the North American Great Basin. *Biological Conservation* 243(2020):108409.

Coates, P.S., Prochazka, B.G., O'Donnell, M.S., Aldridge, C.L., Edmunds, D.R., Monroe, A.P., Ricca, M.A., Wann, G.T., Hanser, S.E., Wiechman, L.A., and Chenaille, M.P. 2021a. Range-wide greater sage-grouse hierarchical monitoring framework - Implications for defining population boundaries, trend estimation, and a targeted annual warning system. U.S. Geological Survey Open-File Report 2020-1154, 243 p. <<https://doi.org/10.3133/ofr20201154>>. Accessed November 28, 2022.

- Coates, P. S., B. G. Prochazka, M. A. Ricca, K. B. Gustafson, P. Ziegler, and M. L. Casazza. 2017. Pinyon and Juniper Encroachment into Sagebrush Ecosystems Impacts Distribution and Survival of Greater Sage-Grouse. *Rangeland Ecology & Management* 70:25-38.
- Coates, P. S., W. C. Webb, S. J. Dettenmaier, S. M. Harju, and D. J. Delehanty. 2021*b*. Synthesis of nest predation impacts of common ravens on sensitive avian species. *Human-Wildlife Interactions* 15(3):350-373.
- Connelly, J. W. and C. E. Braun. 1997. Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. *Wildlife Biology* 3:229–234.
- Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation Assessment of Greater Sage-grouse and Sagebrush Habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming.
- Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. *Wildlife Society Bulletin* 28:967-985.
- Conover, M. R. and A. J. Roberts. 2016. Declining populations of greater sage-grouse: where and why. *Human-wildlife Interactions* 10(2):217-229.
- Conover, M. R. and A. J. Roberts. 2017. Predators, Predator Removal, and Sage-grouse. *Journal of Wildlife Management* 81(1):7-15.

- Cook, A. A., T. A. Messmer, and M. R. Guttery. 2017. Greater Sage-grouse Use of Mechanical Conifer Reduction Treatments in Northwest Utah. *Wildlife Society Bulletin* 41(1):27-33.
- Dahlgren, D. K., R. Chi, and T. A. Messmer. 2006. Greater Sage-grouse Response to Sagebrush Management in Utah. *Wildlife Society Bulletin* 34(4):975-985.
- Dahlgren, D. K., T. A. Messmer, B. A. Crabb, M. R. Kohl, S. N. Frey, E. T. Thacker, R. T. Larsen, and R. J. Baxter. 2019. Sage-Grouse Breeding and Late Brood-Rearing Habitat Guidelines in Utah. *The Wildlife Society Bulletin* 43(4):576-589.
- Dahlgren, D. K., T. A. Messmer, B. A. Crabb, R. T. Larsen, T. A. Black, S. N. Frey, E. T. Thacker, R. J. Baxter, and J. D. Robinson. 2016. Seasonal Movements of Greater Sage-grouse Populations in Utah: Implications for Species Conservation. *Wildlife Society Bulletin* 40(2):288-299.
- Davies, K. W., J. D. Bates, and A. M. Nafus. 2012. Comparing Burned and Mowed Treatments in Mountain Big Sagebrush Steppe. *Environmental Management* 50:451-461.
- DeLong, A. K., J. A. Crawford, D. C. DeLong. 1995. Relationships between vegetational structure and predation of artificial sage grouse nests. *Journal of Wildlife Management* 59(1):88-92.
- Dinkins, J. B., M. R. Conover, C. P. Kirol, and J. L. Beck. 2012. Greater sage-grouse (*Centrocercus urophasianus*) select nest sites and brood sites away from avian predators. *Auk* 129:600–610.

- Dinkins, J. B., M. R. Conover, C. P. Kirol, J. L. Beck, and S. N. Frey. 2014. Greater Sage-Grouse (*Centrocercus urophasianus*) select habitat based on avian predators, landscape composition, and anthropogenic features. *The Condor* 116:629–642.
- Dinkins, J. B., M. R. Conover, C. P. Kirol, and S. T. Mabray. 2013. Do artificial nests simulate nest success of greater sage-grouse? *Human-wildlife Interactions* 7(2):299-312.
- Dinkins J. B., L. R. Perry, J. L. Beck, and J. D. Taylor. 2021. Increased abundance of the common raven within the ranges of greater and Gunnison sage-grouse: influence of anthropogenic subsidies and fire. *Human-Wildlife Interactions* 15(3):270-288.
- Duerr, A. E., P. H. Bloom, K. Ross, T. A. Miller, M. A. Braham, A. L. Fesnock, and T. E. Katzner. 2021. Influence of anthropogenic subsidies on movements of common ravens. *Human-Wildlife Interactions* 15(3):318-330.
- Ellis, K. S., R. T. Larsen, and D. N. Koons. 2020. Dependence of spatial scale in landscape associations with cause-specific predation of snowy plover nests. *Ecosphere* 11(9):e03257.
- Ellsworth, L. M., D. W. Wroblewski, J. B. Kauffman, and S. A. Reis. 2016. Ecosystem Resilience Is Evident 17 Years after Fire in Wyoming Big Sagebrush Ecosystems. *Ecosphere* 7:e01618.
- Engel, K. A., and L. S. Young. 1992. Movements and habitat use by common ravens from roost sites in southwestern Idaho. *The Journal of Wildlife Management* 56(3):596-602.

- Frey, S. N., R. Curtis, and K. Heaton. 2013. Home Range and Movements of Greater Sage-grouse in its Southern-most Distribution. All Current Publications, paper 1192. <[https://digitalcommons.usu.edu/extension\\_curall/1192](https://digitalcommons.usu.edu/extension_curall/1192)>. Accessed November 12, 2020.
- Gibson, D., E. J. Blomberg, M. T. Atamian, S.P. Espinosa, and J.S. Sedinger. 2018. Effects of power lines on habitat use and demography of Greater Sage-grouse. *Wildlife Monographs* 200:1-41.
- Harju, S. M., P. S. Coates, S. J. Dettenmaier, J. B. Dinkins, P. J. Jackson, and M. P. Chenaille. 2021. Estimating trends of common raven populations in North America, 1966-2018. *Human-Wildlife Interactions* 15(3):248-269.
- Harju, S. M., C. V. Olson, J. E. Hess, and B. Bedrosian. 2018. Common raven movement and space use: influence of anthropogenic subsidies within greater sage-grouse nesting habitat. *Ecosphere* 9(7):e02348.
- Holloran, M. J., and S. H. Anderson. 2003. Direct identification of northern sage-grouse, *Centrocercus urophasianus*, nest predators using remote sensing cameras. *Canadian Field-Naturalist* 117(2):308-310.
- Kristan III, W. B., and W. I. Boarman. 2003. Spatial pattern of risk of common raven predation on desert tortoises. *Ecology* 84:2432–2443.
- Kristan III, W. B., W. I. Boarman, and J. J. Crayon. 2004. Diet composition of common ravens across the urban-wildland interface of the West Mojave Desert. *Wildlife Society Bulletin* 32(1):244-253.

- Lockyer, Z. B., P. S. Coates, M. L. Casazza, S. Espinosa, and D. J. Delehanty. 2013. Greater sage-grouse nest predators in the Virginia Mountains of northwestern Nevada. *Journal of Fish and Wildlife Management* 4:242–254.
- Miller, R. F., T. Svejcar, and J. A. Rose. 1999. Conversion of shrub steppe to juniper woodland. *Proceedings: Ecology and Management of Pinyon-Juniper Communities Within the Interior West*. USDA Forest Service, RMRS-P-9:275-284.
- Miller, R. F., T. J. Svejcar, and J. A. Rose. 2000. Impacts of western juniper on plant community composition and structure. *Journal of Range Management* 53:574–585.
- Miller, R. F., R. J. Tausch, E. D. MacArthur, D. D. Johnson, and S. C. Sanderson. 2008. Age structure and expansion of piñon-juniper woodlands: a regional perspective in the Intermountain West. *Research Paper RMRS-RP-69*. US Department of Agriculture, Forest Service, Rocky Mountain Research Station, Fort Collins, CO, USA 15 p.
- Moynahan, B. J., M. S. Lindberg, J. J. Rotella, and J. W. Thomas. 2007. Factors affecting nest survival of greater sage-grouse in northcentral Montana. *Journal of Wildlife Management* 71(6):1773-1783.
- O’Neil, S. T., P. S. Coates, B. E. Brussee, P. J. Jackson, K. B. Howe, A. M. Moser, L. J. Foster, and D. J. Delehanty. 2018. Broad-scale occurrence of a subsidized avian

predator: Reducing impacts of Ravens on Sage-Grouse and other sensitive prey. *Journal of Applied Ecology* 55(6):2641–2652.

Reinhardt, J. R., S. Filippelli, M. Falkowski, B. Allred, J. D. Maestas, J. C. Carlson, and D. E. Naugle. 2020. Quantifying Pinyon-Juniper Reduction within North America's Sagebrush Ecosystem. *Rangeland Ecology & Management* 73:420-432.

Restani, M. and J. S. Lueck. 2020. The emerging conflict of common ravens roosting on electric power transmission line towers in Montana, USA. *Human-Wildlife Interactions* 14(3):451-460.

Ritchie, M. E., M. L. Wolfe, and R. Danvir. 1994. Predation of artificial Sage Grouse nests in treated and untreated sagebrush. *Great Basin Naturalist* 54(2):122-129.

Roth, C. L., S. T. O'Neil, P. S. Coates, M. A. Ricca, D. A. Pyke, C. L. Aldridge, J. A. Heinrichs, S. P. Espinosa, and D. J. Delehanty. 2022. Targeting Sagebrush (*Artemisia* Spp.) Restoration Following Wildfire with Greater Sage-Grouse (*Centrocercus Urophasianus*) Nest Selection and Survival Models. *Environmental Management* 70:288–306.

Sauer, J. R., D. K. Niven, J. E. Hines, D. J. Ziolkowski, Jr, K. L. Pardieck, J. E. Fallon, and W. A. Link. 2019. The North American Breeding Bird Survey, Results and Analysis 1966 - 2019. Version 2.07.2019 USGS Patuxent Wildlife Research Center, Laurel, MD.

- Schroeder, M. A., C. L. Aldridge, A. D. Apa, J. R. Bohne, C. E. Braun, S. D. Bunnell, J. W. Connelly, P. A. Deibert, S. C. Gardner, M. A. Hilliard, G. D. Kobriger, S. M. McAdam, C. W. McCarthy, J. J. McCarthy, D. L. Mitchell, E. V. Rickerson, and S. J. Stiver. 2004. Distribution of sage-grouse in North America. *Condor* 106:363–376.
- Schroeder, M. A., J. R. Young, and C. E. Braun. 2020. Greater Sage-Grouse (*Centrocercus urophasianus*), version 1.0. In *Birds of the World* (A. F. Poole and F. B. Gill, Editors). Cornell Lab of Ornithology, Ithaca, NY, USA.
- Schupp, E. W., C. S. Boyd, and S. Green. 2015. Woody Fuels Reduction in Wyoming Big Sagebrush Communities. *Great Basin Factsheet Series* (13).
- Sinclair, A. R. E., R. P. Pech, C. R. Dickman, D. Hik. P. Mahon, and A. E. Newsome. 1998. Predicting effects of predation on conservation of endangered prey. *Conservation Biology* 12(3):564-575.
- Peebles, L. W. and M. R. Conover. 2017. Winter ecology and spring dispersal of common ravens in Wyoming. *Western North American Naturalist* 77(3):293-308.
- Porensky, L. M., R. McGee, and D. W. Pellatz. 2020. Long-term grazing removal increased invasion and reduced native plant abundance and diversity in sagebrush grassland. *Global Ecology and Conservation* 24:e01267.
- Pyke, D. A., S. E. Shaff, J. C. Chambers, E. W. Schupp, B. A. Newingham, M. L. Gray, L. M. Ellsworth. 2022. Ten-year ecological responses to fuel treatments within semiarid Wyoming big sagebrush ecosystems. *Ecosphere* 12(7):e4176.



- Pyke, D. A., S. E. Shaff, A. I. Lindgren, E. W. Schupp, P. S. Doescher, J. C. Chambers, J. S. Burnham, and M. M. Huso. 2014. Region-Wide Ecological Responses of Arid Wyoming Big Sagebrush Communities to Fuel Treatments. *Rangeland Ecology & Management* 67:455– 67.
- Utah Division of Wildlife Resources. 2019. Utah Conservation Plan for Greater Sage-grouse. <[https://wildlife.utah.gov/sage-grouse/Utah\\_Greater\\_Sage-grouse\\_Plan.pdf](https://wildlife.utah.gov/sage-grouse/Utah_Greater_Sage-grouse_Plan.pdf)>. Accessed November 10, 2021.
- Watters, M. E., T. L. Mclash, C. L. Aldridge, and R. M. Brigham. 2002. The effect of vegetation structure on predation of artificial Greater Sage-Grouse nests. *Ecoscience* 9(3):314-319.
- Webb, W. C., J. M. Marzluff, and J. Hepinstall-Cymerman. 2011. Linking resource use with demography in a synanthropic population of common ravens. *Biological Conservation* 144(9):2264-2273.

## CHAPTER II

LIVESTOCK PRODUCTION AND AGRICULTURAL PRACTICES IMPACT  
COMMON RAVEN OCCURRENCE WITHIN GREATER SAGE-GROUSE HABITAT  
IN SOUTHERN UTAH, USA

**Abstract**

There are many anthropogenic subsidies within the Intermountain West that promote and sustain common raven (*Corvus corax*; hereafter raven) populations via food, water, and habitat services. These subsidies are important for ravens in arid settings where these resources would otherwise be difficult to obtain. Raven occurrence within sagebrush ecosystems is of particular concern as they are one of the most common nest predators of the sagebrush-obligate species the greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse). To better understand the drivers of raven occurrence within and near sage-grouse habitat in southern Utah, I conducted point-count surveys ( $n = 810$ ) from February through June of 2021 and 2022. Using generalized linear mixed models, I found that the 3 most important factors that influenced raven occurrence were the calendar date of the survey, proximity to agricultural practices, and the presence of livestock. Ravens had higher rates of occurrence during the months of February through April compared to May through June. This indicate that ravens are highly active within sage-grouse habitat prior to both the raven and sage-grouse nesting seasons. Ravens were also highly associated with agricultural land (e.g., cropland, pastures) and livestock. According to my models, when livestock were present the odds of raven occurrence increased by a multiplicative factor of 3.61 (95% CI: 2.33, 5.67).

These findings may help guide land managers because they suggest that sage-grouse populations would benefit from decisions that, whenever it is feasible, minimize livestock presence near lekking and brood-rearing habitats, as this may in turn decrease raven occurrence and sage-grouse nest depredation events. Additionally, lethal removal efforts may be more effective if they were targeted near livestock operations where ravens are often present.

## **Introduction**

Breeding bird surveys indicate that common raven (*Corvus corax*; hereafter raven) populations within the Great Basin, United States, have been increasing by approximately 3% every year since 1966; this is slightly higher than the 2.5% average annual increase recorded across the country (Sauer et al. 2019). Much of this growth in raven populations can be attributed to anthropogenic activities and structures that provide subsidies (nesting habitat, food, and water) for ravens (O'Neil et al. 2018). The most prevalent nesting and roosting subsidies that humans provide are transmission or power lines as they provide ample support and space for large stick nests (Coates et al. 2014, Restani and Lueck 2020). Food subsidies can be found in landfills (Kristan et al. 2004, Peebles and Conover 2017), along paved roads (i.e. roadkill; Kristan et al. 2004, Harju et al. 2018), and throughout agricultural (Engel and Young 1992) and livestock operations (Coates et al. 2016). There is a cause for concern when anthropogenic subsidies support raven populations in areas where they would not normally flourish, such as the semi-arid sagebrush (*Artemisia* spp.) habitats throughout the Great Basin (Coates et al. 2020, Duerr et al. 2021).

Little is known about raven movement and dispersal throughout the western United States. Like most birds, raven movements differ dramatically between their breeding and non-breeding seasons. One study in Wyoming found that during the breeding season, breeding ravens, when off of their nests, traveled an average of 0.8 km (range 0.5-1.3 km) from the nest, while nonbreeding ravens traveled an average of 10.4 km (range 4.1-26.3 km) from the geographic center of their GPS locations (Harju et al. 2018). Breeding ravens traveled short distances from their nests to nearby subsidies such as roads or railroads; nonbreeding ravens were more likely to travel greater distances to point-source subsidies such as landfills. A separate study within the Mojave Desert of southern California tracked raven movements during the nonbreeding season (September to December) and found that ravens traveled on average 41.5 km (range 0.1-206.1 km) each day (Duerr et al. 2021). Duerr et al. (2021) found that, during the nonbreeding season, ravens relied heavily on urban areas with numerous subsidies (both food and habitat), but the ravens would also consistently spend part of the day within less developed areas where there were no anthropogenic subsidies.

Ravens are generalist omnivores and scavengers; they have a wide diet and have been known to exploit novel resources when found (Boarman and Heinrich 1999). As their populations have increased and spread throughout the West (Harju et al. 2021), there has been a focus on how ravens affect several sensitive species of wildlife. Three primary prey species of concern are the desert tortoise (*Gopherus agassizii*; Kristan and Boarman, 2003), snowy plover (*Charadrius nivosus*; Ellis et al. 2020), and the greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse; Dinkins et al. 2012 and 2014, Lockyer et al 2013, Coates et al. 2020, Coates et al. 2021). Generalist omnivores

like ravens pose a significant threat to these sensitive prey species. Because ravens can rely so heavily on human subsidies, their populations may continue to grow even as their natural prey species' numbers begin to dwindle, thus exasperating the negative effect the ravens can have on these populations (Sinclair et al. 1998, Coates et al. 2021, Dinkins et al. 2021).

A review by Conover and Roberts (2017) found that ravens were the most common nest predator of Greater sage-grouse (*Centrocercus urophasianus*), responsible for 35% of depredated nests across 13 different sage-grouse studies. There have also been multiple studies that specifically looked at how the presence of ravens negatively impacts the success rate of nesting sage-grouse (Bui et al. 2010, Gibson et al. 2018, Coates et al. 2020). Each of these studies quantified how raven activity could be attributed to the anthropogenic subsidies discussed above. For instance, Coates et al. (2020) were able to use 10 years of raven surveys in the Great Basin to determine anthropogenic and environmental covariates associated with raven detections; they found that sage-grouse nest survival was negatively impacted when raven density exceeded about  $0.4 \text{ km}^{-2}$ .

While research has been conducted to survey ravens within sage-grouse habitat in Idaho, Nevada, and Wyoming (Bui et al. 2010, Dinkins et al. 2012, Coates et al. 2016, Peebles and Conover 2017, Harju et al. 2018, and O'Neil et al. 2018, Coates et al. 2020), there has been no published reports regarding ravens within sagebrush ecosystems in Utah. In past studies, point-count surveys have been used to estimate raven densities in relation to habitat covariates or in relation to sage-grouse populations (Bui et al. 2010, Dinkins et al. 2012, O'Neil et al. 2018, Coates et al. 2020). Most raven point-count surveys have taken place during the sage-grouse breeding and brood-rearing seasons

(March-August) and occurred at survey points identified via stratified random sampling across known sage-grouse habitat. There have been limited raven survey efforts in Utah, and there is little research into raven habitat use within sagebrush ecosystems during the winter (but see Peebles and Conover 2017). It is important to study raven populations within sage-grouse habitat in Utah because many of the sage-grouse populations are fragmented due to the natural landscape and due to human activities (Schroeder et al. 2004, Utah Division of Wildlife Resources 2019). Also, these sage-grouse populations are on the southern margin of the species' range; studies show that populations on the margins of their range tend to have higher genetic variability amongst individuals and populations (Sexton et al. 2009). It is important to conserve and study these populations that may have important adaptations for hotter, drier climates.

The objective of this study was to conduct point-count surveys for ravens within and outside of sage-grouse management areas (defined as priority sage-grouse habitat by Utah Division of Wildlife Resources 2019; hereafter SGMAs) in southern Utah to analyze what natural and anthropogenic factors may lead to higher rates of raven occurrence. I was also interested in how raven occurrence would change throughout the seasons from winter to summer. I hypothesized that ravens would be positively associated with proven subsidies such as paved roads and agricultural practices. I also hypothesized that ravens would be more likely to be present when livestock were present, even in remote areas away from other human activities.

## **Study Area**

Raven point-count surveys took place within and near the Bald Hills and Panguitch SGMAs within Iron and Beaver counties in southern Utah, USA (Figure 2.1). The surveys were located on private land and within U.S. Bureau of Land Management (BLM) land that is utilized by livestock producers (both for cattle and sheep). Survey points varied in their elevation from 1,532 to 2,570 m (above sea level). The sage-grouse habitat was generally classified as Inter-mountain Basins big sagebrush shrubland that was dominated by various subspecies of big sagebrush (*Artemisia tridentata* spp.), black sagebrush (*Artemisia nova*), rabbitbrush (*Chrysothamnus* sp.) and bitterbrush (*Purshia tridentata*). The prominent grass species present were crested wheatgrass (*Agropyron cristatum*), Indian ricegrass (*Achnatherum hymenoides*), bottlebrush squirreltail (*Elymus elymoides*), bluebunch wheatgrass (*Pseudoroegneria spicata*), and the annual invasive, cheatgrass (*Bromus tectorum*). Several survey points were also within pinyon pine (*Pinus edulis*) and juniper (*Juniperus osteosperma*) forest (hereafter called pinyon-juniper forest). The surveys covered a total area of roughly 1,017 km<sup>2</sup> (Figure 2.2).

## Methods

*Point-count Surveys.* To maximize the number of point-count surveys across the region, I ensured the survey points were all located directly on paved or dirt roads. To select survey locations, I used ArcGIS Pro (Esri 2022) to create a fishnet grid with 4 km spacing and overlaid this onto a spatial layer of the study area. The study area was centered within the Bald Hills and Panguitch SGMAs, however the fishnet grid extended 5km buffer outside the boundaries of the SGMAs. I decided to include survey points that were outside of the sagebrush communities because ravens are able to fly from nontarget habitats into sage-grouse habitats; Harju et al. (2018) found that the nonbreeding ravens

in their study flew on average 10.4 km from the center of their geographic activities. Using the attribute table of the fishnet grid, I then randomly selected 48 points inside of this 5 km buffer around the SGMAs. Finally, I selected a location on the nearest road to that randomly generated point. Barring poor driving conditions, I visited the survey points every 2 weeks between February and June of 2021 and 2022.

In 2021, I had the help of 4 trained biological technicians who were each assigned to a subset of the survey points ( $n = 48$  survey points). In 2022, I conducted all of the surveys myself ( $n = 43$  survey points; 5 were inaccessible due to driving conditions and time restraints). Each survey was 5 minutes long; Bonthoux and Balent (2012) found that 5-minute surveys tend to be an sufficient amount of time to detect birds, especially if they are numerous and inconspicuous. The majority (86% of all surveys) were conducted within 6 hours after sunrise; the remaining surveys were conducted between 6 to 10 hours after sunrise. Each survey consisted of the surveyor driving to the designated location and recording the time, weather conditions (e.g., temperature, wind, cloud-cover), presence of a temporary water source (within 1 km) (e.g., an active agricultural sprinkler), and presence of any livestock (within 1 km) (both cattle and sheep). The surveyor then exited their vehicle and, for 5 minutes, scanned the area with and without binoculars, counting any ravens detected either visually or audibly within 1 km; most of the surveys took place in very open, flat habitat that enabled the surveyor to see this far. The surveyor also recorded the cardinal direction and distance of each bird or group of birds using a compass and a range finder.

*Data Analysis.* Using ArcGIS Pro (Esri 2022), I mapped possible anthropogenic subsidies so that I could obtain distances from each survey point to paved roads, producer



feeding lots, agricultural operations (irrigated croplands and pastures), and water related features (such as cattle ponds, wildlife guzzlers, livestock catchments, etc.). While I obtained mapped data of the local paved roads and water-related features from the local BLM office, I used a combination of aerial imagery, my familiarity with the area, and LANDFIRE Existing Vegetation Type layer to create mapped polygons where the livestock feeding lots and farming practices were located (LANDFIRE 2016, U.S. Geological Survey, 2021). I had hoped to include distribution lines in these analyses, however there was no mapped data available for this feature. I log-transformed all of the variables that measured distance and I used a square root transformation on the 'hours since sunrise' variable to meet parametric assumptions for the mixed models. Before building my models, I tested for collinearity amongst my transformed variables using the R package 'corrplot' with the intention of removing any variables if they were found to have an absolute correlation coefficient of  $>0.7$  (Dormann et al. 2013, Wei and Simko 2021).

To analyze my survey data, I built generalized linear mixed models (GLMMs) using the R package 'lme4' (Bates et al. 2015). The response variable for these models was the number of ravens observed per individual survey. The random effect for all of the models was a combined factor of year (2021 or 2022) and survey point number. The fixed effects varied by model and included 10 different variables: 'livestock presence', 'ordinal date', 'sqrt(hours since sunrise)', 'temperature', 'log(distance to agriculture)', 'log(distance to paved road)', 'log(distance to producer feeding lot)', 'log(distance to cliff)', 'log(distance to water feature)', and 'log(distance to forest)'. All of the models assumed a negative binomial distribution using a log link function; I first tried to use a

poisson distribution but I ran into issues of zero-inflation with my raven count data. To compare nested models, I used likelihood ratio tests (via ANOVA). When a likelihood ratio test resulted in a P-value of  $< 0.05$ , I assumed that the 2 models were significantly different from one another and that the additional fixed effects were contributing a significant amount of data to the model (Fox et al. 2015). In this way, I was able to determine the “best” performing model that had the fewest predictors as fixed effects; this was important to avoid over-fitting the model. I then used the ‘DHARMA’ package to test my model for over/underdispersion (Hartig 2022). I used ‘influence.ME’ in order to analyze my model for potential outliers (Nieuwenhuis 2012).

I developed a separate set of GLMMs (family = binomial; link = logit) where the response variable was whether or not ravens were present at a given survey; with these models, I estimated the influence of each fixed effect variable on the odds of raven occurrence. I used the same step-wise process as above to build, compare, and test the assumptions of the models.

## **Results**

In 2021, I visited each of the 48 survey points a total of 4 to 11 times; in 2022, I visited 43 survey points 6 to 11 times each. The variation in the number of surveys at each point was due to accessibility issues because of winter weather. Between the 2 years, the average survey took place 3.7 hours after sunrise ( $q_{25} = 2.07$ ,  $q_{50} = 3.42$ ,  $q_{75} = 5.13$ ). Over the 2 years, I conducted a total of 810 surveys across the study area; I detected ravens in approximately 45% of these surveys. When I detected ravens, the average number of ravens counted within a single survey was 3.6 (range 1-107 ravens).

Raven occurrence varied significantly across the seasons and between the 2 years (Figures 2.3 and 2.4); for both years, raven occurrence decreased by approximately 16% between the end of winter (February-April;  $\bar{x} = 0.52$ ) and the start of summer (May-June;  $\bar{x} = 0.36$ ).

Based on my tests for collinearity, no continuous variables needed to be removed from the modeling process. By systematically removing fixed effect variables one at a time while running individual nested ANOVAs, I determined the model that contained the most significant fixed effects. The null model was not significant compared to models that included the fixed effects. Only 3 of the 10 variables were found to be significant in the model, including ‘ordinal date’, ‘livestock presence’ (present or absent), and the ‘log(distance to agriculture)’ (Table 2.1). The best fit model had a significant interaction term ( $P = 0.04$ ) between ‘livestock presence’ and ‘log(distance to agriculture)’. Including this interaction makes ecological sense as livestock presence in the study area was often associated with pastures and agriculture. While evaluating the model’s assumptions, I did not find any significant dispersion issues (Appendix A: Figure A1; Hartig 2022). However, 4 outliers were found within the random effects groups (Nieuwenhuis et al. 2012). I fitted the model again with these 4 groups removed and compared the new model with the ‘complete’ model (Appendix A: Table A1); given that the coefficient estimates and P-values did not change in a manner that would alter the model’s interpretation, I decided to leave the outliers in the dataset.

After building and comparing the GLMMs (family = binomial, link = logit) with the binary response variable of raven occurrence, I found that the best performing model had the same 3 fixed effect variables as the initial model (‘ordinal date’, ‘livestock presence’,

and 'log(distance to agriculture)')(Table 2.2). This model suggests that for every unit increase in the 'log(distance to agriculture)', the odds of raven occurrence decreases by a multiplicative factor of 0.64 (95% CI: 0.52, 0.78). Also, when livestock is present, the odds of raven occurrence increases by a multiplicative factor of 3.61 (95% CI: 2.33, 5.67). No interaction term was significant for this model. I found no significant dispersion issues with this model, and I found 1 outlier that did not alter the model's interpretation when removed.

After livestock presence was determined to be significant within the model, I ran a chi-square test on raven occurrence vs livestock presence. I found that raven occurrence was higher than when livestock was present compared to when livestock was absent ( $\chi^2 = 46.2$ ,  $df = 1$ ,  $P\text{-value} < 0.001$ ).

## **Discussion**

Webb et al. (2021) completed a synthesis of all of the available literature on the common raven within the context of conservation applications related to raven ecology (occurrence, resource use, and demography). In doing so, they established which topics they deemed as understudied; based on their review, the effects of livestock, season, and time of day are all topics that would benefit from further research. They also broke down the research geographically, and there seems to be a lack of raven studies within central and southern Utah. My survey data adds important findings to this body of literature and begins to fill in the gaps listed above. While time of day did not end up being a significant predictor for raven occurrence in my study, livestock presence and season (or ordinal date) were both significant.

This study is also unique in that I fitted the models with 2 different response variables: number of ravens present and raven occurrence (presence/absence). Within the literature, raven occurrence is the more typical response variable of interest; however, I was also interested in how the predictor variables effected the total number of ravens present at a given survey. These 2 models ended up looking very similar, but interestingly, the raven occurrence model did not have an interaction term of livestock and distance to agriculture. This means that, independent of how far the survey point was to agriculture, the presence of livestock had a significant effect on the presence of ravens, and independent of whether or not livestock was present, the closer the point was to agriculture, the more likely it was that ravens would be present. However, due to the interaction within the model, when looking at the number of ravens observed, it is important to not consider these 2 predictors independent from one another.

My data indicates that raven occurrence was higher in the months of February-April compared to May-June. This finding differs from that of Boarman et al.'s (2006) study within the Mojave Desert, where they saw a significant decrease in their raven counts during the February - April months. Ravens in the West typically nest between March and July (Kristan and Boarman 2003, Harju et al. 2018). Given that nonbreeding ravens have been found to travel greater distances compared to breeding ravens (Harju et al. 2018), it is possible that the drop in raven occurrence throughout the season was because breeding ravens began to travel less after they selected their nesting habitat. In the nonbreeding season, local ravens tend to roost in massive groups that disperse into the sagebrush ecosystem at sunrise; during the breeding season, there are fewer instances of these large groups of ravens moving across the landscape, as raven pairs disperse and

claim nesting habitats (Moffett personal obs., 2022). It is possible that many nonbreeding ravens detected within these surveys were simply moving through the area and were only opportunistically utilizing that habitat (Stewart et al. 2018).

My surveys exhibited a positive relationship between livestock presence (both sheep and cattle) and raven occurrence. This finding supports previous research with similar trends within Idaho and across the Great Basin (Coates et al. 2016, O'Neil et al. 2018). There are several likely reasons why ravens select habitat near livestock. First, there are numerous accounts of ravens preying on newborn calves and lambs, at the dismay of livestock producers (Peebles and Spencer 2020). Livestock production also often provides food and water subsidies to ravens via feed, troughs, or cattle ponds. Lastly, livestock moving across the landscape may flush small mammals or birds that ravens may opportunistically depredate.

The last significant predictor variable in my raven occurrence model was the distance of the survey point to the nearest agriculture or cropland. This model supports previous research that has also found that ravens are more likely to occur close to agricultural practices (Coates et al. 2020, Duerr et al. 2021). More detailed data could be collected to analyze this pattern more closely. There are different agriculture-related land uses within this study area (i.e., livestock corrals and feeding lots, croplands, rangeland grazing livestock), and it would be interesting to distinguish between them as it relates to raven occurrence.

In addition to more detailed information about the various local agricultural practices, there are several pieces of data that would have been beneficial for my analyses. For

instance, given how much ravens rely on powerlines for roosting and nesting (Coates et al. 2014, Restani Lueck 2020), it would have been useful to have access to map data of the local distribution lines within my study area. It also would have been very interesting to have been able to find all active raven nests within the study area and to have identified breeding vs nonbreeding ravens during the surveys. This information would have been valuable in the context of my raven occurrence analyses because when ravens nest within the sagebrush-steppe, they primarily forage close to their nest (rather than fly farther to point-source subsidies) where they may find and depredate sage-grouse nests and chicks (Harju et al. 2018). Further research is necessary to better understand the resource use and spatial ecology of ravens within sage-grouse habitat.

### **Management Implications**

One of the primary conservation concerns with ravens in southern Utah is their impact on sage-grouse populations via nest depredation. This is of particular concern for the populations within southern Utah which are smaller and more spatially fragmented than sage-grouse populations to the north (Frey et al. 2013, Beers 2023). Land managers have been aware of this issue for some time now, and it continues to be necessary to study the patterns and behaviors of these corvids. For instance, this occurrence data supports the growing body of evidence that livestock activities attract ravens into landscapes that they may not have utilized otherwise. One potential management strategy would be minimizing livestock use near large, active leks from March to July (the lekking season through the brood-rearing season), due to the fact that sage-grouse tend to nest close to leks (Dahlgren et al. 2016). Another strategy would be limiting the placement of temporary water sources for livestock in sage-grouse nesting and brood-

rearing habitats. However, this strategy would be difficult to implement because much of the rangeland in southern Utah is managed for both livestock grazing and sage-grouse habitat. An alternative strategy is the lethal removal of ravens via poisoned bait; this has been found to have positive effects on sage-grouse nest success (Dinkins et al. 2016). It is agreed that lethal removal is a short-term solution that is spatially limited with its effects, however it can be a useful tool when raven densities reach a threshold whereby they will do severe damage to a prey population (Delehanty 2021). More research is needed to better understand the optimal placement and timing of poisoned bait so that it has the strongest impact on the ravens that utilize sage-grouse habitat. Long-term conservation goals may be better served with management strategies focused on removing as many point-source subsidies from the landscape as possible (e.g., roadkill, cattle carcasses, garbage, unnecessary standing structures).

Based on my raven count data, the effect of agriculture on the number of ravens present did not decrease until the agriculture was about 5 km away from a survey point (Figure 2.5). Given this finding along with previous research on how far breeding and nonbreeding ravens will travel within a day (Harju et al. 2018, Duerr et al. 2021), land managers focused on conserving species such as sage-grouse may want to consider the impacts that agricultural and livestock operations may have on nearby habitats. For example, sage-grouse habitat improvement projects may have greater success when implemented far away from these operations as ravens (especially breeding ravens) will be less likely to congregate in large numbers.

### **Literature Cited**



- Bates, D., M. Maechler, B. Bolker, and S. Walker. 2015. Fitting Linear Mixed-Effects Models Using lme4. *Journal of Statistical Software* 67(1):1–48.
- Beers, A. T. 2023. Local and Regional Landscape Characteristics Driving Habitat Selection by Greater Sage-Grouse Along a Fragmented Range Margin. All Graduate Theses and Dissertations. 8695. <<https://doi.org/10.26076/4f07-36a8>>. Accessed March 6, 2023.
- Boarman, W. I., and B. Heinrich. 1999. Common Raven. In A. Poole and F. Gill, editors. *The birds of North America*, Number 476. The Birds of North America, Philadelphia, Pennsylvania, USA.
- Boarman, W.I., M. A. Patten, R. J. Camp, and S. J. Collis. 2006. Ecology of a population of subsidized predators: Common ravens in the Mojave Desert, California. *Journal of Arid Environments* 67:248–261.
- Bonthoux, S., and G. Balent. 2012. Point count duration: five minutes are usually sufficient to model the distribution of bird species and to study the structure of communities for a French landscape. *Journal of Ornithology* 153:491–504.
- Bui, T. V. D., J. M. Marzluff, and B. Bedrosian. 2010. Common raven activity in relation to land use in western Wyoming: Implications for greater sage-grouse reproductive success. *Condor* 112:65–78.
- Coates, P. S., B. E. Brussee, K. B. Howe, K. B. Gustafson, M. L. Casazza, and D. J. Delehanty. 2016. Landscape characteristics and livestock presence influence

common ravens: relevance to greater sage-grouse conservation. *Ecosphere* 7(2):e01203.

Coates, P. S., K. B. Howe, M. L. Casazza, and D. J. Delehanty. 2014. Common raven occurrence in relation to energy transmission line corridors transiting human-altered sagebrush steppe. *Journal of Arid Environments* 111(2014):68–78.

Coates, P. S., S. T. O’Neil, B. E. Brussee, M. A. Ricca, P. J. Jackson, J. B. Dinkins, K. B. Howe, A. M. Moser, L. J. Foster, and D. J. Delehanty. 2020. Broad-scale impacts of an invasive native predator on a sensitive native prey species within the shifting avian community of the North American Great Basin. *Biological Conservation* 243(2020):108409.

Coates, P. S., W. C. Webb, S. J. Dettenmaier, S. M. Harju, and D. J. Delehanty. 2021. Synthesis of nest predation impacts of common ravens on sensitive avian species. *Human-Wildlife Interactions* 15(3):350–373.

Conover, M. R., and A. J. Roberts. 2017. Predators, predator removal, and sage-grouse. *Journal of Wildlife Management* 81(1):7–15.

Delehanty, D. J. 2021. Opinion: Raven control from a conservation biology perspective. *Human-wildlife Interactions* 15(3):534–543.

Dinkins, J. B., M. R. Conover, C. P. Kirol, and J. L. Beck. 2012. Greater sage-grouse (*Centrocercus urophasianus*) select nest sites and brood sites away from avian predators. *Auk* 129:600–610.

- Dinkins, J. B., M. R. Conover, C. P. Kirol, J. L. Beck, and S. N. Frey. 2014. Greater Sage-Grouse (*Centrocercus urophasianus*) select habitat based on avian predators, landscape composition, and anthropogenic features. *The Condor* 116:629–642.
- Dinkins, J. B., M. R. Conover, C. P. Kirol, J. L. Beck., and S. N. Frey. 2016. Effects of common raven and coyote removal and temporal variation ins climate on greater sage-grouse nesting success. *Biological Conservation* 202:50–58.
- Dinkins J. B., L. R. Perry, J. L. Beck, and J. D. Taylor. 2021. Increased abundance of the common raven within the ranges of greater and Gunnison sage-grouse: influence of anthropogenic subsidies and fire. *Human-Wildlife Interactions* 15(3):270–288.
- Dormann, C. F., J. Elith, S. Bacher, C. Buchmann, G. Carl, G. Carre, J. R. Garcia Marquez, B. Gruber, B. Lafourcade, P. J. Leitaó, T. Munkemuller, C. McClean, P. E. Osborne, B. Reineking, B. Schroder, A. K. Skidmore, D. Zurell, and S. Lautenbach. 2013. Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. *Ecography* 36:27–46.
- Duerr, A. E., P. H. Bloom, K. Ross, T. A. Miller, M. A. Braham, A. L. Fesnock, and T. E. Katzner. 2021. Influence of anthropogenic subsidies on movements of common ravens. *Human-Wildlife Interactions* 15(3):318–330.
- Ellis, K. S., R. T. Larsen, and D. N. Koons. 2020. Dependence of spatial scale in landscape associations with cause-specific predation of snowy plover nests. *Ecosphere* 11(9):e03257.

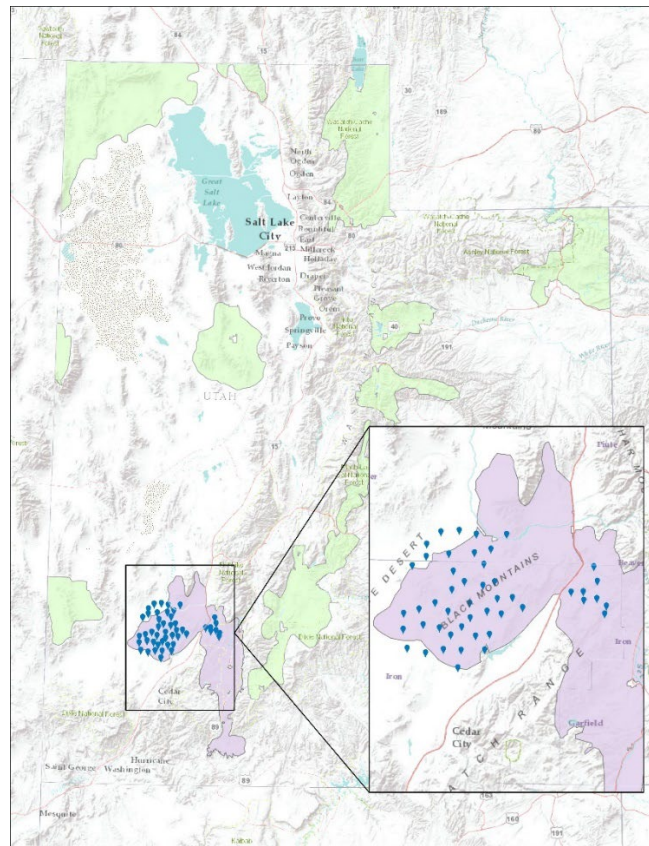
- Engel, K. A., and L. S. Young. 1992. Movements and habitat use by common ravens from roost sites in southwestern Idaho. *Journal of Wildlife Management* 56(3):596–602.
- Esri. 2022. ArcGIS Pro: Version 3.0. Environmental Systems Research Institute, Redlands, CA.
- Fox, G. A., S. Negrete-Yankelevich, and V. J. Sosa. 2015. *Ecological statistics: contemporary theory and application*. Oxford University Press, Oxford, United Kingdom.
- Gibson, D., E. J. Blomberg, M. T. Atamian, S.P. Espinosa, and J.S. Sedinger. 2018. Effects of power lines on habitat use and demography of greater sage-grouse. *Wildlife Monographs* 200:1–41.
- Harju, S. M., P. S. Coates, S. J. Dettenmaier, J. B. Dinkins, P. J. Jackson, and M. P. Chenaille. 2021. Estimating trends of common raven populations in North America, 1966-2018. *Human-Wildlife Interactions* 15(3):248–269.
- Harju, S. M., C. V. Olson, J. E. Hess, and B. Bedrosian. 2018. Common raven movement and space use: influence of anthropogenic subsidies within greater sage-grouse nesting habitat. *Ecosphere* 9(7):e02348.
- Hartig, F. 2022. DHARMA: Residual diagnostics for hierarchical (multi-level/mixed) regression models. R package version 0.4.5.

- Kristan III, W. B., and W. I. Boarman. 2003. Spatial pattern of risk of common raven predation on desert tortoises. *Ecology* 84:2432–2443.
- Kristan III, W. B., W. I. Boarman, and J. J. Crayon. 2004. Diet composition of common ravens across the urban-wildland interface of the West Mojave Desert. *Wildlife Society Bulletin* 32(1):244–253.
- LANDFIRE. 2016. Existing Vegetation Type Layer, LANDFIRE 2.0.0. U.S. Department of the Interior, Geological Survey, and U.S. Department of Agriculture. <<http://www.landfire/viewer>> Accessed April 14, 2021.
- Lockyer, Z. B., P. S. Coates, M. L. Casazza, S. Espinosa, and D. J. Delehanty. 2013. Greater sage-grouse nest predators in the Virginia Mountains of northwestern Nevada. *Journal of Fish and Wildlife Management* 4:242–254.
- Nieuwenhuis, R., T. G. Manfred, and B. Pelzer. 2012. influence.ME: tools for detecting influential data in mixed effects models. *R Journal* 4(2):38–47.
- O’Neil, S. T., P. S. Coates, B. E. Brussee, P. J. Jackson, K. B. Howe, A. M. Moser, L. J. Foster, and D. J. Delehanty. 2018. Broad-scale occurrence of a subsidized avian predator: Reducing impacts of ravens on sage-grouse and other sensitive prey. *Journal of Applied Ecology* 55(6):2641–2652.
- Peebles, L. W., and M. R. Conover. 2017. Winter ecology and spring dispersal of common ravens in Wyoming. *Western North American Naturalist* 77(3):293–308.

- Peebles, L. W., and J. O. Spencer Jr. 2020. Common ravens. Wildlife Damage Management Technical Series; USDA-APHIS Wildlife Services.  
<<https://digitalcommons.unl.edu/nwrewdmts/24>>. Accessed March 14, 2023.
- Restani, M., and J. S. Lueck. 2020. The emerging conflict of common ravens roosting on electric power transmission line towers in Montana, USA. *Human-Wildlife Interactions* 14(3):451–460.
- Sauer, J. R., D. K. Niven, J. E. Hines, D. J. Ziolkowski, Jr, K. L. Pardieck, J. E. Fallon, and W. A. Link. 2019. The North American Breeding Bird Survey, results and analysis 1966 - 2019. Version 2.07.2019 USGS Patuxent Wildlife Research Center, Laurel, MD, USA.
- Schroeder, M. A., C. L. Aldridge, A. D. Apa, J. R. Bohne, C. E. Braun, S. D. Bunnell, J. W. Connelly, P. A. Deibert, S. C. Gardner, M. A. Hilliard, G. D. Kobriger, S. M. McAdam, C. W. McCarthy, J. J. McCarthy, D. L. Mitchell, E. V. Rickerson, and S. J. Stiver. 2004. Distribution of sage-grouse in North America. *Condor* 106:363–376.
- Sexton, J. P., P. J. McIntyre, A. L. Angert, and K. J. Rice. 2009. Evolution and ecology of species range limits. *Annual Review of Ecology, Evolution, and Systematics* 40:415–436.
- Sinclair, A. R. E., R. P. Pech, C. R. Dickman, D. Hik. P. Mahon, and A. E. Newsome. 1998. Predicting effects of predation on conservation of endangered prey. *Conservation Biology* 12(3):564–575.

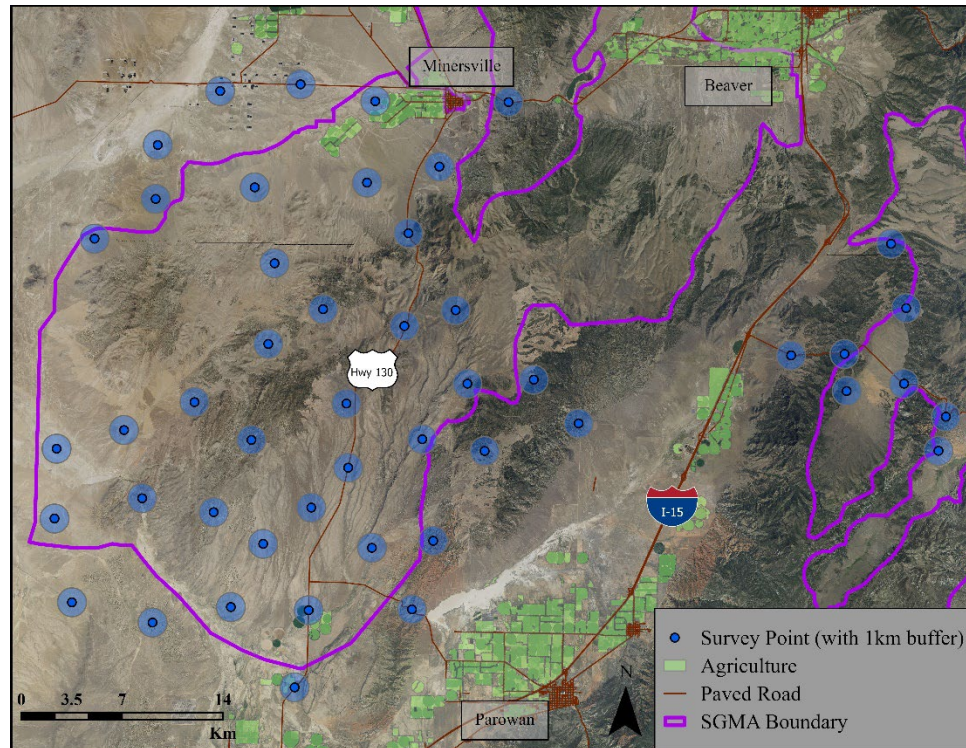
- Stewart, F. E. C., J. T. Fisher, A. C. Burton, and J. P. Volpe. 2018. Species occurrence data reflect the magnitude of animal movements better than the proximity of animal space use. *Ecosphere* 9(2): e02112.
- U.S. Geological Survey. 2021. National Agricultural Imagery Program, 60cm – Resolution NAIP imagery, Iron and Beaver Counties, UT Dataset. <<https://gis.utah.gov/data/aerial-photography/>>. Accessed January 10, 2023.
- Utah Division of Wildlife Resources. 2019. Utah Conservation Plan for Greater Sage-grouse. <[https://wildlife.utah.gov/sage-grouse/Utah\\_Greater\\_Sage-grouse\\_Plan.pdf](https://wildlife.utah.gov/sage-grouse/Utah_Greater_Sage-grouse_Plan.pdf)>. Accessed November 10, 2021.
- Webb W. C., P. S. Coates, S. J. Dettenmaier, and D. J. Delehanty. 2021. Occurrence, resource use, and demography of the common raven in North America: a research synthesis. *Human-Wildlife Interactions* 15(3):304–317.
- Wei, T. and V. Simko. 2021. R package 'corrplot': Visualization of a Correlation Matrix: Version 0.92.

## Figures

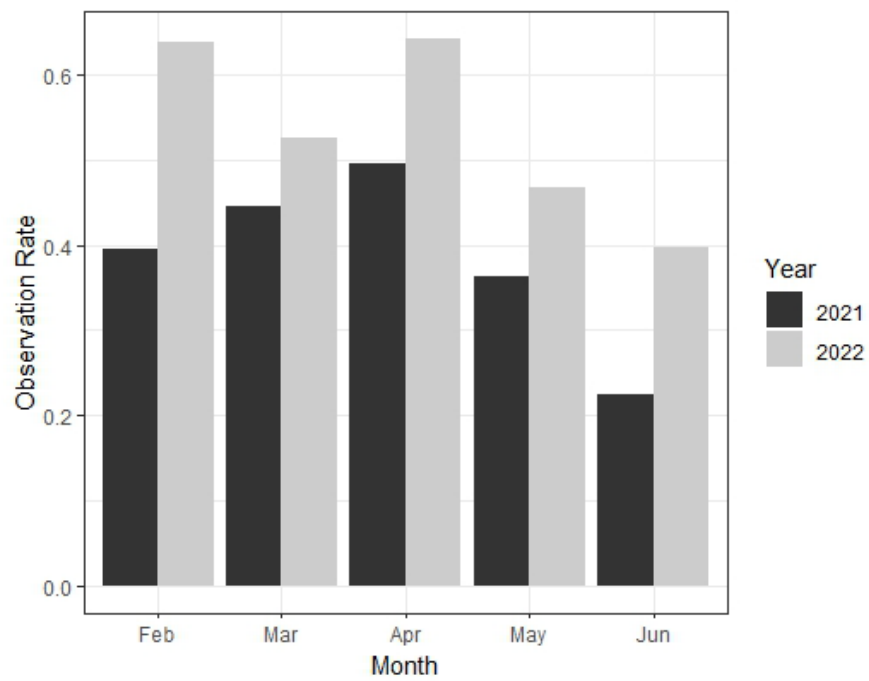


**Figure 2.1.** Map of Sage-grouse Management Areas (SGMAs) in the state of Utah, USA. 49 raven survey points are symbolized with blue markers within the Bald Hills and Panguitch SGMAs. Surveys were conducted February–June 2021–2022.

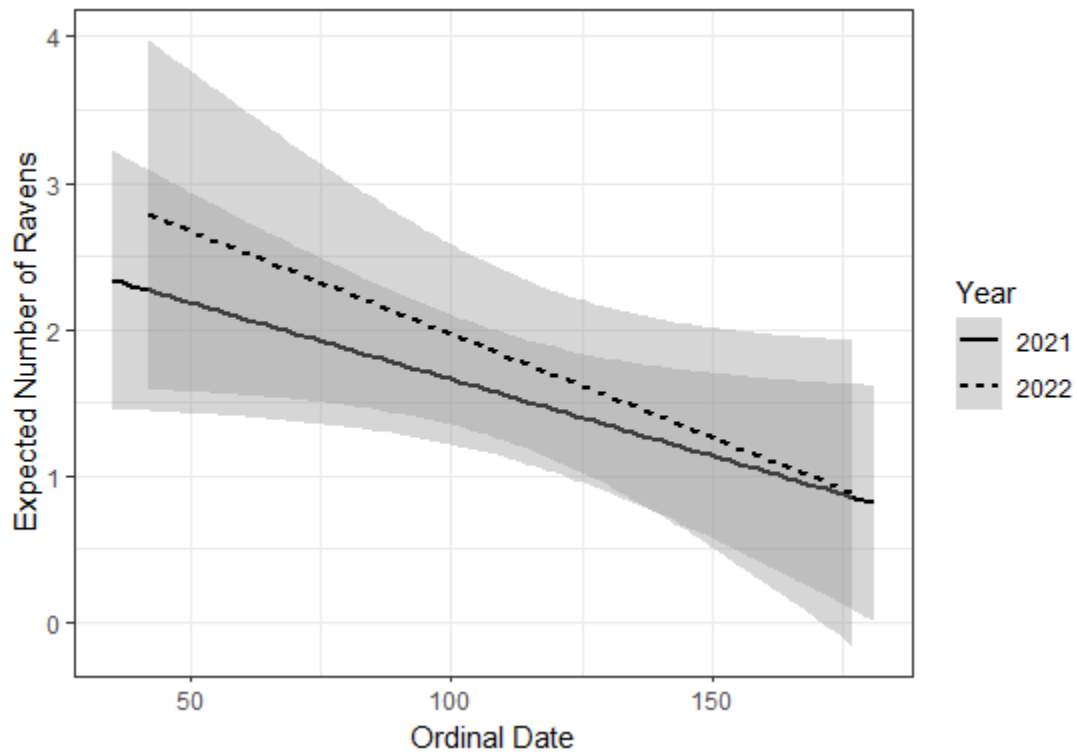




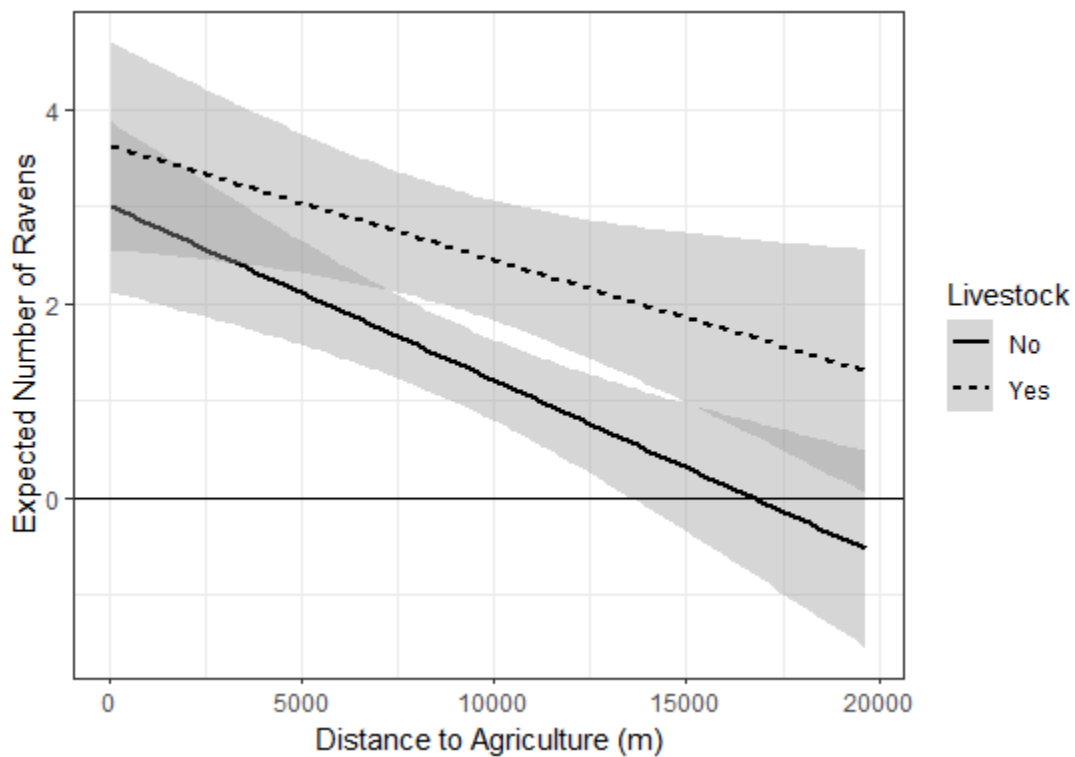
**Figure 2.2.** Map of survey points with 1 km buffer around each point. Nearby agricultural practices (feeding lots, croplands, etc.) are shaded in green. The purple boundary shows the Bald Hills and Panguitch SGMA located in Iron and Beaver counties, Utah, USA. Surveys were conducted February–June 2021–2022.



**Figure 2.3.** Rate of raven detection (number of times ravens were detected / total surveys) across all surveys from February–June 2021–2022. Raven point-count surveys took place in Iron and Beaver counties, Utah, USA.



**Figure 2.4.** Expected number of ravens observed at a given survey point according to ordinal date (February-June) (with 95% confidence interval). The expected counts were derived via a linear model using raven point-count survey data gathered in Iron and Beaver counties, Utah, USA; February–June 2021–2022.



**Figure 2.5** Expected number of ravens observed at a given survey point (with 95% confidence interval) according to the survey point's distance to agricultural practices (e.g., cropland) and according to livestock presence. A reference horizontal line was added at 0 expected ravens. The expected counts were derived using a linear model using raven point-count survey data gathered in Iron and Beaver counties, Utah, USA; February–June 2021–2022.

## Tables

**Table 2.1.** Parameter estimates of fixed effects from fitted generalized linear mixed model (family = negative binomial, link = logit) using data from 810 point-count surveys of the common raven (*Corvus corax*), Iron and Beaver counties, Utah, USA; February–June 2021–2022.

Model				
Number of ravens ~ (1  survey point AND year) + livestock present + log(Distance to ag) + livestock present * log(distance to ag) + ordinal date				
Fixed Effects	Description	$\beta$	SE	P-value
Livestock present	No livestock present during survey vs livestock present (within 1 km)	-1.41	1.02	0.17
Log(distance to ag)	Log(distance to the nearest agriculture)	-0.48	0.08	<0.001
Livestock present * Log(distance to ag)	Interaction term of livestock presence and distance to agriculture	0.25	0.119	<0.05
Ordinal date	Date according to Julian calendar of the given year	-0.01	0.002	<0.001

**Table 2.2.** Parameter estimates of fixed effects from fitted generalized linear mixed model (family = binomial, link = logit) where the binary response variable was raven occurrence (present or absent); the data was from 810 point-count surveys of the common raven (*Corvus corax*), Iron and Beaver counties, Utah, USA; February–June 2021–2022.

Model					
Raven presence ~ (1  survey point AND year) + livestock present + log(distance to ag) + ordinal date					
Fixed Effects	Description	$\beta$	SE	Exp( $\beta$ ) (95% CI)	P-value
Livestock present	No livestock present during survey vs livestock present (within 1 km)	1.28	0.23	3.61 (2.33, 5.67)	<0.001
Log(distance to ag)	Log(distance to the nearest agriculture)	-0.44	0.10	0.64 (0.52, 0.78)	<0.001
Ordinal date	Date according to Julian calendar of the given year	-0.01	0.002	0.99 (0.99, 1.0)	<0.001

## CHAPTER III

ARTIFICIAL NEST DEPREDATION BY THE COMMON RAVEN WITHIN  
VARIOUS SAGEBRUSH TREATMENTS IN SOUTHERN UTAH, USA**Abstract**

With increases in rural and urban development throughout the American West, common raven (*Corvus corax*; hereafter raven) populations have been rising as anthropogenic subsidies provide both food and nesting habitat. As generalist scavengers and predators, ravens have been found to pose a threat to several threatened or sensitive species, including the greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse). The purpose of this research was to quantify the overall threat that ravens may pose as nest predators to sage-grouse within sagebrush habitat that has undergone augmentation or restoration treatments within the Bald Hills and Panguitch Sage-grouse Management Areas (SGMAs) in southern Utah. I conducted an artificial nest experiment over the course of 2 nesting seasons in 2021 and 2022. I constructed artificial nests using chicken eggs to identify nest depredation events in sagebrush habitat where either pinyon pine and juniper (hereafter pinyon-juniper) has been removed or where sagebrush has been thinned (via bullhog, hand-thin, or harrow). In 2 separate study sites, I placed artificial nests underneath sagebrush in treated or intact habitat ( $n = 115$ ) with a trail camera to monitor for nest predators. With the binary response variable of nest outcome (intact vs depredated), I used logistic regressions to fit generalized linear models for the 2 separate study sites. In the Bald Hills site, I found that the nests placed within habitat that had been treated for pinyon-juniper removal were more likely to be depredated than

nearby intact sagebrush. In the Buckskin Valley site, I found that ravens were more likely to depredate nests placed in close proximity to a nest that the ravens had also found. These findings may help to guide land managers expectations of sage-grouse nest survival within sagebrush habitat that has undergone sagebrush vegetation treatments.

## **Introduction**

Before European settlement of North America, sage-grouse occupied a range that was almost double what it is today (Schroeder et al. 2004). This decrease in the distribution (and in the overall population) of sage-grouse has in large part been attributed to habitat loss and fragmentation (Connelly and Braun 1997, Schroeder et al. 2004). Much of the habitat loss is due to human expansion into sagebrush for agricultural uses (Connelly and Braun 1997). As a sagebrush obligate, sage-grouse are unique in that they require different types of sagebrush seasonally and within different life stages. Their life history requires specific habitat-types for the winter (non-breeding), lekking, nesting, early brood-rearing, and late brood-rearing seasons (Dahlgren et al. 2016, and 2019). During the nesting season, hens require moderately dense sagebrush with a mixture of forbs and grasses that provide vertical concealment for the hen and her nest (Dahlgren et al. 2016). While mammalian predators rely on olfactory cues for depredating nests, avian predators, such as ravens, likely depend on their ability to see the movement of the hen on or near the nest (Coates and Delehanty 2008). Therefore, when sagebrush habitat becomes overly dense, resulting in decreased forb and grass cover, or when cheatgrass (*Bromus tectorum*) dominates the native forbs and grasses, sage-grouse hens and their nests have significantly less concealment (Connelly et al. 2000).



During the last century there has been a dramatic increase in both pinyon pine (*Pinus edulis*, *Pinus monophylla*) and juniper (*Juniperus* spp.; pinyon-juniper forests) within the Intermountain West. Much of this expansion of pinyon-juniper forest has been into sagebrush ecosystems where the increase in canopy cover has led to a decrease in shrubs, forbs, and grasses (Miller et al. 2000). Sage-grouse avoid stands of pinyon-juniper adjacent to sagebrush habitat and that the sage-grouse that avoid these trees have a higher rate of survival (Coates et al. 2017). As land managers have been working to increase and improve sage-grouse habitat, there have been massive efforts to remove stands of pinyon-juniper to restore sagebrush communities. One analysis found that within 6 years (2011-2013 and 2015-2017), 1.6% of pinyon-juniper habitat had been reduced (either from wildfire or by management) across the Great Basin and into the Columbia and Colorado plateaus (Reinhardt et al. 2020). Reinhardt et al. (2020) estimate this rate of reduction is just about matching the rate of expansion of pinyon-juniper throughout the range. During the 6 years that were analyzed, about 691 km<sup>2</sup> of pinyon-juniper were reduced within the state of Utah, and most of that management took place within areas designated as Priority Areas for Conservation (PACs) for sage-grouse (Reinhardt et al. 2020).

There are several treatment options for reducing or removing pinyon-juniper stands within sagebrush ecosystems. The choice of treatments can depend on management goals including what successional phase of encroachment the habitat is in on a scale of 1 to 4 (early, mid, late, and closed), which is based on tree density and the amount of sagebrush understory (Miller et al. 1999). Managers in Utah have used the following treatments to remove pinyon-juniper: mastication (e.g., bull hog), hand-

thinning (e.g. lop and scatter), chaining, and fire (Cook et al. 2017, Utah Division of Wildlife Resources 2019, Reinhardt et al. 2020). Multiple studies have found that sage-grouse will use the mixed sagebrush/forb/grass habitat as it grows back post pinyon-juniper removal (Frey et al. 2013, Cook et al. 2017). However, little is known about sage-grouse nest and brood survival rates within habitats that have been treated for pinyon-juniper removal.

In addition to pinyon-juniper removal, sagebrush habitats can sometimes require restoration via sagebrush thinning or removal. This form of restoration becomes necessary as stands of sagebrush become overly old and dense with consequential losses of native perennial grasses and forbs; this is an issue for both livestock managers who need more vegetation for their cattle and for wildlife biologists who want more forbs for breeding sage-grouse (Connelly et al. 2004). These dense stands of sagebrush can also lead to higher intensity wildfires due to high connectivity across large sections of habitat (Schupp et al. 2015). Whether it is for livestock foraging, wildlife conservation, or for woody fuels reduction, land managers may decide to use one or more of the following methods to reduce sagebrush cover: prescribed fires, mechanical methods, herbicide applications, or livestock management (Connelly et al. 2004).

While many studies have investigated the vegetation structure of sagebrush communities following these treatments (e.g., Davies et al. 2012, Porensky et al. 2020, Pyke et al. 2022, Roth et al. 2022), there has been limited research on how these differing sagebrush treatments impact sage-grouse populations throughout the year. One such study took place on Parker Mountain in Utah where it was found that sage-grouse broods preferentially utilized habitat that had been treated with Tebuthiuron, an herbicide,

compared to control plots (Dahlgren et al. 2006). One of the main conclusions of this study was that sage-grouse use was highest within 10 m of the Tebuthiuron treatment boundary, suggesting that sage-grouse were still associating with the adjacent sagebrush cover in the control plots. A separate study in northern Utah used an artificial nest experiment to determine differences in nest depredation rates between treated (chained and seeded 25 years prior) and untreated sagebrush; they found that fewer nests were depredated in the treated habitat, although the treatment had occurred quite a long time before the study took place (Ritchie et al. 1994). Overall there is a gap in the literature exploring sage-grouse activities throughout the year within habitat that has been treated for sagebrush thinning or removal.

It is difficult to say what limiting factors are the most significant to sage-grouse population growth, as it is likely that this varies across their range. Conover and Roberts (2016) reviewed and summarized some of the literature concerning this question and suggested several factors related to recruitment that may be important. This included increases in nest and brood predators as well as losses of suitable sagebrush habitat that can provide the necessary dietary requirements for sage-grouse chicks (namely forbs and associated insects) (Conover and Roberts 2016). There are numerous habitat factors that impact nest success for sage-grouse (grass and forb height, sagebrush height and cover, presence of taller shrubs, etc.), however the primary cause for nest failure is often nest depredation (Moynahan et al. 2007, Lockyer et al. 2013).

Recorded nest predators of sage-grouse include ravens, black-billed magpies (*Pica hudsonia*), American badgers (*Taxidea taxus*), western gopher snakes (*Pituophis catenifer*), coyotes (*Canis latrans*), elk (*Cervus elaphus*), weasels (*Mustela* spp.), and

bobcats (*Lynx rufus*) (Holloran and Anderson 2003, Coates et al. 2008, Lockyer et al. 2013). A review by Conover and Roberts (2017) found that ravens were the most common nest predator, responsible for 35% of depredated nests across 13 different sage-grouse studies. In northwestern Nevada, Lockyer et al. (2013) found that ravens were the most frequent sage-grouse nest predator, depredating about 47% of the total depredated nests. There have also been multiple studies that specifically looked at how the presence of ravens negatively impacts the success rate of nesting sage-grouse (Bui et al. 2010, Gibson et al. 2018, Coates et al. 2020). Each of these studies quantified how raven activity could be attributed to the anthropogenic subsidies discussed above. For instance, Coates et al. (2020) were able to use 10 years of raven surveys in the Great Basin to determine anthropogenic and environmental covariates associated with raven detections; they found that sage-grouse nest survival was negatively impacted where raven density exceeded about 0.40 km<sup>-2</sup>.

Because both sagebrush habitat and anthropogenic activities vary throughout the western US, it is difficult to extrapolate the conclusions of one sage-grouse depredation study to different habitats. While the above studies took place primarily in Nevada and Wyoming, there has been little to no research on raven activity in sage-grouse habitat in Utah. It is important to conduct similar studies in Utah for several reasons. First, sage-grouse populations throughout Utah have historically been, and continue to be, fairly fragmented due to the nature of the landscape and due to anthropogenic activities (Schroeder et al. 2004, Utah Division of Wildlife Resources 2019). Studies have shown that populations that are geographically isolated are also genetically isolated; individuals and populations along range margins often have greater variability amongst them (Sexton

et al. 2009). As sage-grouse habitat continues to become more fragmented throughout its entire range, it is useful to use Utah's sage-grouse populations as indicators for what we may expect to see within other populations. Also, Utah has unique characteristics for sage-grouse nesting and brood-rearing habitats that considerably differ from that of other portions of the species' range (Dahlgren et al. 2016). Sage-grouse populations within southern Utah may be disproportionately affected by raven activities as they are often smaller and more fragmented than those in other portions of their range (Frey et al. 2013, Beers 2023).

One of the most common ways to study questions regarding nest depredation events is to set up artificial nests that can mimic the nests of a study species. Within sage-grouse research, artificial nests have been used to compare depredation rates between differing nesting habitats and as a means of identifying nest predators (Ritchie et al. 1994, DeLong et al. 1995, Watters et al. 2002, Dinkins et al. 2013). These artificial nest experiments all involved placing chicken, quail, and/or plasticine eggs underneath sagebrush within a slight bowl impression in the dirt. While there have been multiple studies that placed trail cameras on real sage-grouse nests to identify predators (e.g. Coates et al. 2008), only one of the above artificial nest experiments included trail cameras on or near the artificial nests (Dinkins et al. 2013); the others did not use cameras and used their best educated guesses to determine nest predators (or they were not interested in identifying predators; Ritchie et al. 1994, DeLong et al. 1995, Watters et al. 2002).

In general, artificial nest experiments must be conducted with a lot of intention when it comes to the questions that are being asked and to the conclusions that are being

drawn (Dinkins et al. 2013). For this project, I focused on the predatory behaviors of local ravens rather than drawing conclusions about depredation rates for the local sage-grouse populations. The primary objective of this study was to determine the effect of vegetation treatments (sagebrush thinning and pinyon-juniper removal) on raven depredation rates of simulated sage-grouse nests. I hypothesized that artificial nests would be more likely to be depredated when they were closer to intact pinyon-juniper and when they were within habitat of intact sagebrush (no treatment).

### **Study Area**

The artificial nest experiments took place within the Bald Hills and Panguitch SGMAs within Iron county in southern Utah, USA (Figure 3.1). The nests were placed within BLM land that is utilized by livestock producers (both for cattle and sheep) for much of the year, including during the sage-grouse nesting season. The study site within the Bald Hills SGMA had an average elevation of about 1,930 m. The Buckskin Valley study site within the Panguitch SGMA had an average elevation of about 2,165 m. The habitat within both sites was generally classified as Inter-mountain Basin big sagebrush shrubland that was dominated by various subspecies of big sagebrush (*Artemisia tridentata* spp.), black sagebrush (*Artemisia nova*), rabbitbrush (*Chrysothamnus* sp.) and bitterbrush (*Purshia tridentata*). The prominent grass species present were crested wheatgrass (*Agropyron cristatum*), Indian ricegrass (*Achnatherum hymenoides*), bottlebrush squirreltail (*Elymus elymoides*), bluebunch wheatgrass (*Pseudoroegneria spicata*), and the annual invasive, cheatgrass (*Bromus tectorum*). The forb understory was dominated by scarlet globemallow (*Sphaeralcea coccinea*), longleaf phlox (*Phlox longifolia*), granite prickly phlox (*Linathus pungens*), various lupines (*Lupinus* spp.), rose

heath (*Chaetopappa ericoides*), and roughseed cryptantha (*Cryptantha flavoculata*). Both study sites were within valleys surrounded by pinyon pine (*Pinus edulis*) and juniper (*Juniperus osteosperma*) forest (hereafter called pinyon-juniper forest), however Buckskin Valley also had patches of oak shrubs (*Quercus* spp.) surrounding the sagebrush habitat.

Initially I planned on pooling the artificial nest data between the 2 study sites and analyzing it all together. However, it became evident while in the field that it would be necessary to block the experiment based on the 2 sites. First, the age of vegetation treatment methods differed greatly between the 2 sites. In the Bald Hills, the treatments were lop and scatter (2017) and bullhog (2018) to remove pinyon-juniper forests and harrowing to thin sagebrush (2018; Figure 3.2). In the Buckskin Valley, the treatments were lop and scatter (2019, 2020) and harrow (2015, 2017; Figure 3.3). Second, the study sites differed in their proximity to anthropogenic activities and subsidies that may affect raven activities. The Bald Hills study site is about 15 km south of Minersville, UT (population <900) where there are large pig farming facilities as well as agricultural activities including irrigation pivots. Enoch (population about 8,000) is about 20 km to the south of Bald Hills, where there are livestock producers and irrigation pivots, before blending into Cedar City (population about 37,200). Conversely, Buckskin Valley sits about 9 km east of Interstate 15, and directly adjacent to Utah Highway 20, 2 major travel corridors in southern Utah. It is about 19 km south of the nearest town, Beaver, UT (population about 3,500), and it is about 9 km from irrigated pivots that are situated alongside I-15.

## **Methods**

*Artificial Nest Experiment.* During 2021 and 2022, I randomly generated and then selected locations for 80 artificial nests based on the following spatial criteria. For each study site (Bald Hills and Buckskin Valley), 20 artificial nests were placed within sagebrush habitat that had been treated within the last 10 years and 20 nests were located in areas that had not been treated within this timeframe, resulting in 40 nests per study site. The nests were all placed within 4 km of an active sage-grouse lek; on average, successful sage-grouse nests in Utah are 2.4 km away from the nearest lek (Dahlgren et al. 2016). The nests were placed at a minimum of 160 m apart (Ritchie et al. 1994). Lastly, most of nests were placed within areas where sage-grouse have been known to be active during the nesting season (May-June, 2014-2015) based on transmitter data from previous studies in the area (Hansen 2016). Using this transmitter data, I created a home range estimate for the sage-grouse using ArcGIS Pro (Esri 2022) and selected artificial nest sites within the estimated 80% home range. However, there was a portion of the study area within the Bald Hills that was treated for pinyon-juniper removal starting in 2018, 3 years after the transmitter data was collected. Therefore, I do not have transmitter data showing sage-grouse activity within those study sites.

In both 2021 and 2022, the artificial nests were placed during the first week of May. In 2021, I placed the artificial nests on the landscape at night so that the ravens would be unlikely to observe these efforts. However, this method proved highly time consuming, made it difficult to select the most reasonable nest bush, and led to many trail cameras being placed ineffectively. For these reasons, I placed artificial nests during the day in 2022, and I recorded any raven activities I observed as I placed the nests.



To select a nest shrub, I walked to the selected GPS location; once there I selected the nearest shrub that was an appropriate height (minimum of about 30 cm) and that was slightly taller and had more coverage than its neighboring shrubs (Popham and Gutierrez 2003, Dahlgren et al. 2019). I also selected the sagebrush based on the greatest visually estimated presence of grasses and forbs obstructing the base of the shrub. Next, I dug out a nest bowl using my boot or a rubber mallet, before placing 3 small, white chicken eggs on the dirt while wearing latex gloves (Dinkins et al. 2013). I placed a trail camera (Campark Mini Trail Camera, Campark Electronics, Portland, OR) on every nest; the distances between the cameras and nests varied (60-360 cm) depending on my ability to effectively hide the camera in the nest bush or adjacent shrub. The trail cameras were often attached to nearby shrubs, however stakes (green, plastic, 3 ft tall) were used to mount the cameras when necessary. The nests were left alone for a minimum of 21 days before I returned to collect the trail cameras and conduct habitat analyses.

At each nest shrub, I took vegetation measurements to characterize the habitat at and near the sagebrush. I measured the height and width of the nest shrub as well as the distance between the “nest bowl” and the camera. Next, I created 2 15m transects beginning at the center of the nest shrub and travelling in 2 randomly selected cardinal directions. Along these transects, I placed a measuring tape, and identified each shrub that intercepted this tape. For each intercepting shrub, I measured its height and the length of the tape (cm) it intercepted using the shrub-intercept technique (Canfield 1941). I then calculated an average shrub height and an average shrub intercept value around each nest shrub. Along each of the 2 transects, I measured nest obscurity around the nest

bowl by visually estimating the height of obscuration of a 1m Robel pole while standing 5m from the nest shrub with an eye-level height of about 1.5m (Robel et al. 1970).

*Data Analysis.* I investigated the relationship between artificial nest outcomes and various habitat factors relating to anthropogenic activities and vegetation structure. The response variable for all of my analyses was the binary outcome of whether or not an artificial nest was depredated by a raven. Depending on the placement of the trail camera, it was sometimes possible to observe the raven in the act of depredating the nest (Figure 3.4). Otherwise, a nest was considered depredated by a raven if a raven was observed (via the trail cameras) on the ground or on a nearby shrub; if a raven was only observed flying in the background, this was not considered enough evidence that the raven had found and depredated the nest. The predictor variables included the microhabitat measurements discussed above as well as several variables measuring the distance from the artificial nests to various features of interest. Using ArcGIS Pro (Esri 2022), I measured the distance from each artificial nest to the closest paved road, dirt road, tree (pinyon-juniper), water related feature (e.g., cattle ponds, wildlife guzzlers, livestock catchments, etc.), and depredated artificial nest. If I was unable to determine definitively that a raven visited and depredated a nest, then that data point was removed from the analyses. I analyzed the artificial nest data with a generalized linear model framework using R software (R Core Team 2022) within Rstudio (Posit Team 2022).

I first tested all of the predictor variables for collinearity using the R package ‘corrplot’ with the intention of removing any variables if they were found to have an absolute correlation coefficient of  $>0.7$  (Dormann et al. 2013, Wei and Simko 2021). Considering each study site separately (Bald Hills and Buckskin Valley), I ran single

variable logistic regressions using the ‘glm’ function in the R package ‘stats’ (R Core Team 2022). For each of the 11 predictor variables, any variables that had a P-value of  $<0.15$  were moved into the next analyses (Table 3.1). I then followed a step-wise model building process by hand, limiting each model to combinations of 2 predictor variables. I decided to limit all of the models to only 2 predictor variables to prevent overfitting of the model; this was especially a concern due to the limitations of the sample size per study site. Once I had all the possible models containing the “significant” predictors, I ranked them using Akaike’s Information Criterion corrected for small sample sizes (AICc) (Burnham and Anderson 2002) using the R package ‘AICcmodavg’ (Mazerolle 2020). When comparing models, I looked to see whether the top models were within 2 AICc scores of one another as this may indicate that there is a marginal difference between the performance of the models (Grueber et al. 2011). Finally, I exponentiated the coefficient estimates from the ‘best’ models from each study site to extrapolate meaning from the models.

## **Results**

After removing all of the data points where the nest outcomes (i.e., predator identity) were uncertain, my dataset (for both 2021 and 2022) included 49 artificial nests within the Buckskin Valley site (18 nests in treated habitat and 31 nests in untreated habitat) and 66 artificial nests within the Bald Hills site (35 nests in treated habitat and 31 nests in untreated habitat). Over the 2 years of the study, 18 of the 49 nests were depredated within the Buckskin Valley site, and 31 of the 66 nests were depredated within the Bald Hills site. Unfortunately, many nests from 2021 were removed from the analyses because the nest predators could not be identified due to camera placement. Across both years, the

time from nest placement to depredation ranged from 1 to 18 days (mean = 6.9 days, median = 7 days). For both study sites, the null models were insignificant compared to the models discussed below.

For the Buckskin Valley site, the single-variable logistic regressions suggested that nest outcome (probability of being depredated) was negatively associated with the nest's distance to the nearest depredated nest and positively associated with the nest's distance to the nearest water feature, the height of the nest shrub, and if the nest was in treated habitat (Table 3.1). I used these 4 variables to build 10 generalized linear models (family = binomial), containing 1 or 2 variables each. I used an AICc analysis to compare and rank these models. The top 3 performing models were all within 2 AICc scores of one another, indicating a marginal difference in their predictive abilities (Grueber et al. 2011; Table 3.2). All 3 of the top models included distance to the nearest depredated nest; the top model also included the nest shrub height, while the 2<sup>nd</sup> top model included whether or not the nest was in treated habitat (Table 3.2). The predictor coefficients ( $\beta$ ) for these models were log(odds); to interpret these values, I exponentiated the  $\beta$ 's and calculated their 95% confidence intervals (Table 3.3). The top performing model suggests that for every doubling in distance to the nearest depredated nest, the odds of nest depredation change by a multiplicative factor of 0.44 ( $2^{-1.18}$ ). It also suggests that for every unit (dm) increase in nest shrub height, the odds of nest depredation increase by a multiplicative factor of 1.28 ( $e^{0.25}$ ). The 2<sup>nd</sup> top model suggests a similar trend with the distance to the nearest depredated nest, and it suggests that a nest placed in treated habitat rather than untreated habitat increases the odds of depredation by a multiplicative factor of 3.8 ( $e^{1.3}$ ), or in other words, the odds of depredation go up by 280%.

For the Bald Hills site, the single variable logistic regressions determined a total of 8 “significant” variables (Table 3.1) I built 35 generalized linear models (family = binomial) using these variables; each model was limited to 1 or 2 of these variables. Using the same AICc analysis as above, I ranked and compared the models, focusing on the 2 models that were within 2 AICc scores of one another (Table 3.2). The top model had ‘year’ (2 levels: 2021, 2022) and ‘treatment’ (2 levels: treated, untreated) as the predictor variables. The 2<sup>nd</sup> top model had ‘year’ and ‘treatment type’ (4 levels: none, bullhog, harrow, lop and scatter) as the predictor variables. With both of these logistic regression models, I ran into an issue of complete or quasi-separation whereby the response variable was almost completely divided by the predictor variables; this issue likely arose because of the small sample size and because all of the predictor variables in the models were categorical (Mansournia et al. 2018). In order to better evaluate these models, I ran them using the R package ‘logistf’ which uses the Firth penalization method (Heinze et al. 2022). Both models suggested an increase in the odds of nest depredation if the nest was placed in treated habitat rather than untreated habitat (Appendix B; Table B1), however because ‘year’ was one of the predictor variables, there were still issues of complete separation. I therefore ran the models without ‘year’ using ‘logistf’, and the standard error went down significantly while the models show the same pattern of treated vs untreated habitat (Table 3.4). Using the ‘DHARMA’ package, I tested the top performing models from both study sites for over/underdispersion as well as outliers (Hartig 2022). These diagnostic tests found no significant issues.

After determining that ‘treatment’ was a significant predictor variable for both the Buckskin Valley and Bald Hills sites, I ran Fisher’s exact tests for each site. The Fisher’s

exact test is a more suitable test than the Chi-square test when the sample size is small enough that one of the cells of the expected values in the contingency table is  $<5$  (Bower 2003). The Buckskin Valley data did not reject the null hypothesis; there was no significant relationship between treatment and nest outcome (depredation) (P-value = 0.22). The Bald Hills data, however, did reject the null hypothesis; there was a significant relationship between treatment and nest outcome (P-value  $< 0.001$ ), where significantly more nests were depredated within treated habitat.

## **Discussion**

In all 49 observations of artificial nests being depredated, ravens were the only nest predator (Figure 3.4). Similar to what was reported in Coates et al. (2008), the cameras detected multiple rodents around the nests, however they never were filmed depredating the eggs. The only other potential predator detected via trail camera was a golden eagle (*Aquila chrysaetos*), however, this eagle only arrived after a raven had already depredated the artificial nest.

Within both study sites (Buckskin Valley and Bald Hills), the artificial nests were close enough to one another that it seemed unlikely that their relative distances to anthropogenic factors (such as roads and water features) would vary enough to have a significant effect on nest depredation. While some of these variables were significant within the single-variable analyses (Table 3.1), they were not found to be significant within the top performing models for either study site. Given that previous research has found that anthropogenic subsidies lead to higher rates of sage-grouse nest depredation by ravens (Coates et al. 2014, Gibson et al. 2018, O'Neil et al. 2018, Coates et al. 2020),

it is likely that I would have found more significant trends with the subsidies in my study sites if the artificial nests had been spread out across greater distances, thus creating more variation amongst the nests. The small sample sizes for both study sites somewhat limit the interpretation of the logistic regressions. Many of the analyses were unable to meet the sample size 'rule-of-thumb' of 10:1 subjects to predictors (Harrell, 2001); ideally, I would have had 10 intact nests and 10 depredated nests per variable (e.g. each treatment type).

Within the Buckskin Valley, the most significant variable was how close the artificial nest was to a depredated nest. This was a potential measure of how ravens were able to find and depredate nests that were close to one another. It is likely that once a raven finds one artificial nest, it can develop a search image for nearby nests meaning that nest density has a negative effect on nest outcomes; Ellis et al. (2020) had similar findings when they studied the spatial pattern of snowy plover (*Charadrius nivosus*) nest depredation by ravens. Because I was interested in identifying nest predators, I placed hidden trail cameras on every artificial nest. This may have provided the ravens with an easier way to detect and key-in on the artificial nests. Dinkins et al. (2013) reported a slight trend of camera presence being positively associated with more depredation events. However they had data limitations which meant that they could not include this in their overall model. Bravo et al. (2020) found that the presence of a camera lead to fewer nest depredations but that many predators (mostly corvids) still visited and did not depredate the nest; they hypothesized this behavior was due to neophobia. Most ravens that depredated the artificial nests at my study sites displayed signs of neophobia (e.g.,

approaching the nest and waiting or pecking at the ground near the nest) and were often hesitant at first to depredate the nest.

Within the Bald Hills, the primary predictor variable for artificial nest outcome was whether the nest was placed within treated or untreated habitat. The majority of these nests were placed within habitat that was treated for pinyon-juniper removal via lop and scatter or bullhog in 2017 and 2018, respectively (Figure 3.2). During the single-variable logistic regression analyses, ‘distance to nearest depredated nest’ was also found to be significant; it is possible that the relationship between nest outcome and treatment is simply reflecting the clustering of nests that were found by the ravens. However, the ‘nest shrub height’ was also found to be significant during the single-variable analyses. Interestingly, within the Bald Hills site, the taller shrubs were less likely to be depredated, but in the Buckskin Valley site, the shorter shrubs were less likely to be depredated. According to the state of Utah’s guidelines for sage-grouse breeding habitat within the “Wahsatch” cluster of sage-grouse populations (which includes the Bald Hills SGMA), the q50 for shrub height is 41 cm (Dahlgren et al. 2019). The average shrub height around the artificial nest shrubs within the Bald Hills site was 55 cm ( $\pm$  16 cm SD), and within the Buckskin Valley site was 62 ( $\pm$  20 cm SD). Given the artificial nest outcomes, I hypothesize that in general, much of the sagebrush within the Bald Hills site is still recovering from habitat treatments while some of the sagebrush within the Buckskin Valley is old and overgrown. Dahlgren et al. (2019) cautioned that within their Utah sites, they did not find that there were any “maximum habitat values” that would not be considered habitat for nesting sage-grouse. One study of sage-grouse nesting habitat in California found that shrub height was greater at successful nests than at unsuccessful



nests. However, they did not identify the reasons for nest failures (Popham and Gutierrez 2003). My artificial nest experiment suggests that sagebrush height likely plays an important role in hiding nests from predators, however it is still difficult to determine what the exact habitat guidelines for nest shrub height would be within southern Utah.

### **Management Implications**

Dinkins et al. (2013) discuss three primary elements of artificial nests that likely lead to issues when comparing artificial and real sage-grouse nests: nest placement, nest appearance, and nest vulnerability. Related to these factors, the authors believe that it is difficult to be confident that just because a species of predator depredates an artificial nest, the same species will depredate actual sage-grouse nests within that habitat. This experiment is a good starting point for understanding the potential impacts that past habitat treatments may pose to nesting sage-grouse within the Bald Hills and Panguitch SGMAs. Ravens are clearly abundant within these areas and are active within the sagebrush habitat during the sage-grouse nesting season. In order to better understand the current impact that ravens have on these sage-grouse populations, efforts should be made to monitor actual sage-grouse nests and to trap and track the movements of ravens in the area.

Dahlgren et al. (2019) recommended caution when reducing shrub canopy within sage-grouse habitat given the fact that they found that meeting minimum habitat requirements was more difficult and more significant than having grasses and sagebrush that may be considered too tall. Hess and Beck (2012) found that mowed habitats of Wyoming big sagebrush were unable to meet minimum habitat guidelines in terms of

grass and shrub heights, up to 9 years post-treatment, compared to untreated habitat. In the context of my artificial nest experiments, I did not have enough nests placed within harrow treatments (sagebrush reduction) to determine how these nests fared compared to intact sagebrush. However, my data does suggest that shrub height may be a significant factor for concealing the artificial nests.

Pinyon-juniper removal is a method of augmenting existing sagebrush habitat. While burning pinyon-juniper has been found to promote longer lasting positive effects on sagebrush cover, because the fire burns most of the tree seedbank, the initial sagebrush cover is reduced compared to methods such as cutting (Davies et al. 2019). Frey et al. (2013) found that 4 years after pinyon-juniper removal (via mechanical and hand-thinning methods), the sagebrush habitat did not meet minimum habitat guidelines for shrub and grass height and coverage. Nonetheless, they found that sage-grouse began using the treated areas within the first year of treatment; they hypothesized that this may have reflected how limited suitable habitat availability was in the area. My artificial nest experiments showed a pattern of nests being depredated within both bullhog and lop and scatter treatments (3-5 years post treatment), particularly at the Bald Hills site. Within the context of these projects, land managers may be advised that sage-grouse nests may be more at risk of depredation by ravens until the sagebrush habitat has had adequate time to recover post-treatments.

### **Literature Cited**

Beers, A. T. 2023. Local and Regional Landscape Characteristics Driving Habitat Selection by Greater Sage-Grouse Along a Fragmented Range Margin. All

Graduate Theses and Dissertations. 8695. <<https://doi.org/10.26076/4f07-36a8>>.

Accessed March 6, 2023.

Bower, K. M. 2003. When to Use Fisher's Exact Test. American Society for Quality, Six Sigma Forum Magazine 2(4):35–37.

Bui, T. V. D., J. M. Marzluff, and B. Bedrosian. 2010. Common raven activity in relation to land use in western Wyoming: Implications for Greater Sage-Grouse reproductive success. Condor 112:65–78.

Burnham, K. P. and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. 2nd ed. New York, Springer-Verlag.

Canfield, R. H. 1941. Application of the line interception method in sampling range vegetation. Journal of Forestry 39:388–394.

Coates, P. S., J. W. Connelly, and D. J. Delehanty. 2008. Predators of Greater Sage-grouse nests identified by video monitoring. Journal of Field Ornithology 79(4):421–428.

Coates, P. S. and D. J. Delehanty. 2008. Effects of environmental factors on incubation patterns of greater sage-grouse. The Condor 110(4):627–638.

Coates, P. S., S. T. O'Neil, B. E. Brussee, M. A. Ricca, P. J. Jackson, J. B. Dinkins, K. B. Howe, A. M. Moser, L. J. Foster, and D. J. Delehanty. 2020. Broad-scale impacts of an invasive native predator on a sensitive native prey species within the

shifting avian community of the North American Great Basin. *Biological Conservation* 243(2020):108409.

Coates, P. S., B. G. Prochazka, M. A. Ricca, K. B. Gustafson, P. Ziegler, and M. L. Casazza. 2017. Pinyon and Juniper Encroachment into Sagebrush Ecosystems Impacts Distribution and Survival of Greater Sage-Grouse. *Rangeland Ecology & Management* 70:25-38.

Connelly, J. W. and C. E. Braun. 1997. Long-term changes in sage grouse *Centrocercus urophasianus* populations in western North America. *Wildlife Biology* 3:229–234.

Connelly, J. W., S. T. Knick, M. A. Schroeder, and S. J. Stiver. 2004. Conservation Assessment of Greater Sage-grouse and Sagebrush Habitats. Western Association of Fish and Wildlife Agencies. Unpublished Report. Cheyenne, Wyoming.

Connelly, J. W., M. A. Schroeder, A. R. Sands, and C. E. Braun. 2000. Guidelines to manage sage grouse populations and their habitats. *Wildlife Society Bulletin* 28:967-985.

Conover, M. R. and A. J. Roberts. 2016. Declining populations of greater sage-grouse: where and why. *Human-wildlife Interactions* 10(2):217-229.

Conover, M. R. and A. J. Roberts. 2017. Predators, Predator Removal, and Sage-grouse. *Journal of Wildlife Management* 81(1):7-15.

- Cook, A. A., T. A. Messmer, and M. R. Guttery. 2017. Greater Sage-grouse Use of Mechanical Conifer Reduction Treatments in Northwest Utah. *Wildlife Society Bulletin* 41(1):27-33.
- Dahlgren, D. K., R. Chi, and T. A. Messmer. 2006. Greater Sage-grouse Response to Sagebrush Management in Utah. *Wildlife Society Bulletin* 34(4):975-985.
- Dahlgren, D. K., T. A. Messmer, B. A. Crabb, M. R. Kohl, S. N. Frey, E. T. Thacker, R. T. Larsen, and R. J. Baxter. 2019. Sage-Grouse Breeding and Late Brood-Rearing Habitat Guidelines in Utah. *The Wildlife Society Bulletin* 43(4):576-589.
- Dahlgren, D. K., T. A. Messmer, B. A. Crabb, R. T. Larsen, T. A. Black, S. N. Frey, E. T. Thacker, R. J. Baxter, and J. D. Robinson. 2016. Seasonal Movements of Greater Sage-grouse Populations in Utah: Implications for Species Conservation. *Wildlife Society Bulletin* 40(2):288-299.
- Davies, K. W., J. D. Bates, and A. M. Nafus. 2012. Comparing Burned and Mowed Treatments in Mountain Big Sagebrush Steppe. *Environmental Management* 50:451-461.
- Davies, K. W., R. C. Rios, J. D. Bates, D. D. Johnson, J. Kerby, C. S. Boyd. 2019. To burn or not to burn: Comparing reintroducing fire with cutting an encroaching conifer for conservation of an imperiled shrub-steppe. *Ecology and Evolution* 9(16), 9137e9148.

- DeLong, A. K., J. A. Crawford, D. C. DeLong. 1995. Relationships between vegetational structure and predation of artificial sage grouse nests. *Journal of Wildlife Management* 59(1):88-92.
- Dinkins, J. B., M. R. Conover, C. P. Kirol, and S. T. Mabray. 2013. Do artificial nests simulate nest success of greater sage-grouse? *Human-wildlife Interactions* 7(2):299-312.
- Dormann, C. F., J. Elith, S. Bacher, C. Buchmann, G. Carl, G. Carre, J. R. Garcia Marquez, B. Gruber, B. Lafourcade, P. J. Leitao, T. Munkemuller, C. McClean, P. E. Osborne, B. Reineking, B. Schroder, A. K. Skidmore, D. Zurell, and S. Lautenbach. 2013. Collinearity: a review of methods to deal with it and a simulation study evaluating their performance. *Ecography* 36:27–46.
- Ellis, K. S., R. T. Larsen, and D. N. Koons. 2020. Dependence of spatial scale in landscape associations with cause-specific predation of snowy plover nests. *Ecosphere* 11(9):e03257. [10.1002/ecs2.3257](https://doi.org/10.1002/ecs2.3257)
- Esri. 2022. ArcGIS Pro: Version 3.0. Environmental Systems Research Institute, Redlands, CA.
- Frey, S. N., R. Curtis, and K. Heaton. 2013. Home Range and Movements of Greater Sage-grouse in its Southern-most Distribution. All Current Publications, paper 1192. <[https://digitalcommons.usu.edu/extension\\_curall/1192](https://digitalcommons.usu.edu/extension_curall/1192)>. Accessed November 12, 2020.

- Gibson, D., E. J. Blomberg, M. T. Atamian, S.P. Espinosa, and J.S. Sedinger. 2018. Effects of power lines on habitat use and demography of Greater Sage-grouse. *Wildlife Monographs* 200:1-41.
- Grueber, C. E., S. Nakagawa, R. J. Lawn, and I. G. Jamieson. 2011. Multimodel inference in ecology and evolution: challenges and solutions. *Journal of Evolutionary Biology* 24: 699-711.
- Harrell, F.E. 2001. *Regression Modeling Strategies: With Applications to Linear Models, Logistic Regression, and Survival Analysis*. Springer, New York.
- Hartig, F. 2022. DHARMA: Residual Diagnostics for Hierarchical (Multi-Level / Mixed) Regression Models. R package version 0.4.5.
- Heinze G., M. Ploner, L. Jiricka. 2022. logistf: Firth's Bias-Reduced Logistic Regression. R package version 1.24.1, <https://CRAN.R-project.org/package=logistf>.
- Holloran, M. J., and S. H. Anderson. 2003. Direct identification of northern sage-grouse, *Centrocercus urophasianus*, nest predators using remote sensing cameras. *Canadian Field-Naturalist* 117(2):308-310.
- Lockyer, Z. B., P. S. Coates, M. L. Casazza, S. Espinosa, and D. J. Delehanty. 2013. Greater sage-grouse nest predators in the Virginia Mountains of northwestern Nevada. *Journal of Fish and Wildlife Management* 4:242–254.

- Mansournia, M. A., A. Geroldinger, S. Greenland, and G. Heinze. 2018. Separation in Logistic Regression: Causes, Consequences, and Control. *American Journal of Epidemiology* 187(4): 864-870.
- Mazerolle, M. J. 2020. AICcmodavg: Model selection and multimodel inference based on (Q)AIC(c). R package version 2.3-1. <https://cran.r-project.org/package=AICcmodavg>.
- Miller, R. F., T. Svejcar, and J. A. Rose. 1999. Conversion of shrub steppe to juniper woodland. *Proceedings: Ecology and Management of Pinyon-Juniper Communities Within the Interior West*. USDA Forest Service, RMRS-P-9:275-284.
- Miller, R. F., T. J. Svejcar, and J. A. Rose. 2000. Impacts of western juniper on plant community composition and structure. *Journal of Range Management* 53:574–585.
- Moynahan, B. J., M. S. Lindberg, J. J. Rotella, and J. W. Thomas. 2007. Factors affecting nest survival of greater sage-grouse in northcentral Montana. *Journal of Wildlife Management* 71(6):1773-1783.
- R Core Team. 2022. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL <https://www.R-project.org/>.
- Reinhardt, J. R., S. Filippelli, M. Falkowski, B. Allred, J. D. Maestas, J. C. Carlson, and D. E. Naugle. 2020. Quantifying Pinyon-Juniper Reduction within North



America's Sagebrush Ecosystem. *Rangeland Ecology & Management* 73:420–432.

Ritchie, M. E., M. L. Wolfe, and R. Danvir. 1994. Predation of artificial Sage Grouse nests in treated and untreated sagebrush. *Great Basin Naturalist* 54(2):122–129.

Robel, R., J. Briggs, A. Dayton, and L. Hulbert. 1970. Relationships between visual obstruction measurements and weight of grassland vegetation. *Journal of Range Management* 24:295–297.

Roth, C. L., S. T. O'Neil, P. S. Coates, M. A. Ricca, D. A. Pyke, C. L. Aldridge, J. A. Heinrichs, S. P. Espinosa, and D. J. Delehanty. 2022. Targeting Sagebrush (*Artemisia* Spp.) Restoration Following Wildfire with Greater Sage-Grouse (*Centrocercus Urophasianus*) Nest Selection and Survival Models. *Environmental Management* 70:288–306.

Schroeder, M. A., C. L. Aldridge, A. D. Apa, J. R. Bohne, C. E. Braun, S. D. Bunnell, J. W. Connelly, P. A. Deibert, S. C. Gardner, M. A. Hilliard, G. D. Kobriger, S. M. McAdam, C. W. McCarthy, J. J. McCarthy, D. L. Mitchell, E. V. Rickerson, and S. J. Stiver. 2004. Distribution of sage-grouse in North America. *Condor* 106:363–376.

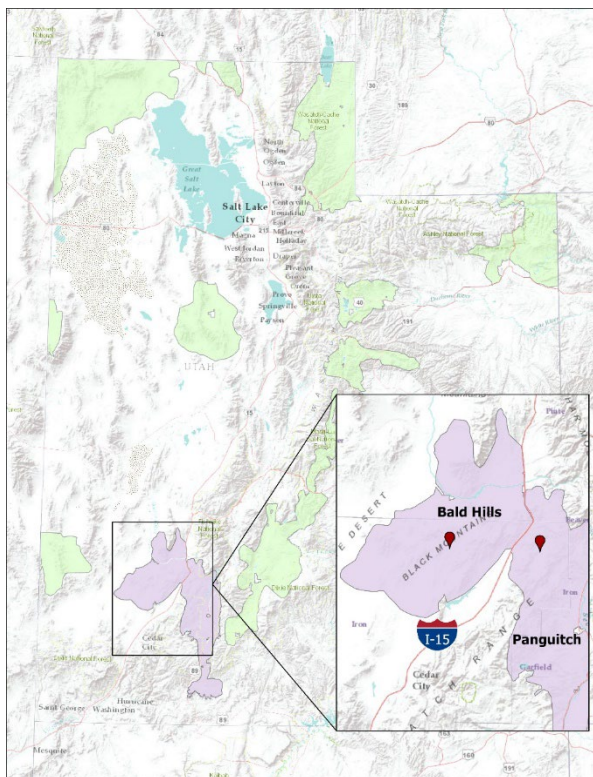
Schupp, E. W., C. S. Boyd, and S. Green. 2015. Woody Fuels Reduction in Wyoming Big Sagebrush Communities. *Great Basin Factsheet Series* (13).

- Sexton, J. P., P. J. McIntyre, A. L. Angert, and K. J. Rice. 2009. Evolution and ecology of species range limits. *Annual Review of Ecology, Evolution, and Systematics* 40:415–436.
- Popham, G. P. and R. J. Gutierrez. 2003. Greater sage-grouse *Centrocercus urophasianus* nesting success and habitat use in northeastern California. *Wildlife Biology* 9:327-334.
- Porensky, L. M., R. McGee, and D. W. Pellatz. 2020. Long-term grazing removal increased invasion and reduced native plant abundance and diversity in sagebrush grassland. *Global Ecology and Conservation* 24:e01267.
- Posit team. 2022. RStudio: Integrated Development Environment for R. Posit Software, PBC, Boston, MA. URL <http://www.posit.co/>.
- Pyke, D. A., S. E. Shaff, J. C. Chambers, E. W. Schupp, B. A. Newingham, M. L. Gray, L. M. Ellsworth. 2022. Ten-year ecological responses to fuel treatments within semiarid Wyoming big sagebrush ecosystems. *Ecosphere* 12(7):e4176.
- Utah Division of Wildlife Resources. 2019. Utah Conservation Plan for Greater Sage-grouse. <[https://wildlife.utah.gov/sage-grouse/Utah\\_Greater\\_Sage-grouse\\_Plan.pdf](https://wildlife.utah.gov/sage-grouse/Utah_Greater_Sage-grouse_Plan.pdf)>. Accessed November 10, 2021.
- Watters, M. E., T. L. Mclash, C. L. Aldridge, and R. M. Brigham. 2002. The effect of vegetation structure on predation of artificial Greater Sage-Grouse nests. *Ecoscience* 9(3):314-319.

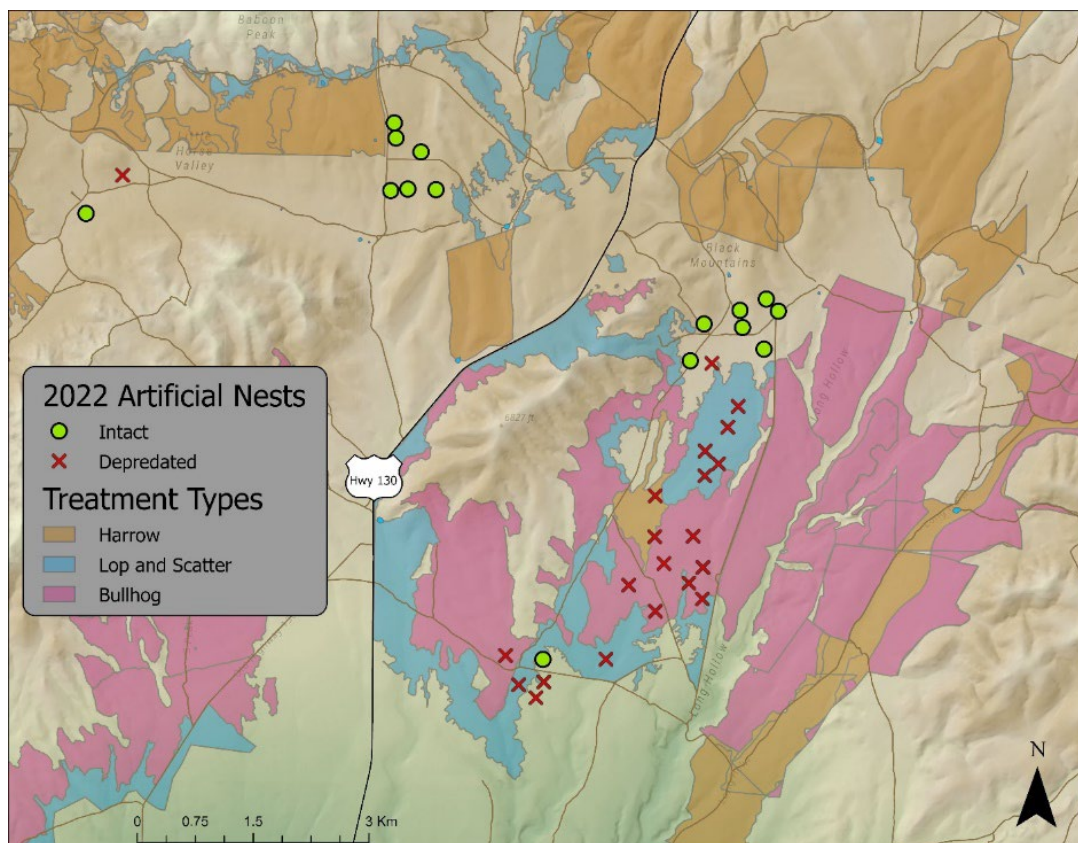
Wei, T. and V. Simko. 2021. R package 'corrplot': Visualization of a Correlation Matrix:

Version 0.92. Available from <https://github.com/taiyun/corrplot>.

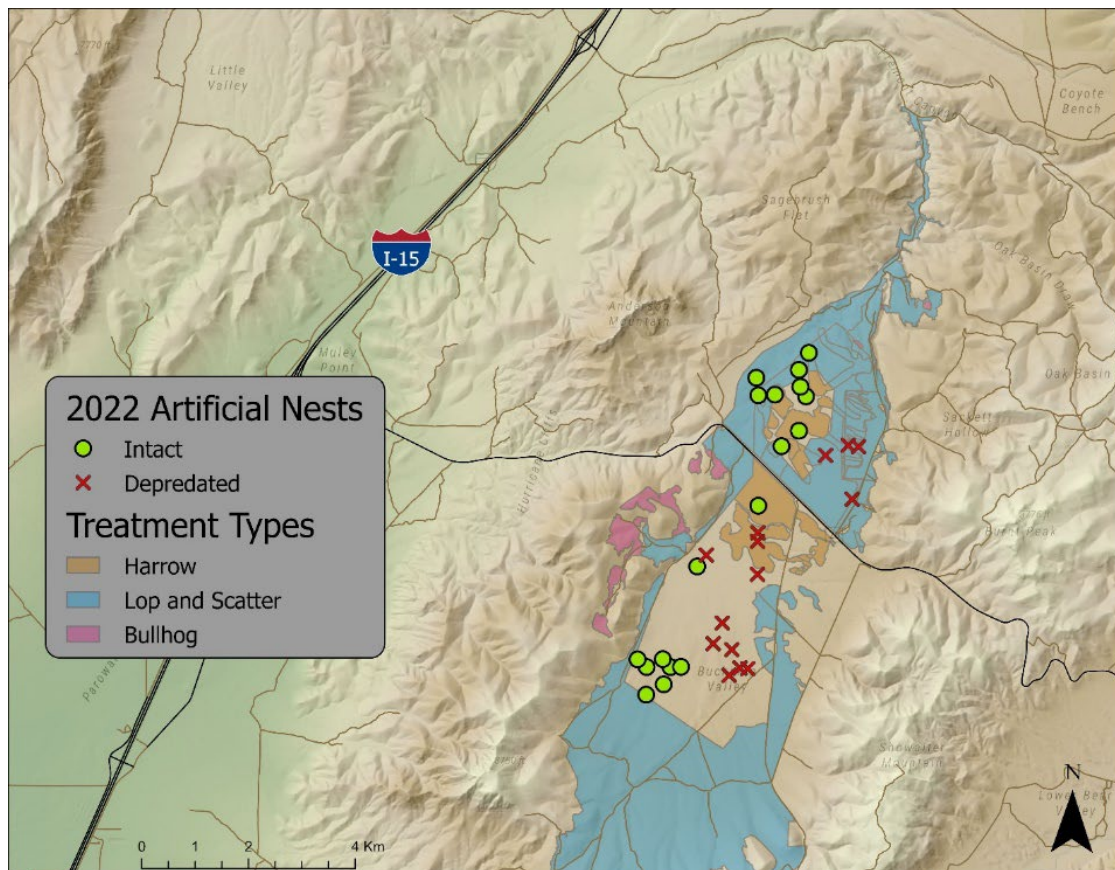
## Figures



**Figure 3.1.** Map of Sage-grouse Management Areas (SGMAs) in the state of Utah, USA. The 2 study sites for the artificial nest experiments (Bald Hills and Buckskin Valley) are indicated with red markers. Artificial nest experiments took place in May–June 2021–2022.



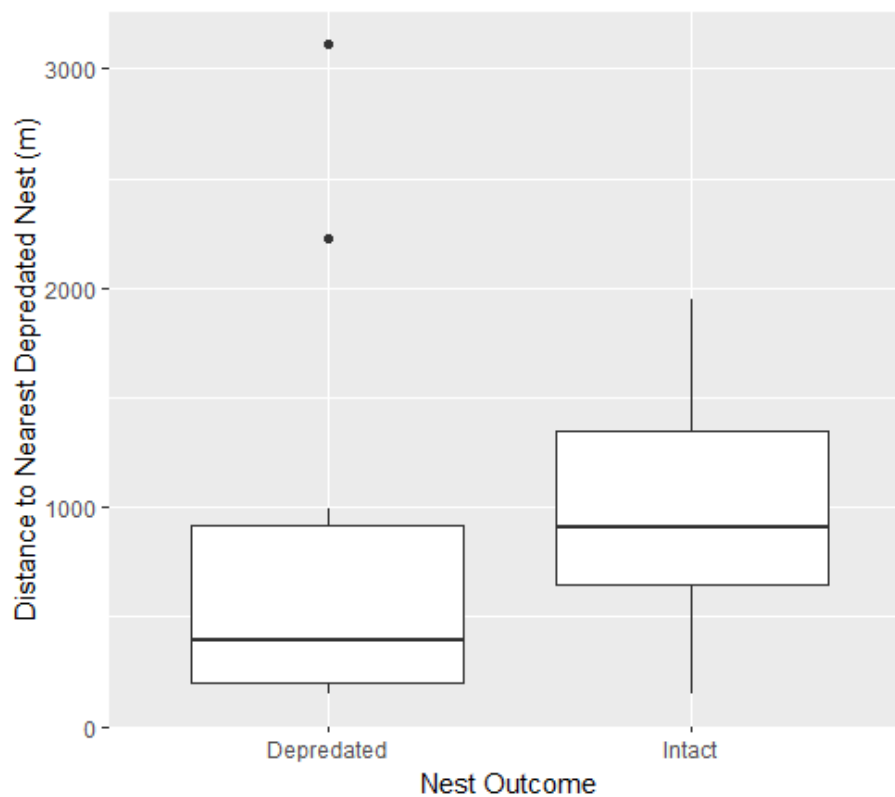
**Figure 3.2.** Map of artificial nest outcomes for 2022 within the Bald Hills study site, Iron County, Utah, USA. The harrow, lop and scatter, and bullhog treatments took place in 2018, 2017, and 2018, respectively.



**Figure 3.3.** Map of artificial nest outcomes for 2022 within the Buckskin Valley study site, Iron County, Utah, USA. The harrow and lop and scatter treatments took place in 2015 and 2019, respectively.



**Figure 3.4.** Trail camera image of a raven depredating an artificial nest in May of 2021, Iron County, Utah, USA.



**Figure 3.5.** Artificial nest outcomes in the Buckskin Valley depending on distance of artificial nest to the nearest depredated nest. All nests were placed 2021—2022 in Iron County, UT, USA.



## Tables

**Table 3.1.** Variables that were selected after single variable logistic regression analyses describing depredation of 115 artificial nests ( $p > 0.15$ ), Iron County, Utah, USA, 2021–2022.

Buckskin Valley				
Variable	Description	$\beta$	SE	P-value
Distance depredated nest	log(distance to the nearest depredated nest) in m	-1.13	0.44	<0.01
Distance water	log(distance to the nearest water related feature) in cm	0.001	0.0009	0.14
Nest shrub height	Height of the artificial nest shrub in dm	0.25	0.13	0.05
Treatment	Treatment in 2 levels: untreated vs treated	0.89	0.62	0.15
Bald Hills				
Variable	Description	$\beta$	SE	P-value
Distance water	log(distance to the nearest water related feature) in m	2.59	0.75	<0.001
Distance depredated nest	log(distance to the nearest depredated nest) in m	-1.53	0.41	<0.001
Distance dirt road	log(distance to the nearest dirt road) in m	1.16	0.39	<0.01
Distance tree	log(distance to the nearest tree) in m	0.6	0.27	<0.05

Nest shrub height	Height of the artificial nest shrub in dm	-0.34	0.16	<0.05
Year	2 levels: 2021 vs 2022	1.03	0.51	<0.05
Treatment type (Bullhog)	Treatment type in 4 levels: Bullhog vs none	2.95	0.72	<0.001
Treatment type (L&S)	Treatment type in 4 levels: lop & scatter vs none	3.3	0.96	<0.001
Treatment	Treatment in 2 levels: untreated vs treated	3.13	0.67	<0.001

---

**Table 3.2.** Top performing generalized linear models predicting artificial nest outcome per study site that were within 2 Akaike's information criterion units (adjusted for small sample sizes;  $AIC_c$ ) ( $n = 115$ ), Iron County, Utah, 2021–2022.

Buckskin Valley				
Model	K	$AIC_c$	$\Delta AIC_c$	$AIC_c$ Weight
Distance to depredated nest + nest shrub height	3	59.03	0	0.38
Distance to depredated nest + treatment	3	59.47	0.44	0.3
Distance to depredated nest	2	60.78	1.76	0.16
Bald Hills				
Model	K	$AIC_c$	$\Delta AIC_c$	$AIC_c$ Weight
Year + treatment	3	50.67	0	0.6
Year + treatment type	5	51.73	1.05	0.35

**Table 3.3.** Parameter estimates for the top 3 performing generalized linear models (family = binomial, link = logit) for the Buckskin Valley study site that were within 2 Akaike's information criterion units (adjusted for small sample sizes;  $AIC_c$ ) ( $n = 115$ ), Iron County, Utah, USA, 2021–2022.

Model	Variables	$\beta$	SE	Exp( $\beta$ ) (95% CI)
Model 1	log(Distance to depredated nest) (m)	-1.18	0.46	0.31 (0.11, 0.72)
$\Delta AIC_c = 0$	Artificial nest shrub height (dm)	0.25	0.13	1.28 (1.0, 1.7)
Model 2	log(Distance to depredated nest) (m)	-1.3	0.48	0.27 (0.10, 0.65)
$\Delta AIC_c = 0.44$	Treatment: Untreated vs Treated	1.3	0.72	3.8 (0.96, 17)
Model 3	log(Distance to depredated nest) (m)	-1.1	0.44	0.32 (0.13, 0.72)
$\Delta AIC_c = 1.76$				

**Table 3.4.** Parameter estimates for the top 2 performing generalized linear models (family = binomial, link = logit) for the Bald Hills study site that were within 2 Akaike's information criterion units (adjusted for small sample sizes; AIC<sub>c</sub>) ( $n = 115$ ), Iron County, Utah, USA, 2021–2022.

Model	Variables	$\beta$	SE	Exp( $\beta$ ) (95% CI)
Model 1	Treatment: Untreated vs Treated	2.98	0.64	19.8 (6.08, 77.90)
Model 2	Treatment Type: none vs bullhog	2.8	0.69	16.5 (4.65, 70.40)
	Treatment Type: none vs harrow	3.42	1.63	30.6 (2.06, 4503)
	Treatment Type: none vs L&S	3.03	0.88	20.8 (4.18, 144)

## CHAPTER IV

## CONCLUSIONS

Wildlife biologists, land managers, livestock producers, and farmers have all taken notice of the rise in common raven (*Corvus corax*; hereafter raven) populations throughout the intermountain west. By depredating calves and by roosting and defecating on equipment, they have become a nuisance to many people who live on working lands in the west. In the context of wildlife and natural resource management, ravens have become a problematic predator for several sensitive species in Utah, including the greater sage-grouse (*Centrocercus urophasianus*; hereafter sage-grouse). My thesis was designed to study the presence and behaviors of ravens within and near the Bald Hills and Panguitch Sage-grouse Management Areas (SGMAs) in order to better understand the threat that they pose as nest predators to the local sage-grouse populations. These sage-grouse are near the southern-most extent of their distribution, and their populations are smaller and more fragmented than they are in other portions of their range.

In Chapter 2, I utilized point-count survey data ( $n = 810$  surveys) from 2021 and 2022 in order to examine what factors led to higher raven counts and higher rates of raven occurrence throughout the Bald Hills and Panguitch SGMAs. These survey points varied in their proximity to raven point-source subsidies such as towns, paved roads, and irrigated croplands. The surveys themselves varied in their calendar date (February - June), time of day, and whether or not livestock were present. By counting the number of ravens present, I was able to model this data in 2 different ways, with the continuous response variable of the number of raven present and with the binary response variable of

raven occurrence. Both fitted models had the same three significant fixed effects: Julian calendar date, distance from the survey point to the nearest agricultural land, and livestock presence/absence during the survey. The primary difference between the 2 models was that when the response variable was the number of ravens present, there was a significant interaction term between the livestock presence and the distance to agriculture; this means that one can't consider these 2 predictor variables as independent from one another. However, when considering raven occurrence as the response variable, livestock presence increases the odds of raven occurrence by a multiplicative factor of 3.61 (95% CI: 2.33, 5.67), independent of the survey's proximity to agriculture. Also, for every unit increase in the log-transformed distance to agriculture, the odds of raven occurrence decreased by a multiplicative factor of 0.64 (95% CI: 0.52, 0.78), independent of whether or not livestock is present. Lastly, looking across both years, raven occurrence decreased by approximately 15% between the end of winter (February-April;  $\bar{x} = 0.52$ ) and the start of summer (May-June;  $\bar{x} = 0.36$ ). This difference in raven occurrences across the seasons likely reflects the decrease in raven movements that has been previously observed between the nonbreeding and breeding seasons.

In Chapter 3, I conducted a close examination of the risks that ravens pose as nest predators to the local sage-grouse by completing an artificial nest experiment during the sage-grouse breeding seasons in 2021 and 2022. This experiment comprised of placing chicken eggs underneath individual sagebrush with trail cameras in order to determine whether or not the nest was depredated by a raven. I placed these nests within sagebrush that was intact, sagebrush that had been thinned via harrow treatments, and in sagebrush that had grown after pinyon-juniper removal treatments (via lop and scatter or bullhog). I

set up these artificial nests within treated and untreated sagebrush in 2 different study sites: Bald Hills and Buckskin Valley. I fitted and compared separate logistic regression models for each of these study sites. In the Bald Hills site, I found that the nests placed within habitat that had been treated for pinyon-juniper removal were more likely to be depredated than nearby intact sagebrush. This may have been an indication that the sagebrush community had not had enough time to recover and propagate since the pinyon-juniper removal treatments (both bullhog and lop and scatter) that took place in 2017 and 2018. In the Buckskin Valley site, I found that ravens were more likely to depredate nests placed under taller shrubs and under shrubs that were in close proximity to a nest that the ravens had also found. The fitted model for the Buckskin Valley site suggested that for every unit (dm) increase in nest shrub height, the odds of nest depredation increased by a multiplicative factor of 1.28. More research is needed in order to understand the differences in how these various vegetation treatments alter the sagebrush and in how this may affect nesting sage-grouse. However, my findings may help managers understand that the recovery time for sagebrush within pinyon-juniper removal treatments may be longer than previously thought, particularly if the aim is to have sagebrush that can successfully deter sage-grouse nest predators.

This thesis examined the current threat of ravens as sage-grouse nest predators within southern Utah. My research supports a growing body of literature that has been detailing the anthropogenic subsidies that attract and bolster raven populations throughout the intermountain west. My research makes clear the difficulties of managing a sensitive, sagebrush obligate species like the sage-grouse within a fragmented habitat that is surrounded by the human activities and industries which support raven

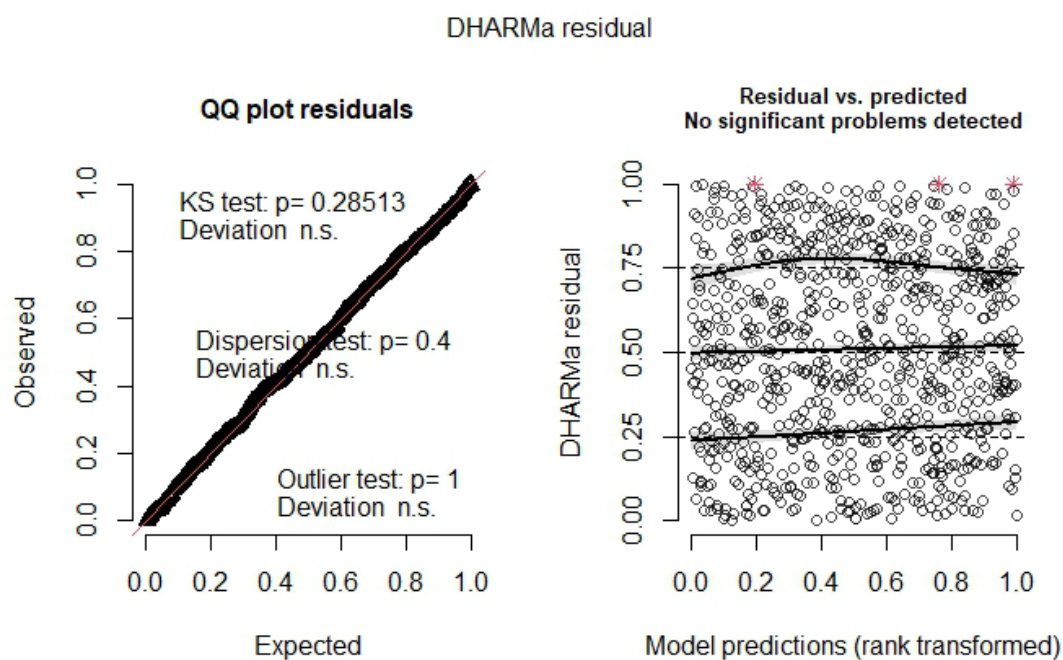


populations. In order to prevent nest depredation events, it is important to reduce food, water, and habitat subsidies near lekking, nesting, and brood-rearing sage-grouse. One important management strategy would be, wherever it is feasible to do so, minimizing instances of livestock ranging within these habitats from March through July. Learning how to manage raven populations via removing subsidies would benefit not only sage-grouse, but other sensitive species in the southwest such as the desert tortoise (*Gopherus agassizii*) and the pinyon jay (*Gymnorhinus cyanocephalus*), both of which are prey of the raven.

APPENDICES

## APPENDIX A

## CHAPTER II SUPPORTING INFORMATION



**Figure A1.** DHARMA’s residual diagnostic plots for the fitted generalized linear mixed model: number of ravens  $\sim (1 | \text{survey point and year}) + \text{livestock present} + \log(\text{distance to ag}) + \text{ordinal date}$ . No dispersion issues were detected. The data was from 810 point-count surveys of the common raven (*Corvus corax*), Iron and Beaver counties, Utah, USA; February–June 2021–2022.

**Table A1.** Parameter estimates of the fixed effects from a fitted generalized linear mixed model (family = negative binomial, link = logit) where the outliers were removed ( $n = 733$  surveys; no. of groups = 87), Iron and Beaver counties, Utah, USA; February–June 2021–2022.

Model				
Number of ravens $\sim (1   \text{survey point AND year}) + \text{livestock present} + \log(\text{distance to ag}) + \text{ordinal date}$				
Fixed Effects	Description	$\beta$	SE	P-value
Livestock present	No livestock present during survey vs livestock present (within 1km)	0.64	0.16	<0.001
Log(distance to ag)	log(distance to the nearest agriculture)	-0.43	0.08	<0.001
Ordinal date	Date according to Julian calendar of the given year	-0.009	0.002	<0.001

## APPENDIX B

## CHAPTER III SUPPORTING INFORMATION

**Table B1.** Top logistic regression models for artificial nest outcomes in the Bald Hills study site with year included as a parameter. Year was removed for the primary analysis in order to deal with complete separation in the model ( $n = 115$ ), Iron County, Utah, USA, 2021-2022.

Model	Variables	$\beta$	SE	Exp( $\beta$ )	95% CI Exp( $\beta$ )
Model 1	Year 2023 vs 2022	3.5	1.5	33.7	(3.9, 4447)
$\Delta AIC_c = 0$	Treatment: Untreated vs Treated	5	1.5	144	(16, 19163)
Model 2	Year 2023 vs 2022	4.35	1.77	77.2	(4.00, 2.25)
$\Delta AIC_c = 1.05$	Treatment Type: none vs bullhog	6.04	1.87	420	(22, 141449)
	Treatment Type: none vs harrow	2.84	1.64	17	(1.12, 2528)
	Treatment Type: none vs L&S	4	1.42	54.8	(5.38, 7448)