COLORADO PARKS & WILDLIFE

2020 Avian Research Summary Report

JANUARY 2021

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WILDLIFE RESEARCH SUMMARIES

JANUARY – DECEMBER 2020

AVIAN RESEARCH PROGRAM

COLORADO DIVISION OF PARKS AND WILDLIFE
Research Center, 317 W. Prospect, Fort Collins, CO 80526

Executive Summary

This Wildlife Research Report contains abstracted summaries of wildlife research projects conducted by the Avian Research Section of Colorado Parks and Wildlife (CPW) during 2020. These are long-term projects (2–10 years) in various stages of completion, each of which addresses applied questions to benefit the management of various bird species and wildlife habitats in Colorado. More technical and detailed reports of most of these projects can be accessed from the project principal investigator listed at the beginning of each summary, or on the CPW website at http://cpw.state.co.us/learn/Pages/ResearchBirds.aspx and http://cpw.state.co.us/learn/Pages/ResearchHabitat.aspx.

In 2020, research projects in the Section address various aspects of the ecology and management of wildlife populations and the habitats that support them, human-wildlife interactions, and new approaches to field methods in wildlife management. This report includes summaries of 14 current research projects addressing management-related information needs for a variety of species of conservation concern and game species and their habitats. These projects are grouped under Greater Sage-Grouse Conservation (5 summaries), Wildlife Habitat Conservation (2 summaries), Spatial Ecology (1 summary), Grassland Bird Conservation (1 summary), Raptor Conservation (2 summaries), Quail Conservation (2 summaries), and Wetland Bird Conservation (1 summary).

Also included in this report is a listing of publications, presentations, workshops and participation on various committees and working groups by Avian Research staff during 2020. Communicating research results and using their subject matter expertise to inform management and policy issues is a priority for CPW scientists. Copies of peer-reviewed research publications can be obtained from the CPW Library.

We are grateful for the numerous collaborations that support these projects and the opportunity to work with and train graduate students and technicians that will serve wildlife management in the future. Research collaborators include statewide CPW personnel, Colorado State University, University of Nebraska-Lincoln, University of Wisconsin-Madison, Bureau of Land Management, U.S. Fish and Wildlife Service, U.S. Geological Survey, City of Fort Collins, Species Conservation Trust Fund, GOCO YIP internship program, Colowyo Coal Company L.P., EnCana Corp, ExxonMobil/XTO Energy, Marathon Oil, WPX Energy, Conoco-Phillips, Rocky Mountain Bird Observatory, Denver Audubon, and the private landowners who have provided access for research projects.

STATE OF COLORADO
Jared Polis, *Governor*

DEPARTMENT OF NATURAL RESOURCES
Dan Gibbs, *Executive Director*

PARKS AND WILDLIFE COMMISSION

DIRECTOR'S LEADERSHIP TEAM

AVIAN RESEARCH STAFF

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Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Greater sage–grouse response to surface mine mitigation

Period Covered: January 1 – December 31, 2020

Principal Investigators: Anthony D. Apa tony.apa@state.co.us and A. Kircher

Project Collaborators: Bill deVergie, Area Wildlife Manager; Brad Petch, Senior Terrestrial Biologist; Trevor Balzer, Sagebrush Habitat Coordinator; Kathy Griffin, Grouse Conservation Coordinator; Brian Holmes, Conservation Biologist, Colowyo Coal Company, L.P., Tri-State Energy; R. Scott Lutz, University of Wisconsin-Madison

EXTENDED ABSTRACT

The greater sage-grouse (*Centrocercus urophasianus;* hereafter, sage-grouse) has experienced extensive habitat loss and regional population declines as a result of agriculture, mining practices, and human settlement. Some of the greatest impacts to sage-grouse populations are due to anthropogenic disturbances such as land conversions, and agricultural and natural resource development.

As natural resource development increases, sage-grouse habitat continues to decline range-wide. The effects of anthropogenic disturbance on sage-grouse survival depends on the age of the sage-grouse and the type of disturbance: direct or indirect. Direct impacts include habitat loss due to roads, facilities, well pads, and coal pits, for example, while indirect impacts include the consequences of increased traffic creating excess noise, or predators being attracted to anthropogenic structures and potential perches. One major large-scale disturbances impacting sage-grouse and their habitat is oil and natural gas development (Green et al. 2017, Harju et al. 2010, Holloran et al. 2010, Walker et al. 2007, and others). Although there have been several studies investigating the response of sage-grouse to oil and natural gas development, there has been limited research on the influence of surface coal mine development and the resulting disturbance.

Surface coal mining alters landscape topography and removes all vegetation within the disturbance footprint (Sawyer and Crowl 1968), but there is limited data on how male and/or female sage-grouse lek attendance is impacted by coal mining. More data is needed to understand if or how sage-grouse survival is impacted. We compared sage-grouse survival and lek attendance during pre- and post-coal mine development between control (no mine) and treatment (new mine) areas in Moffat County Colorado. The objectives of our study were to (1) evaluate male and female survival and nest survival; (2) analyze marked GPS-PTT- and VHF-marked male and VHF-marked female lek attendance and duration; and (3) assess a new rump-mount transmitter harness design.

Our study included data collected during several prior studies in an effort to understand how, or if, sage-grouse respond to surface coal mine development. New mine (Collom Mine Site; CMS) construction started in the early-summer of 2018 and was adjacent to the older existing South Taylor Mine Site (STMS) that began construction in the late 1970's.

From 2001–2007, we captured 10 adult and 5 yearling males and 299 adult and 147 yearling females. From 2017–2019, we captured 99 adult and 53 yearling males and 117 adult and 67 yearling females from 2017–2019. Ultimately, we used 630 females and 128 males to estimate and model

survival. We employed a Cox proportional-hazards regression model (hereafter, Cox regression) to assess male and female survival (Cox 1972). The most parsimonious model for female survival included age, while the most parsimonious male survival model was the null model; none of the covariates we measured were influential in describing male survival. The 25-week Kaplan-Meier product-limit estimates for females over all years (2001–2007, 2017–2019) ranged from 0.593 (CI = 0.492–0.715) to 0.757 (CI = 0.640–0.896), respectively. Male survival ranged from 0.315 (CI = 0.217–0.458) to 0.614 (CI = 0.399–0.943) from 2017–2019.

We estimated daily nest survival for 322 nests monitored from 2001–2008 and 2017–2019. One hundred sixty-one nests were successful (>1 egg hatched) and 161 were unsuccessful (depredated or abandoned). One hundred forty-six of these nests in the control and 176 nests were in the treatment. The daily survival rate (DSR) was 0.962 (CI = 0.953–0.970) and 0.973 (CI = 0.966–0.978) for nests in the control and treatment, respectively. The years following CMS construction had a nearly equivalent daily survival rate compared to the years prior to the construction of the CMS.

We also assessed if sage-grouse distribution across the landscape was influenced by mining. We used a two-sample bivariate Kolmogorov-Smirnov test to assess the null hypothesis that individual nest placement had no inherent pattern across the landscape in relation to mine boundaries ($\alpha \leq 0$ .05). We also simulated the bivariate null distance distribution by taking random locations uniformly in the treatment area. We used 5,401 male and female VHF and resampled GPS-PTT treatment locations from the pre-mine disturbance period and 6,839 male and female VHF and GPS-PTT treatment locations from the post-mine disturbance period. We also generated 12,240 random locations to assess a potential disturbance response. The joint distribution of distance from either the STMS or the CMS illustrated no significant difference between time periods ($P = 0.25$). There was no significant difference we when compared the two joint distributions against the null distance distribution simulated by the random locations (P =0.64 for pre-disturbance, $P = 0.60$ for post-disturbance). Our analysis suggested that an individual's distance from the coal mines was not the only factor potentially influencing their distribution on the landscape.

We also investigated sage-grouse lek attendance in relation to mine disturbance. To do this, we deployed 11 VHF dataloggers on 11 leks in our study area in 2018 and 2019. Our dataloggers documented any individual's transmitter frequency if it was positioned a designated distance from the datalogger (display area or lek). We also monitored the activities of GPS-PTT-marked males for comparison. As such, we were able to estimate when individuals were arriving and leaving leks, how many leks they visited in the breeding season, and how many days they attended each lek. We used a Kruskal-Wallis nonparametric ANOVA to test factor variables that may affect lek attendance (sex [male or female], age [adult or yearling], area [control or treatment], transmitter type [GPS or VHF], and year [2018 or 2019]). None of the biologically meaningful comparisons were significantly different ($P < 0.05$). In addition, male attendance did not differ ($P = 0.531$) between GPS-PTT and VHF-marked males. A majority (90%) of males and females attended 1 or 2 leks in a season in 2018 and 85% attended <2 leks in 2019. On average, adult and yearling males attended leks for 25.96 and 15.34% of the breeding season, respectively. Females attended for <10% of the breeding season.

Our final objective focused on a new rump-mounted transmitter harness design. We discovered abrasions resulting from a widely accepted harness design. In order to limit abrasions, we developed a novel harness design and standardized attachment protocol. Our new rump-mount harness design limited abrasions and potentially reduced data collection bias due to fewer transmitter injuries and increased well-being of marked individuals (Kircher et al. 2020).

Our project provides new information about the short-term response of sage-grouse coal mine development using 7 years of pre-disturbance data and 2 years of post-disturbance data. Our study was also the first to investigate how sage-grouse survival and nest survival were influenced by surface coal mine development. We recommend additional long-term data collection to understand the long-term response of sage-grouse to surface coal mine development.

Based on previous research, we expected a 4–10-year time lag, before any demographic impact might be apparent (Green et al. 2017, Harju et al. 2010, Holloran et al. 2010, Walker et al. 2007). Our analysis of sage-grouse demography revealed that this local population had not yet responded to the mine disturbance, due to our immediate post-disturbance research and the short-term nature of our study. Therefore, we suggest that additional post-disturbance data collection is imperative to assess long-term disturbance (Fedy and Doherty 2011). We caution wildlife managers and land managers that our findings should be considered as baseline information for future management and mine activities. As such, our findings suggested there is a limited sage-grouse response to the mine does not imply that no impact occurred, but rather that it has not yet been realized, so care should be taken to enact mitigation that takes any potential impact into account for future large-scale disturbance projects. Since much of the literature documents sage-grouse response to disturbance in the long-term (i.e. Gregory and Beck 2014, Green et al. 2017, Harju et al. 2010), it is not surprising we did not find meaningful differences between years or between control and treatment leks based on daily attendance averages. Blickley et al. (2012) found that once anthropogenic sounds were removed from the lek area that sage-grouse returned to the area in a short time period. At a minimum, disturbance from excess noise or traffic could be limited during peak hours at the lek closest to the mine, but ideally could be limited throughout my study area since inter-lek movements were common in 45% of sage-grouse logged or monitored. By limiting as much disturbance as possible, sage-grouse would be less likely to be displaced and this could help ensure lek persistence (Holloran et al. 2010).

This extended abstract uses the pronouns we and our, but most of this abstract is extracted from Kircher (2020).

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Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Evaluating lek-based monitoring and management strategies for greater sage-grouse in the Parachute-Piceance-Roan population of northwestern Colorado

Period Covered: January 1 – December 31, 2020

Principal Investigator: Brett L. Walker brett.walker@state.co.us

Project Collaborators: Bill deVergie, Stephanie Durno, Brian Holmes, Dan Neubaum, Brad Petch, J.T. Romatzke

All information in this report is preliminary and subject to further evaluation. Information MAY NOT BE PUBLISHED OR QUOTED without permission of the author. Manipulation of these data beyond that contained in this report is discouraged.

ABSTRACT

Effective monitoring and mitigation strategies are crucial for conserving populations of sensitive wildlife species. Concern over the status of greater sage-grouse (*Centrocercus urophasianus*) populations has increased range-wide and in Colorado due to historical population declines, range contraction, continued loss and degradation of sagebrush habitat, and the potential for listing the species under the Endangered Species Act. Despite untested assumptions, lek-count data continue to be widely used as an index of abundance by state and federal agencies to monitor sage-grouse populations. Lek locations are also commonly used as a surrogate to identify and protect important sage-grouse habitat. However, the use of lek counts and lek locations to monitor populations is controversial because how closely lek-count data track actual changes in male abundance from year to year has never been tested. It is also unknown how effective lek buffers are at reducing disturbance to male sage-grouse and the habitats they use in each season. We deployed solar-powered GPS satellite transmitters on male greater sage-grouse to obtain data on male survival, lek attendance, inter-lek movements, and diurnal and nocturnal habitat use around leks and conducted double-observer lek counts to estimate detectability of males on leks during the breeding season in the Parachute-Piceance-Roan population in northwestern Colorado in spring from 2012-2016. I originally planned to use estimates of male survival, detectability of males on leks, lek attendance, inter-lek movement, and the proportion of leks known and counted during the breeding season to generate lek-count data from simulated male populations to evaluate the reliability of current lek-based methods for monitoring population trends. In conjunction with Jessica Shyvers and Jon Runge, I developed a multi-state model to simultaneously estimate daily survival, lek attendance, and inter-lek movement rates of males during the breeding season. That analysis is in progress, with Dr. Jessica Shyvers as a collaborator. I now anticipate submitting a publication on GPS male survival, lek attendance, and inter-lek movement in 2020. I am using local convex hull (t-Locoh) and Brownian bridge movement models to estimate space use in relation to leks to evaluate the performance of lek buffers for conserving important greater sage-grouse seasonal habitats. Male space use analyses are ongoing. Locations of GPS males have also been used to test for the influence of topography on overlap between energy infrastructure and sage-grouse use (Walker et al. 2020).

Walker, B. L., M. A. Neubaum, S. R. Goforth, and M. M. Flenner. 2020. Quantifying habitat loss and modification from recent expansion of energy infrastructure in an isolated, peripheral greater sage-grouse population. Journal of Environmental Management. 255:109819.

Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Hiawatha Regional Energy Development Project and greater sage-grouse conservation in northwestern Colorado and southwestern Wyoming
Phase I: Conservation planning maps and habitat selection

Period Covered: January 1 – December 31, 2020

Principal Investigator: Brett L. Walker brett.walker@state.co.us

Project Collaborators: B. Holmes, B. Petch, B. deVergie

All information in this report is preliminary and subject to further evaluation. Information MAY NOT BE PUBLISHED OR QUOTED without permission of the author. Manipulation of these data beyond that contained in this report is discouraged.

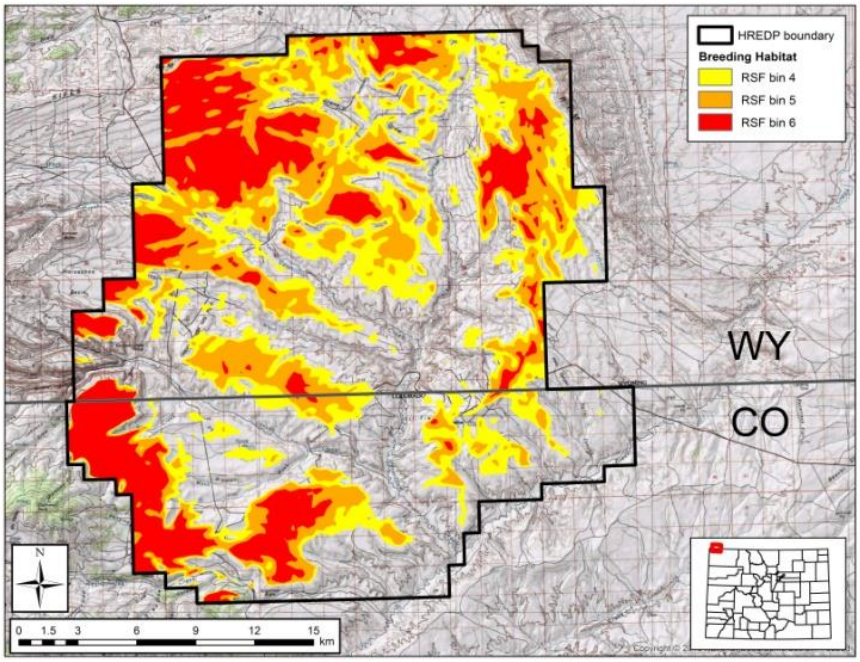
ABSTRACT

Increasing energy development within sagebrush habitat has led to concern for conservation of greater sage-grouse (*Centrocercus urophasianus*) populations, and both industry and regulatory agencies need better information on when and where sage-grouse occur to reduce impacts. Managers also lack landscape-scale habitat guidelines that identify the size and configuration of seasonal habitats required to support sage-grouse use. It is also essential to understand how sage-grouse in local populations select habitat in terms of the relative importance of local (i.e., micro-site) vs. landscape-scale habitat features. Understanding their response to different components of energy infrastructure is also essential for understanding and predicting the effects of specific development proposals. Resource selection functions (RSF) can be combined with geographic information system data to model habitat selection by sage-grouse in response to natural and anthropogenic habitat features at multiple scales and to map key seasonal habitats at high resolution over large areas. Multi-scale habitat use models, landscape-scale habitat guidelines, and high-resolution seasonal habitat-use maps will help streamline planning and mitigation for industry and facilitate sage-grouse conservation in areas with energy development. The proposed Hiawatha Regional Energy Development Project (HREDP) overlaps much of the known winter habitat and a portion of the documented nesting and brood-rearing habitat for the sage-grouse population that breeds in northwestern Colorado. Colorado Parks and Wildlife conducted a field study project tracking VHF females from December 2007 through July 2010. Objectives were to: (1) create validated, high-resolution conservation planning maps based on RSF models that delineate important seasonal sage-grouse habitats within the proposed HREDP boundary, (2) identify landscape-scale seasonal habitat guidelines, (3) evaluate the relative importance of local versus landscape-level habitat features (including vegetation, topography, and energy infrastructure) on sage-grouse wintering and (if possible) nesting habitat selection, and (4) assess whether historical energy development in the Hiawatha area influences current habitat selection. Field data collection was completed in July 2010. Preliminary seasonal RSF maps were completed in March 2010 (Fig. 1). However, analyses were limited by the extent of reliable classified land cover layers on either side of the Colorado-Wyoming state line. CPW's GIS section attempted to produce an improved classified land cover layer from 2010-2014, however, that effort was unsuccessful, so I opted to use the USGS Landfire vegetation layer instead. I completed mapping of annual energy infrastructure within 4 miles of the HREDP boundary from 2006-2015 in 2017. To meet objectives 1-3, I will first conduct RSF analyses and seasonal habitat mapping for the winter and breeding seasons using 2007-2010 VHF locations and micro-site vegetation data. Since field work for this project

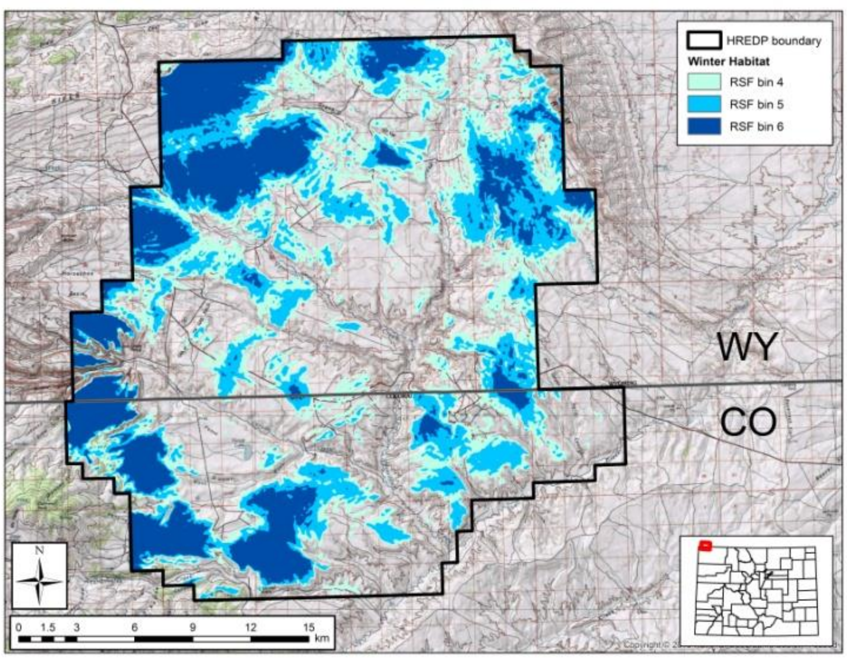
was completed, two additional, higher-resolution datasets have become available that would improve modeling of seasonal habitat. I plan to use two datasets of seasonal locations collected from GPS-marked females in 2009-2013 and GPS-marked males in 2012-2016 to conduct additional RSF analyses to assess habitat selection all three seasons in relation to vegetation cover, topography, and energy infrastructure to complement models based on VHF data. For objective 4, we found that historical and recent energy development within the HREDP were largely coincident (i.e., spatially correlated), so it would be impossible to distinguish the effects of historic vs. recent development on current habitat selection. So, to better assess the effect of historical well pads on likelihood of use by GRSG, we measured micro-site vegetation on abandoned and reclaimed well pads in summer 2010 for comparison against vegetation measured around well pads and around nests and wintering locations. Analyses for objectives 1-3 are ongoing, and analyses for objective 4 will be started after completion of other, higher priority projects.

Figure 1. Preliminary high-resolution (a) breeding and (b) winter habitat selection maps for greater sage-grouse in the Hiawatha Regional Energy Development project area based on vegetation and topography.

a) Breeding



b) Winter



Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Seasonal habitat mapping in the Parachute-Piceance-Roan region of western Colorado

Period Covered: January 1 – December 31, 2020

Principal Investigator: Brett L. Walker brett.walker@state.co.us

Project Collaborators: Brian Holmes, Darby Finley, Steph Durno, Brad Petch, Bill deVergie, J. T. Romatzke

ABSTRACT

Large-scale changes to sagebrush habitats throughout western North America have led to growing concern for conservation of greater sage-grouse (*Centrocercus urophasianus*) and repeated petitions to list the species under the Endangered Species Act. Greater sage-grouse in the Parachute-Piceance-Roan (PPR) region of western Colorado face two major conservation issues: a long-term decline in habitat suitability associated with pinyon-juniper (PJ) encroachment, and potential impacts from rapidly increasing energy development. In 2006, Colorado Parks and Wildlife (CPW) and industry partners initiated a 3-year study to obtain baseline data on greater sage-grouse in the PPR. Using those data, we published validated multi-scale, season-specific, resource selection function (RSF) models for the PPR based on vegetation cover and topography using primarily day-time locations of VHF-marked females (Walker et al. 2016). The second phase of the habitat selection study included examining the effects of energy infrastructure after controlling for topography, other changes to vegetation cover, and non-energy infrastructure. To meet this objective, we first mapped annual changes in four major components of energy infrastructure (well pads, facilities, pipelines, and roads), non-energy infrastructure (buildings, roads) and other landscape changes (e.g., habitat treatments, fires) from 2005-2015. Because of widespread interest in quantifying and predicting land cover changes associated with energy development from management agencies, I published a manuscript describing that mapping (Walker et al. 2020). I then analyzed VHF telemetry data from 2006-2014 to assess female habitat use and selection in relation to energy infrastructure (Walker, *in review*). The majority of females (92-97%) used seasonal areas with < 3% total anthropogenic infrastructure within 1000 m. Females avoided infrastructure with disturbed and reclaimed surface within 1000 m and locations with active energy feature densities of 1-2% during breeding and winter. Breeding females avoided locations near high-activity active energy features. In contrast, females selected locations with intermediate amounts of reclaimed surface (especially pipelines) and locations closer to roads and pipelines in summer-fall and showed no avoidance of locations in relation to road or active energy feature density. All components of infrastructure tested negatively influenced female habitat selection in at least one season and should be included in surface disturbance calculations within greater sage-grouse priority habitat management zones.

Walker, B. L., A. D. Apa, and K. Eichhoff. 2016. Mapping and prioritizing seasonal habitats for greater sage-grouse in northwestern Colorado. Journal of Wildlife Management 80:63-77.

Walker, B. L., S. R. Goforth, M. A. Neubaum, and M. M. Flenner. 2020. Quantifying habitat loss and modification from recent expansion of energy infrastructure in an isolated, peripheral greater sage-grouse population. Journal of Environmental Management 255:190819.

Walker, B. L. *In review*. Resource selection by greater sage-grouse varies by season and infrastructure type in a Colorado oil and gas field. Ecosphere.

Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Using seasonal and dispersal movements of greater sage-grouse to inform management for connectivity

Period Covered: January 1 – December 31, 2020

Principal Investigator: Brett L. Walker brett.walker@state.co.us

Project Collaborators: Brian Holmes, Liza Rossi, Michelle Cowardin

ABSTRACT

Wildlife often undertake long-distance movements, most commonly when migrating between seasonal ranges, when dispersing as juveniles or post-breeding adults, and when dispersing between populations. Conserving and managing habitat within movement corridors is critical for maintaining connectivity between seasonal habitats within populations, maintaining demographic and genetic connectivity between populations, and ensuring the long-term persistence of local and regional populations. Loss of connectivity is often a problem for small, isolated subpopulations at risk of low effective population size, loss of genetic diversity and adaptive potential, and increased inbreeding. Translocations from larger populations can prevent demographic and genetic problems caused by loss of connectivity, but proactive efforts to manage habitat in movement corridors between core and peripheral populations may be a more effective long-term conservation strategy. We are investigating habitat use and selection by greater sage-grouse during long-distance seasonal and dispersal movements to inform efforts to maintain connectivity among populations at the southern edge of the species' range. There are numerous unresolved questions about how greater sage-grouse make such movements in terms of timing, distance, stopovers, movement strategies, the influence of landscape context and topography on movement, and habitat use and selection during movements. Such information will be valuable for assess current linkage zones and informing management, conservation, and restoration within those areas. We are using existing GPS telemetry data from greater sage-grouse management and research projects across Colorado to conduct this investigation. We have compiled telemetry data from all CPW projects planning to contribute data, and we are currently writing code to analyze movements, habitat use, and habitat selection.

Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Effects of Esplanade herbicide at Bitterbrush State Wildlife Area

Period Covered: January 1 – December 31, 2020

Principal Investigators: Danielle B. Johnston daniell.bilyeu@state.co.us (Habitat Researcher, CPW), Trevor Balzer (Habitat Coordinator, CPW)

Project Collaborator: Colton Murray (Property Technician, Bitterbrush State Wildlife Area)

BACKGROUND

Annual species are known to hinder the establishment of bitterbrush seedlings (Hall et al. 1999). Invasion by annual species, in particular cheatgrass (*Bromus tectorum* L.) also increases fire frequency by increasing fine fuels and fuel continuity (Balch et al. 2013; Davies and Nafus 2013). Since 1976 fire has drastically changed the vegetation quality at Bitterbrush State Wildlife Area (SWA), which serves as important mule deer Winter Range, Severe Winter Range, and Critical Winter Range for the D-7 Data Analysis Unit. Eleven fires have burned a total of 2,200 ha (5,500 ac) within the property boundary (67% of its area) and 15,000 ha (37,000 ac) of similar habitat on adjacent property. Recovery of bitterbrush (*Purshia tridentata*) and other shrubs species has been slow. While there has been some recruitment of bitterbrush in areas which have not recently burned (Brian Holmes, *pers. comm.*), areas subjected to several burns over multiple years have little to no shrub recruitment occurring, and invasive annual species remain abundant in burned areas.

Recently, the herbicide indaziflam (trade name EsplAnade® 200 SC, Bayer Corp.) has been shown to provide long-term control of annual grasses, and, to a lesser extent, annual forbs (Sebastian et al. 2016; Sebastian et al. 2017). The herbicide is a cellulose biosynthesis inhibitor and provides a different mode of action than other commonly used herbicides for annual grass control. Recent trials near Boulder, Colorado have resulted in both reduced annual grass cover and increased leader growth on bitterbrush, mountain mahogany (*Cercocarpus montanus*), and fringed sagebrush (*Artemisia frigida, Derek Sebastian, pers. comm.*). However, effects on seedlings may differ from those on mature plants. Indaziflam inhibits root elongation, and may have detrimental effects on seedlings. Detrimental effects of the herbicide may be more than offset by reduced annual competition, but the net effect on seedlings is unknown.

We sought to understand how indaziflam effects mature bitterbrush and other desirable shrubs, quantify its annual grass control performance, and determine its effect on bitterbrush establishment from seed. We chose three study areas which had burned in the last 35 years, experienced low to moderate recovery, and had received no prior habitat treatments aside from seeding. Using prior monitoring data, we identified areas which have potential to show a response in bitterbrush density and/ or leader growth, given a reduction in annual grass competition. We used these criteria:

- At least trace bitterbrush present, OR seeded with bitterbrush within the last 5 years
- Perennial forb cover is less than 40% [perennial forbs hinder bitterbrush production (Cunningham 1971) and seedling survival (Mummey et al. 2018)]
- Dense bitterbrush stands were present at the site prior to fire

Nine plots approximately 25 m X 75 m and 0.2 ha (0.5 ac) in size were established at each site in April 2019. We followed the manufacturer's recommendation to combine indaziflam with glyphosate for a spring application, and compared indaziflam + glyphosate plots with glyphosate only and control plots. Treatments were assigned randomly (n = 3 per site). Colton Murray completed application on 22 April 2019. Indaziflam + glyphosate plots received 73.1 g ai/ha indaziflam (5 oz/ac of Esplanade 200SC which contains 1.7 lbs/gal of ai), 350 g ai/ha glyphosate (8.9 oz/ac of Roundup Power Max which contains 4.5 lbs/gal ai), 188 li/ha (20 gal/ac) of water, and 0.125% v/v non-ionic surfactant (Activator 90, Loveland Products). Glyphosate only plots received 350 g ai/ha glyphosate, 188 li/ha of water, and 0.125% v/v non-ionic surfactant. Control plots received 188 li/ha of clean water only.

Percent cover data was taken on each plot in June 2019 and June 2020. Shrub leader growth, density and winter forage productivity data were taken in August 2019 and October 2020. For details on methods, please see the 2019 annual report.

In October 2019, we established six 1m^2 subplots within each plot to assess herbicide effects on bitterbrush recruitment from seed. We planted bitterbrush seed in mimicked rodent caches, as nearly all bitterbrush seedlings grow from such caches (Vanderwall 1994). Nine evenly spaced caches were planted with10 hard, well-formed seeds 4 cm deep (Hall et al. 1999; Hammon and Noller 2004) using seed which had been collected in July 2018 from Bitterbrush SWA. Half of the subplots received a grazing cage to exclude large herbivores, as bitterbrush seedlings need about two years of protection from herbivory to become established (Paschke et al. 2003; Dyer and Noller 2014). We counted surviving seedlings within subplots in mid-May and late June, 2020. For details on methods, please see the 2019 annual report.

The seedling recruitment component as described in the 2019 study plan was considered a pilot component, because we knew little about the limiting factors on bitterbrush recruitment at Bitterbrush SWA at that time. Although bitterbrush is considered to be one of the easier high-value forage shrubs to establish from seed (Paschke et al. 2003), several factors have the potential to limit bitterbrush recruitment, including drought (Hammon and Noller 2004), annual weeds (Hall et al. 1999), perennial competition (Mummey et al. 2018), rodent cache pilfering (Dittel and Vander Wall 2018), and rodent seedling grazing (Vanderwall 1994). In spring 2020, we noted evidence that rodent cache pilfering and seedling grazing by rodents and/or ants were likely having a large effect on bitterbrush seedlings. Seed husks were found in nearly every subplot, at the location of most individual seed caches (Figures 1a-c). Rodents were the likely culprit of this activity. Although both ants and rodents are prevalent at the site and are known to harvest and consume bitterbrush seeds (Kelrick et al. 1986), only rodents are known to dig for buried seeds (MacMahon et al. 2000). We also observed seedling herbivory (Figures 1c-d) which may have been due to rodents, insects, or both. Rodents are known to consume the entire seedlings (Clements and Young 1996), but we sometimes noted small bites which may have been due to ants or other insects (Figure 1d). Our original study plan included a provision for additional seedling studies if needed to unravel unforeseen influences on seedling recruitment. We added two such studies in 2020: one focused on isolating the influence of cheatgrass control on bitterbrush recruitment, and one focused on the relative importance of cheatgrass control, ants, and rodents on bitterbrush recruitment. Details are included below.

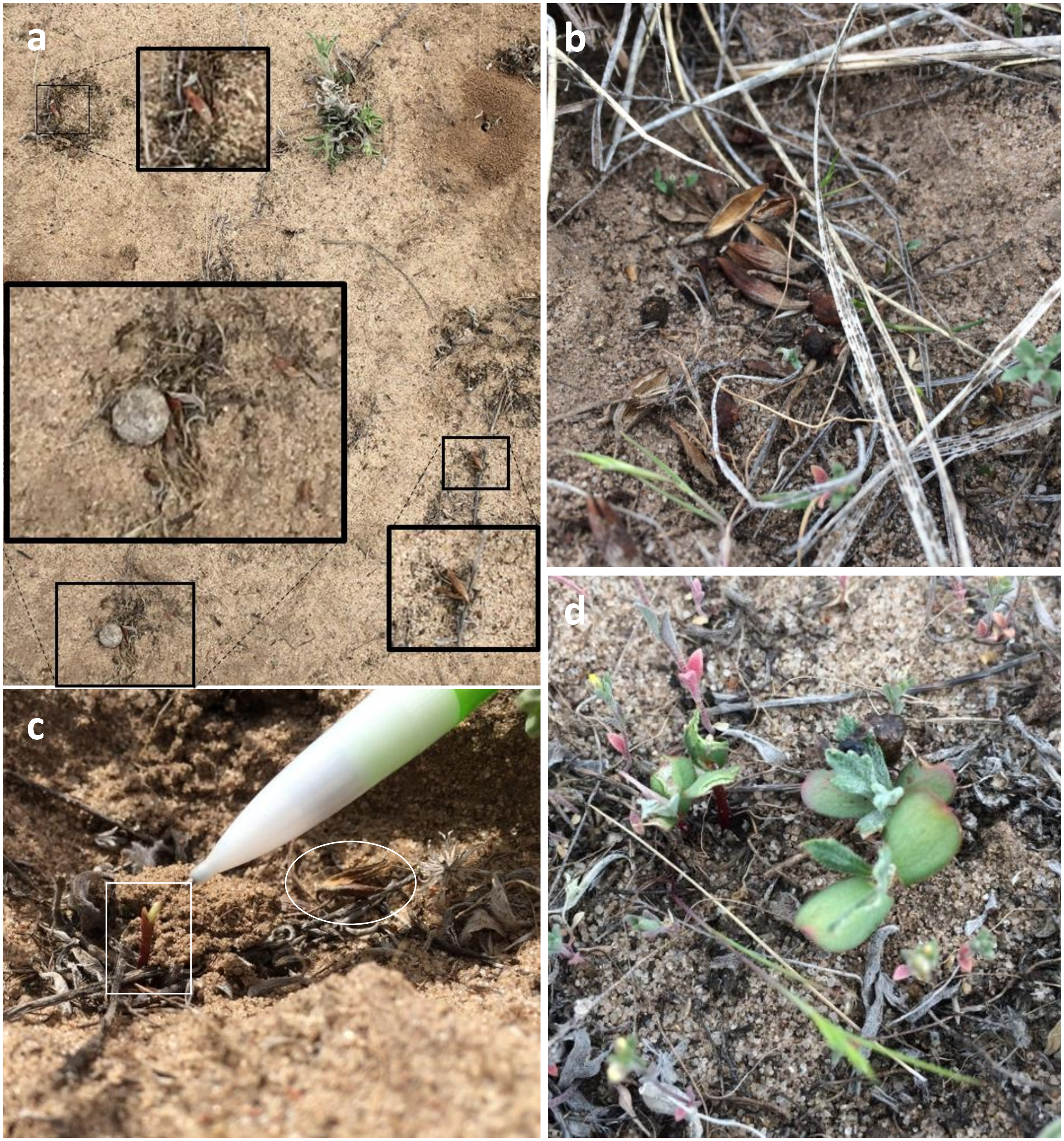


Figure 1. Bitterbrush seed predation and seedling herbivory in experimental subplots: a) Aerial view of seedling subplot, with insets highlighting husks of bitterbrush seeds; b) Close-up of bitterbrush seed husks found atop where a seed cache had been buried; c) a bitterbrush seedling with all cotyledons and leaves browsed off (box) and a bitterbrush seed husk (oval); d) bitterbrush seedlings with evidence of insect browsing on cotyledons.

2020 ACTIVITIES: RESULTS AND NEW STUDIES

Species composition. Plant cover in control plots in 2019 was dominated by annuals such as desert alyssum and cheatgrass, as well as by unpalatable forbs such as hairy golden aster and death camus (Figure 2). In 2020, cover of perennial plants was similar to 2019, but annual cover was much lower.

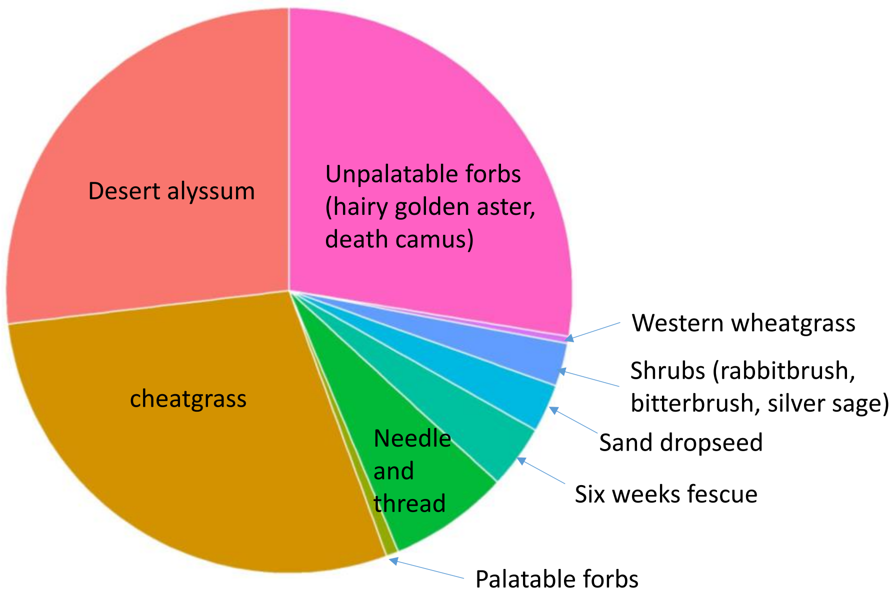


Figure 2. Control plot species composition in 2019. This graph represents relative plant cover. The total of cover for all groups was 86%; in absolute terms, cover values for all groups is slightly lower than what is depicted on this graph. In 2020, cover values for perennials was similar to 2019, but cheatgrass cover dropped from 25% to 10% and desert alyssum cover dropped from 23% to 4%.

Herbicide effects. For percent cover, we analyzed differences between control, glyphosate, and indaziflam + glyphosate plots using a negative bionomial mixed model on number of hits per transect, with year, herbicide treatment, and their interaction as fixed effects, site as a random effect, and plot as a random effect (to account for repeated measures on plots). We used the glmmTMB package in R. The negative bionomial was chosen because it is appropriate for overdispersed count data, and because it had a much lower AIC than Gaussian or poisson distributions for all plant groups.

Cheatgrass cover was lower with both glyphosate and indaziflam + glyphosate treatments in 2019 (Figure 3). In 2020, however, only the indaziflam + glyphosate treatment reduced cheatgrass, bringing it from about 10% in control plots to nearly 0% in indaziflam + glyphosate plots. Desert alyssum was controlled by only the indaziflam + glyphosate treatment in both 2019 and 2020. In 2019, indaziflam + glyphosate reduced desert alyssum cover from 23% to 9%; in 2020 the reduction was from 4% to 0%. Six weeks fescue cover was increased by the glyphosate treatment in 2019, from 3% to 6%. We detected no other significant effects for any of the other dominant plants or plant functional groups.

Shrub leader length data was normally distributed, but the analysis was complicated by the fact that shrub density was patchy. From a cursory inspection of the 2020 data it was obvious that shrubs in dense plots had shorter leader lengths, so species density was included in the models. We had sufficient data to analyze two species: bitterbrush (present at all three sites) and silver sage (present at one site only). We analyzed differences between control, glyphosate, and indaziflam + glyphosate plots using a normal mixed model on plot-averaged values of longest leader per plant, with year, herbicide treatment, species density, and their interactions as fixed effects, site as a random effect (for bitterbrush only), and plot as a random effect (to account for repeated measures on plots).

For bitterbrush leader length, there was no effect of herbicide treatment nor any significant interactions involving herbicide treatment. There was an effect of year, with longer leaders in 2019 (26.3 ± 0.9 cm) than in 2020 (21.9 ± 0.8 cm). In addition there was a highly significant interaction between bitterbrush density and year, as plots with dense patches of bitterbrush (200 to 600 plants/ha) had longest leader values around only 15 cm in 2020 (Figure 4a).

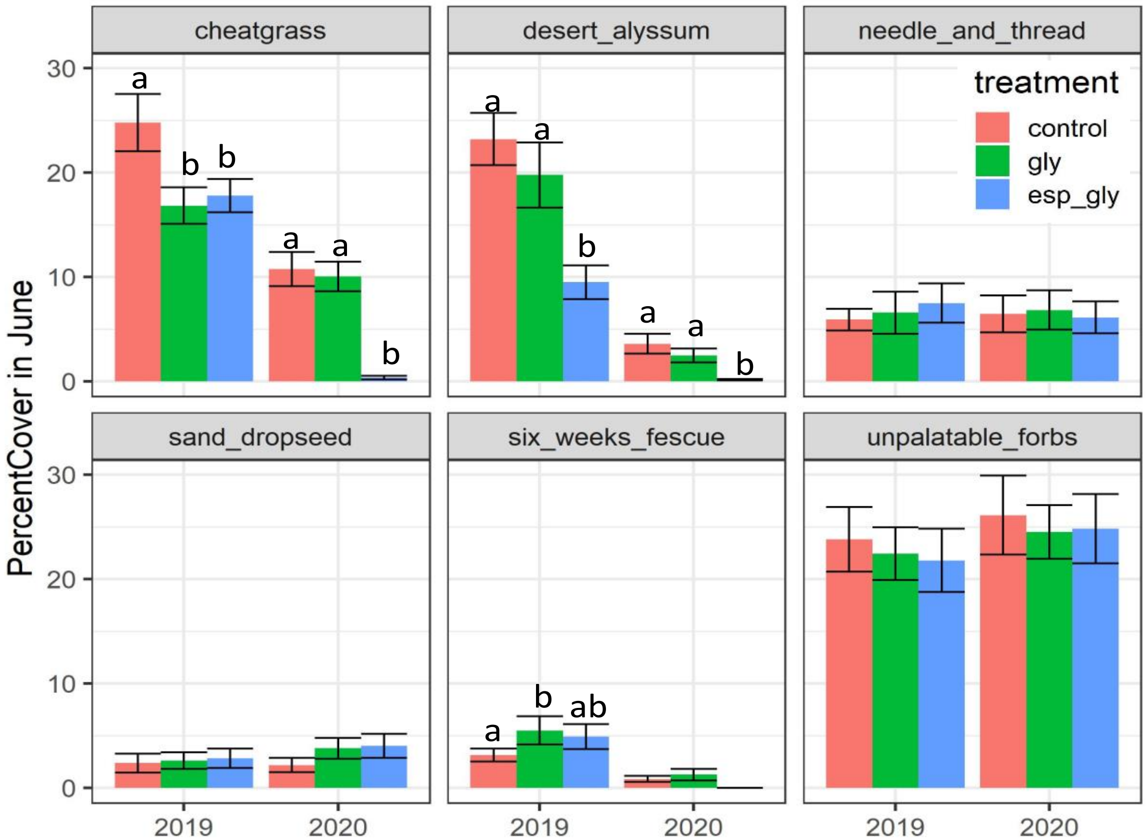


Figure 3. Effects of glyphosate (gly, green bars) and indaziflam + glyphosate (esp_gly, blue bars) on percent cover of dominant plants and functional groups. Letters indicate significantly different means for treatments within a year (alpha = 0.05). Errors bars are SE over nine plots (three at each of three sites).

For silver sage leader length, there was no effect of herbicide treatment nor any significant interactions involving herbicide treatment. There was an effect of year, with longer leaders in 2019 (27.5 ± 1.7 cm) than in 2020 (21.5 ± 0.8 cm). There was no interaction between year and species density, but there was a main effect of density, with shorter leader lengths in denser plots across years (Figure 4b).

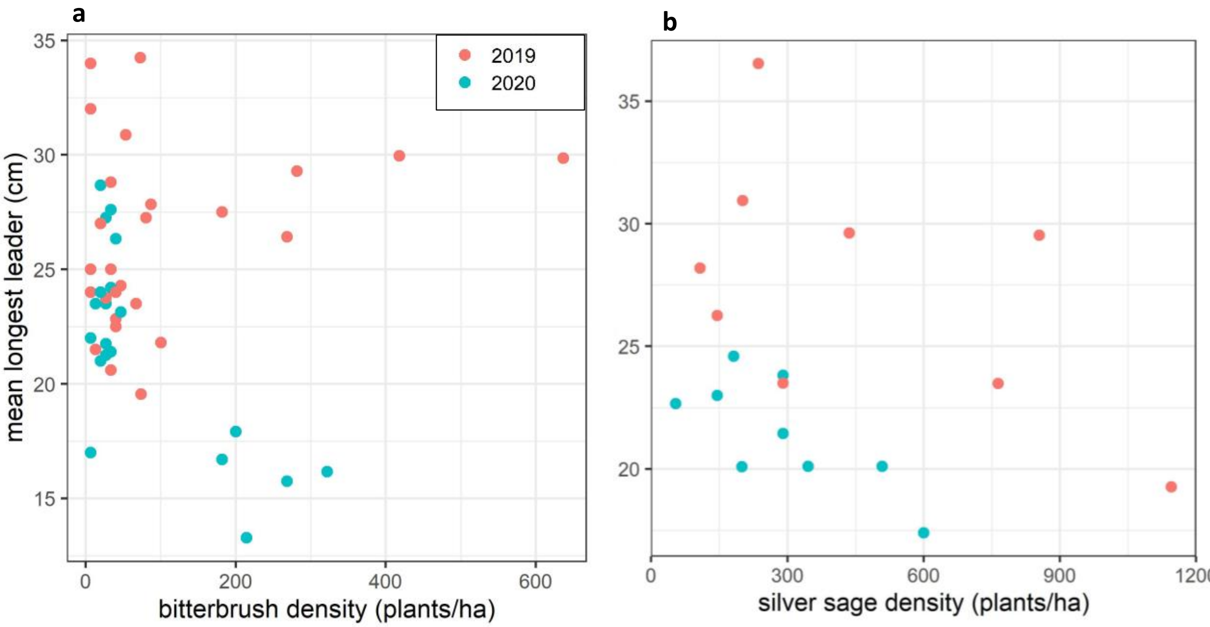


Figure 4. Relationship between plant density and plot-averaged values of longest leader per plant for a) bitterbrush and b) silver sage.

Winter forage biomass. We used an equation to estimate the winter-available forage biomass between 2019 and 2020, the years for which we saw significant changes in leader length. Winter forage biomass is the sum of current-year shoot dry weight, excluding leaves. Our equation was based on bitterbrush plants sampled in the Piceance Basin as well as 10 plants sampled each year 2019-2020 in the Maybell area. The R^2 of the relationship is 0.70, meaning that we can predict a bitterbrush plant's biomass of winter-available forage with 70% accuracy from measurements of its canopy size and longest leader length. We estimate that in our study plots, winter forage biomass fell from 3.5 kg/ha (3.1 lbs/ac) in 2019 to 1.6 kg/ha (1.4 lbs/ac) in 2020. This was due to a combination of shorter leaders, smaller canopy size (1.8 m^2 in 2019 vs 1.2 m^2 in 2020), and fewer plants (276 in 2019 vs 202 in 2020). Many dead plants were noted in fall 2020.

Seedlings. To analyze bitterbrush seedling response we used a negative binomial generalized linear mixed model with fixed effects of herbicide treatment (control, glyphosate, and indaziflam + glyphosate), cage (caged or uncaged), date (May 12, 2020 or June 30, 2020) , and their interactions as well as random effects of site, plot (to account for the split plot design), and subplot (to account for repeated measures on subplots). We used the glmmTMB package in R. We tested poisson, zero inflated poisson, negative binomial, Gaussian, and zero inflated Gaussian models. The poisson model had the lowest AIC, but as the data were slightly overdispersed and the negative binomial had a delta AIC of only 2, the negative bionomial was chosen as being most appropriate.

No effects were highly significant, but an interaction between cage and date ($p = 0.05$), herbicide and date ($p = 0.07$) and a main effect of herbicide ($p = 0.09$) were marginally significant. When testing contrasts of herbicide or cage within dates, no contrasts were significant at alpha = 0.05. However, there was a marginally significant effect of herbicide on May 12, with the indaziflam + glyphosate having less seedlings than the glyphosate treatment ($p = 0.06$, Figure 5).

Very few live seedlings were counted overall. Seedling density averaged less than 1 per subplot on both dates, though 90 seeds were planted in each subplot.

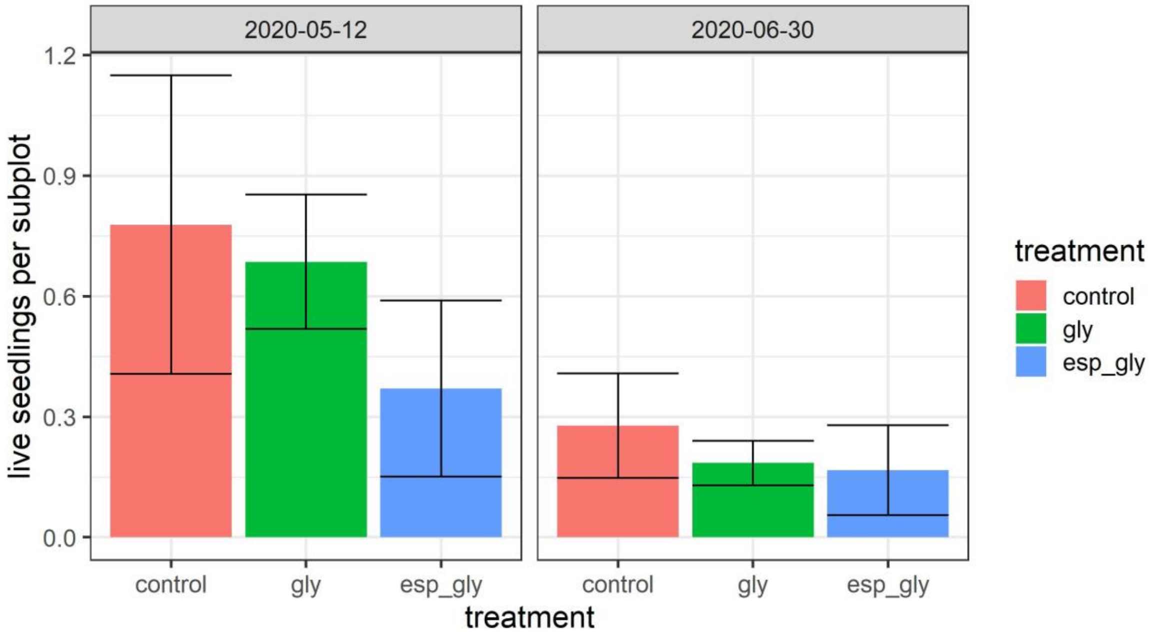


Figure 5. Effects of glyphosate (gly, green bars) and indaziflam + glyphosate (esp_gly, blue bars) on number of live bitterbrush seedlings per subplot (90 seeds were planted per subplot). The difference between glyphosate and indaziflam + glyphosate on May 12 was marginally significant ($p = 0.06$). Errors bars are SE over nine plots (three at each of three sites; seedling subplots were averaged over each plot before computing error bars).

"Red" experiment: isolating the effect of cheatgrass control. Though we saw a marginally significant effect of herbicide in our original seedling study, there was too much variability to achieve conclusive results. As it was obvious that rodents and/or insects were having an effect on seeds and seedlings (Figure 6), we sought to do a follow-up experiment with rodent-proof cages (Figure 7) using a previously successful cage design (Lucero and Callaway 2018). These were applied to every planting location. We also plan to apply an insecticide around each cage in the spring to eliminate variability due to insect herbivory. We are studying two cheatgrass control methods. We included indaziflam, applied at the same rate as the original study. We did not include a glyphosate treatment, because we applied the indaziflam prior to cheatgrass germination in the fall. We did include a treatment of a new product for cheatgrass control called NutraFix. NutraFix is a micronutrient fertilizer which is showing promise for controlling cheatgrass without injury to established perennial plants. Although the mechanism is not understood at this time, the product is high in boron, a micronutrient which can be toxic at high enough concentrations. It appears that there may be a sweet spot of NutraFix concentration which is toxic to cheatgrass but beneficial to other plants. Application rate trials are currently be conducted by CPW and include a study site at Bitterbrush State Wildlife Area (Johnston 2020). However, information about how NutraFix impacts desirable plants at the germination phase is also needed. Indaziflam and NutraFix effects on germination will be quantified for locally-collected bitterbrush and for Pueblo bottlebrush squirreltail (*Elymus elymoides*; Figure 6).

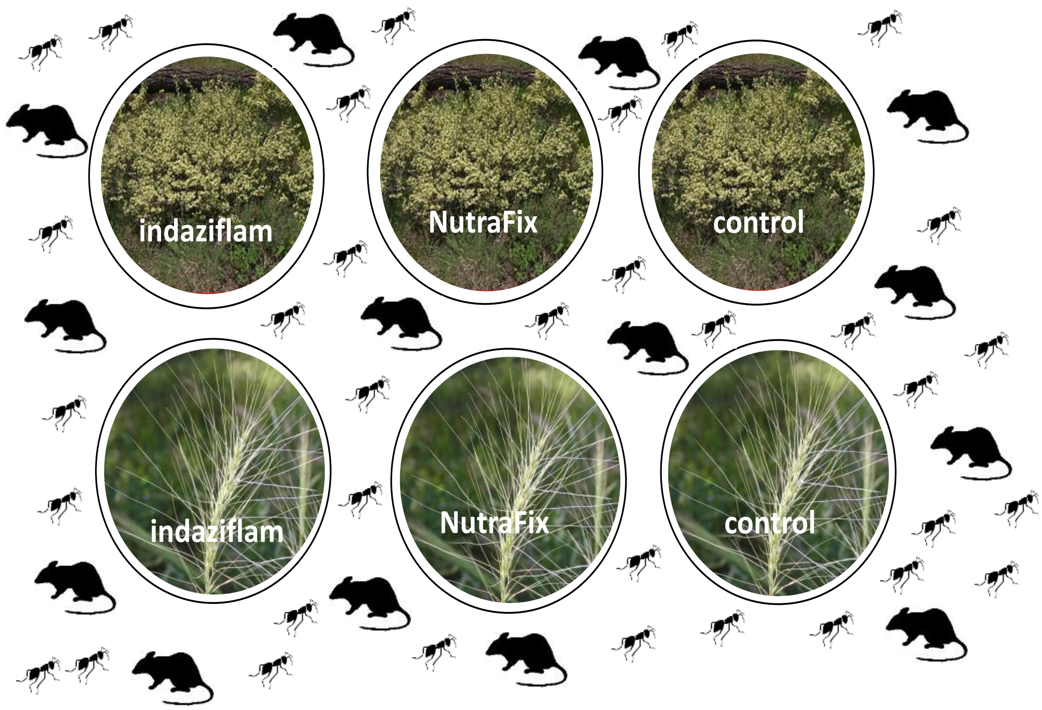


Figure 6. Schematic diagram of one replicate of the Red experiment. Ants and rodents will be excluded in all plots, and indaziflam, NutraFix, and control will be compared for bitterbrush and squirreltail seedlings.

Cheatgrass percent cover in control plots of the original experiment was only about 10% in 2020, compared to about 25% in 2019 (Figure 3). Since 2020 was not a 'good' year for cheatgrass, and since installing the rodent exclusion cages requires a slight ground disturbance which might temporarily reduce local cheatgrass density (Figure 7b), we were concerned that cheatgrass density might not be high enough in winter 20-21 to adequately test the cheatgrass control methods. Furthermore, we were interested in both the immediate effect of cheatgrass control methods on germination as well as their effect when there is a lag between application and planting. For these reasons, we are studying two planting years, 2020 and 2021.

Our study design resulted in 60 cages at each of 3 sites (2 planting years × 3 cheatgrass control treatments × 2 species × 5 replicates), and the design was completely randomized over all 60 cages. We used areas adjacent to the original study at each site, for ease of extending the hotwire to prevent cattle intrusion. (The hotwire will only be erected mid-April to late May, when cattle are present at the site.) We selected locations for cages which are at least 2.5 m apart and at least 2 m from ant nests, and within 1 m of evident cheatgrass. We installed cages and applied cheatgrass control treatments the week of 9/14. We applied 12.5 g of granulated NutraFix by hand to a 0.4 m^2 area with the cage at the center, for an application rate of 340 kg/ha (300 lbs/ac). We applied indaziflam using a backpack sprayer to a 0.4 m^2

area with the cage at the center. By timing our spray we were able to accurately deliver a rate of 73.1 g ai/ha indaziflam (5 oz/ac Esplanade) with 0.25% v/v non-ionic surfactant.

Cages selected for 2020 planting were planted the week of 10/7. Bitterbrush seeds were planted in 3 simulated rodent caches per cage, separated by about 15 cm and each containing 12 hard, well-formed bitterbrush seeds for a total of 36 bitterbrush seeds per cage. Seeds were buried 2 cm deep and were located above the floor of the rodent exclusion cage, which was buried 4 cm in the ground. Bitterbrush seed had been collected in September 2020 from Bitterbrush SWA. Squirreltail seeds were planted in simulated drill seeder rows. Four parallel furrows, 0.7 cm deep, were scratched within each

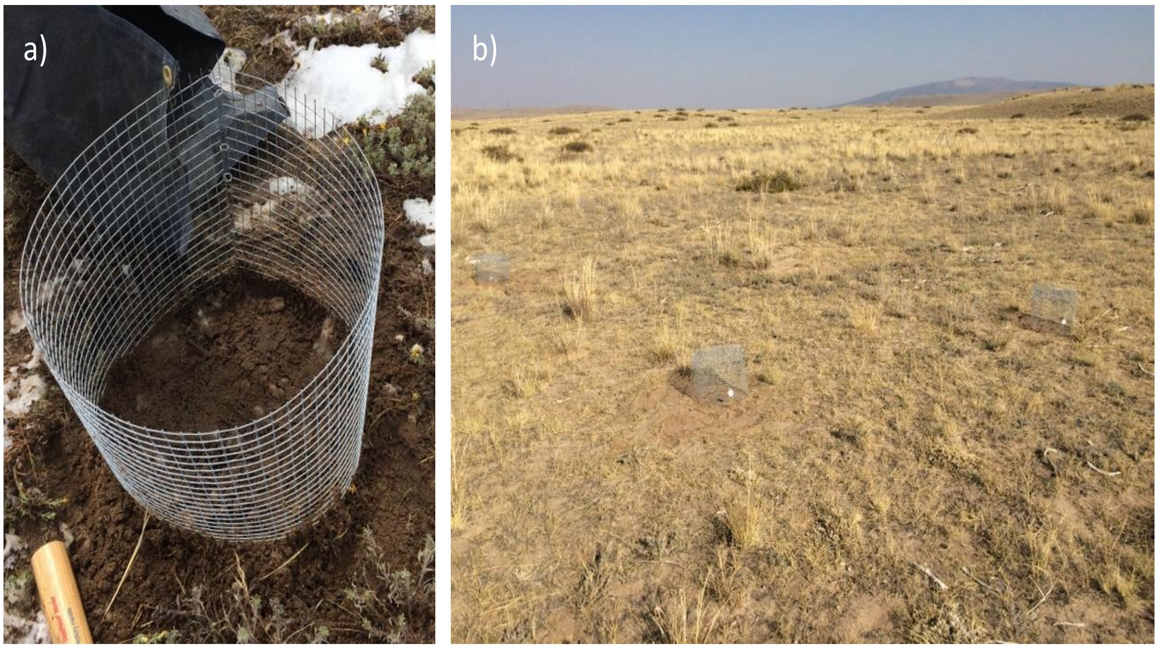


Figure 7. A) a partially built rodent exclusion cage. The cage has a floor buried 4 cm in the ground. After applying seed and treatments, a lid is added and secured with hog rings. B) completed cages.

cage. The two central furrows were slightly longer and accommodated 12 seeds each, while the two shorter, outer furrows accommodated 8 seeds each, for a total of 40 squirreltail seeds planted per cage. A lid of hardware cloth was added to the top of each cage and secured with hog rings. All cages in the Red experiment will be treated with insecticide beginning in mid-March, 2021, to discourage insect herbivory on seedlings. Controlling ants prior to germination is not necessary, as all seeds were buried and ants do not dig for seeds (Davidson et al. 1984; MacMahon et al. 2000). We are still in the process of selecting the appropriate herbicide, but products used in similar prior experiments include permethrin powder (Mills et al. 2018) and thiamethoxam (Bagchi et al. 2014). Either of these products would likely require reapplication to ensure effectiveness over an appropriately long period. Bitterbrush germination in our study area can occur through late April (Hammon and Noller 2004), and seedlings remain in the cotyledon stage, which is particularly attractive to herbivores, for about 10 days (Clements and Young 1996). Therefore, we plan to repeat application as necessary to discourage herbivory through mid-May.

Cages planted in 2020 will be monitored for seedling density 3-4 times in 2021, and at least once in 2022. Cages planted in 2021 will be monitored 3-4 times in 2022 and at least once in 2023 (Table 1). Data analysis will likely include a negative binomial generalized linear mixed model with fixed effects of treatment (control, indaziflam, or NutraFix), date, and their interactions as well as random effects of site and plot (to account for repeated measures on plots). Bitterbrush and squirreltail will be analyzed separately.

"Yellow" experiment: assessing relative impacts of rodents, ants, and cheatgrass control on bitterbrush seedlings. While the red experiment will provide useful information about the impact of cheatgrass control on seedlings in the *absence* of impacts from ants and rodents, we also need to know something about the *relative* impact of cheatgrass, ants, and rodents to inform management decisions such as treatment location, treatment scale, and complementary treatments. In a Great Basin study, rodents had about equal impact on seedling survival to cheatgrass competition (Lucero and Callaway 2018). In the Yellow experiment, we apply rodent exclusion, insecticide, and cheatgrass control in a crossed design (Figure 8). This will allow us to quantify the effect of each of these factors, both in isolation and when combined with each other, at Bitterbrush SWA.

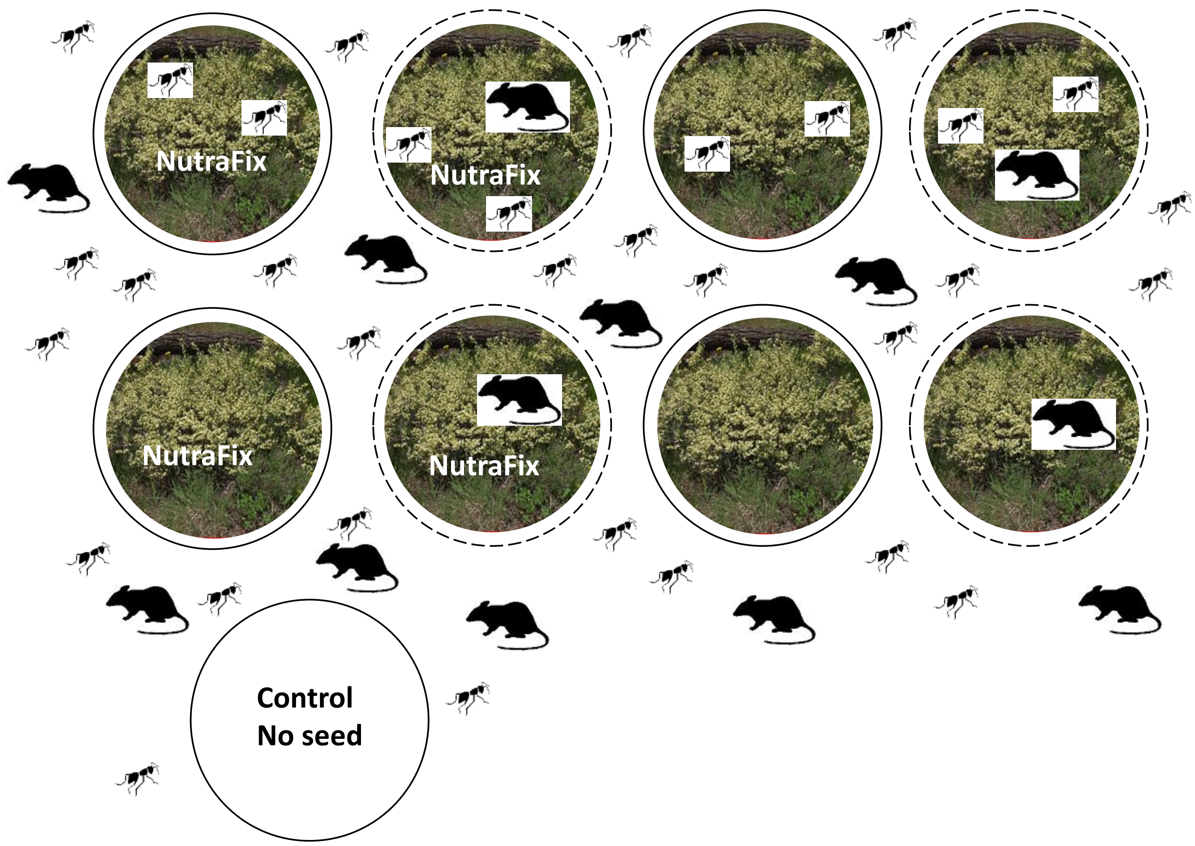


Figure 8. Schematic diagram of one replicate of the Yellow experiment. Ant exclusion, rodent exclusion, and cheatgrass control via NutraFix were crossed. All plots were planted with bitterbrush except for an ant-excluded, rodent-excluded control.

The design includes two levels of rodent exclusion (excluded or not excluded), two levels of ant exclusion (insecticide or no insecticide), and two levels of cheatgrass exclusion (NutraFix or no NutraFix), plus an unseeded control (with both ant and rodent exclusion) for a total of 9 treatments per replicate (Figure 8). Similar to the Red experiment, we have 5 replicates in each of 2 planting years (2020 and 2021) at each of 3 study sites. Unlike the Yellow experiment, the design was implemented in blocks. This was because rodent and ant densities are likely to vary spatially. Cages were spaced by 2m and were at least 2.5 m from ant nests. Blocking also allowed us to spread out replicates over a larger area. Rodent foraging can be inflated in areas of more dense seed sources (Lobo et al. 2013), and this can skew assessments of rodent impacts if artificially placed seeds exceed the range of densities present in an ecosystem (Perez et al. 2006).

Our study represented reasonable densities of food sources because the spacing of our cages and blocks ensured that simulated cache density did not exceed 1 cache/m^2, and bitterbrush cache densities of 1-4 caches/m^2 have been reported for scatter-hoarding rodents (Vanderwall 1994).

To create cages allowing rodent access, we cut a 12 cm X 12 cm access hole at ground level. Cages with NutraFix received 12.5 g of granulated NutraFix applied by hand to a 0.4 m^2 area with the cage at the center, for an application rate of 340 kg/ha (300 lbs/ac). Cages with insecticide will be treated with an appropriate herbicide from mid-March to mid-May, as describe in the section on the Red experiment. Planting and monitoring is also as described for the Red experiment.

Data analysis will likely include a negative binomial generalized linear mixed model with fixed effects of NutraFix treatment, rodent exclusion, insecticide, date, and their interactions as well as random effects of site, block, and plot (to account for repeated measures on plots). The number of interactions in this model may make convergence difficult; if so, we may choose to analyze either the date with maximum seedling density or the final measurement date in a simpler analysis without repeated measures. By quantifying the effect sizes of each significant treatment and interaction, we will gain insight into which of these factors is most limiting for bitterbrush recruitment. Unseeded controls will be compared to seeded, rodent-excluded, insecticide plots in a separate analysis to inform our knowledge of recruitment from the seed bank in the absence of seeding, rodent pilfering, or insect herbivory.

We also implemented a small side-experiment, a 'cage disturbance study', to get a better understanding of how the disturbance of installing the cages affects short-term seedling emergence from the seed bank, under conditions of rodent and ant access. At each of the 3 sites on 10/14, we selected 14 locations and assigned them to receive no cage or a non-functional cage. Neither type of location was seeded. We will monitor these for seedling emergence on the same schedule as the Red experiment. We predict that disturbance will have little long-term effect, because similar rodent exclusion cages were found to have little impact on wind-mediated seed movements in a prior study (Anderson and MacMahon 2001).

DISCUSSION AND FUTURE WORK

In our original study, the indaziflam + glyphosate treatment performed similarly to the glyphosate treatment in 2019, but indaziflam + glyphosate outperformed glyphosate greatly in 2020. In this discussion, 'indaziflam' will refer to the performance of the indaziflam + glyphosate treatment.

Indaziflam was effective at controlling cheatgrass. We did not note any injury to desirable plants nor any decrease in cover of desirable plants due to indaziflam. On the other hand, indaziflam caused a marginally significant decrease in bitterbrush seedling density, and did not bring about any detectible benefits to desirable plants. In prior work, increased bitterbrush leader growth was noted after indaziflam treatment (*Derek Sebastian, pers. comm.*), presumably due to decreased competition from cheatgrass. In this study, we observed that bitterbrush leader length was sensitive to the different growing conditions between 2019 and 2020, and also sensitive to bitterbrush plant density. Even so, we saw no effect of indaziflam on bitterbrush leader length. Overall cheatgrass density was low in 2020, and it is possible that indaziflam will benefit bitterbrush in a year with higher cheatgrass density. It is also possible that the extremely sandy soils in Bitterbrush SWA allow soil moisture to percolate so deeply and quickly that deeply-rooted bitterbrush plants are not greatly impacted by cheatgrass near-surface soil moisture use.

Even if cheatgrass is not substantially competing with bitterbrush for soil moisture at Bitterbrush SWA, cheatgrass may yet be the driving force behind low bitterbrush density through indirect effects. The effect of cheatgrass on the fire cycle is well-known, and it is possible that the fire return interval at Bitterbrush SWA in the presence of cheatgrass is simply too short to allow bitterbrush to fully recover. Cheatgrass may also be influencing the granivore communities in ways that are undesirable for bitterbrush establishment. At a site in Tooele County, Utah, not far from Bitterbrush SWA, cheatgrass invasion caused a 10-fold increase in ant density (Ostoja et al. 2009). Cheatgrass invasion can also either negatively (Ostoja and Schupp 2009) or positively (Richardson et al. 2013) impact rodents. As granivore activity can have impacts on seedlings of similar magnitude to cheatgrass (Lucero and Callaway 2018), any changes in the granivore community could result in an important limiting factor on bitterbrush recruitment.

The original, Red, and Yellow experiments will tell us much about how cheatgrass, rodents, and ants may limit germination once the seed is in the ground, which is relevant to seeding efforts and to understanding what happens after rodent caching. Final reporting on this work is expected winter 2023-4.

In natural settings, there are also potential bottlenecks on bitterbrush seedling recruitment which occur before seed is buried. Seed may be consumed by rodents or birds before or after falling to the ground, or may be taken by ants (Vanderwall 1994). Both ants and rodents can act as consumers and/or dispersers of seed, but rodents are more likely to bury bitterbrush seeds at a depth beneficial for germination. Understanding the rate of seed removal by rodents versus ants, and what percentage of those seeds are eaten versus dispersed, could help elucidate what management actions would be most effective for aiding bitterbrush recruitment. Addressing these questions is beyond the scope of this study plan, but may be considered for a study plan amendment.

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Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Pothole seeder demonstration studies

Period Covered: January 1 – December 31, 2020

Principal Investigator: Danielle B. Johnston danielle.bilyeu@state.co.us (Habitat Researcher, CPW)

Project Collaborators: Trevor Balzer (Habitat Coordinator, CPW), Jim Garner (Habitat Coordinator, CPW), Ivan Archer (Assistant Area Wildlife Manager, CPW), Derek Lovoi (Property Technician, CPW), Mark Hodges (Property Technician, CPW) , Kevin Gunnell (Great Basin Research Center Coordinator, Utah Division of Wildlife Resources), Melissa Landeen (Great Basin Research Center Project Leader, Utah Division of Wildlife Resources), Todd Graham (Ranch Advisory Partners), Jon Moore (Ranch Manager, Mountain Island Ranch), Mary Conover (Owner, Mountain Island Ranch), Ken Holsinger (Ecologist, Bureau of Land Management)

EXTENDED ABSTRACT

A new type of seeder, dubbed a pothole seeder, was designed and built after CPW research indicated that seeding over a roughened surface of mounds and holes could aid in cheatgrass (*Bromus tectorum*) control during plant establishment. Four of the first projects to use the seeder were established 2018-20. All of the projects have dual management and research goals, and some incorporate complementary treatments and/or comparisons to other treatments. All include untreated control plots. Conditions at all study sites in 2020 were extremely dry throughout the course of the growing season. Formal monitoring and data analysis was conducted at only one site in 2020.

The Escalante State Wildlife Area site receives about 213 mm (8 in) of annual precipitation and has saline and patchily hardpan soils. A project comparing pothole seeding to drill seeding was established in fall 2018. Photo point monitoring was conducted in May 2020. Although good plant establishment had occurred in portions of both drill-seeded and pothole plots in 2019, by 2020 many plants had died. Survival of plants appeared to be somewhat better in pothole plots. No effect of potholing on cheatgrass was evident. Overall cheatgrass density was very low.

The Nash Wash site is in Grand County, Utah and receives about 260 mm (10.5 in) of annual precipitation. A project comparing pothole seeding to drill seeding was established in fall 2019. Because of heavy cheatgrass cover, imazapic herbicide at a rate of 70 g ai/ha (4 oz/ac Plateau™) was applied prior to seeding. Informal photo monitoring in June 2020 indicated that potholing plus imazapic was likely providing better cheatgrass control than drill seeding plus imazapic, although it is not clear that control will be sufficient to allow perennial plant establishment. Very little establishment of seeded species was noted in any plots.

The Mountain Island Ranch site in Mesa County receives 360 mm (14.2 in) of annual precipitation and has very sandy soils. A project was initiated in fall 2019. Due to heavy cover of cheatgrass and other undesirable annuals, weed control was applied to all plots prior to seeding, via either imazapic at 70 g ai/ha (4 oz/ac Plateau™) or NutraFix, a cheatgrass control fertilizer, at 373 kg/ha (333 lbs/ac). Weed control method was crossed with seeding method (pothole or drill) in a split-plot design. Plant cover was assessed in May 2020. Very little germination of any seeded species was noted. Cover

of the undesirable non-native annual forb burr buttercup (*Ceratocephala testiculata*) was lower in pothole plots than in drill seeded plots ($p = 0.0009$), but seeding method had no effect on cheatgrass or bulbous bluegrass. The imazapic treatment was not effective on cheatgrass, bulbous bluegrass (*Poa bulbosa*), or burr buttercup and had only a minor effect at reducing cover of Russian thistle (*Salsola tragus*). NutraFix was highly effective at controlling cheatgrass, bulbous bluegrass, and burr buttercup ($p < 0.04$). NutraFix plots were almost entirely dominated by Russian thistle. While Russian thistle is not a desirable plant, it often yields relatively quickly to more desirable vegetation.

The Sims Mesa site in Montrose County receives about 280 mm (11 in) of annual precipitation with coarse, gravelly soils overlying a clay loam layer. The project site is a former plow-and-seed treatment which had been seeded with crested wheatgrass (*Agropyron cristatum*). To rehabilitate this potential Gunnison sage-grouse habitat, the Bureau of Land Management applied several herbicide treatments which successfully killed most of the crested wheatgrass. Pothole seeding was initated in December 2020, and about 7 of the planned 24 ha (18 of 60 ac) were seeded before a spinning disc and broken weld on the potholer forced us to suspend the project. We are currently evaluating the failure and making plans to improve the potholer.

Although we faced challenges of dry conditions, covid-related personnel shortages, and mechanical failure, we continued to learn in 2020. Prior research had indicated that in heavily infested areas, potholing is only a useful tool for cheatgrass control when coupled with an effective herbicide treatment. The patterns observed here are consistent with this idea. We noted anecdotal evidence that potholing improved plant survival in some cases. We plan to complete the Sims Mesa site and conduct formal monitoring at Escalante, Nash Wash, and Mountain Island Ranch in 2021.

Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Evaluating spatial patterns and processes of avian and mammalian wildlife populations

Period Covered: January 1 – December 31, 2020

Principal Investigator: Kevin Aagaard kevin.aagaard@state.co.us

Project Collaborators: Jim Gammonley, Reesa Conrey, Dan Neubaum, Megan Kocina, Tony Apa, Eric Bergman, Chuck Anderson, Andy Holland (Colorado Parks and Wildlife); Mindy Rice, Lief Wiechman (U.S. Fish and Wildlife service); Cameron Aldridge, Julie Heinrichs, Michael Hooper, Mike O'Donnell, Sarah Oyler-McCance, Wayne Thogmartin, Ben West, Mark Wildhaber (USGS); Carolyn Gunn

All information in this report is preliminary and subject to further evaluation. Information MAY NOT BE PUBLISHED OR QUOTED without permission of the principal investigator. Manipulation of these data beyond that contained in this report is discouraged.

EXTENDED ABSTRACT

Evaluating wildlife location data provides substantial information for management. Location data reveal patterns of movement dynamics, species distribution (habitat suitability), and varying habitat use. Understanding these patterns and dynamics is critical for endangered and harvested species. Colorado Parks and Wildlife monitors myriad species of concern for conservation and hunting and thus needs to develop thorough and up-to-date assessments of the spatial patterns and processes of its target species. In collaboration with state wildlife biologists, avian researchers, big game managers, and federal counterparts, I have assisted in evaluating spatial data for several species and populations. Below, I list the active research projects I am associated with, and briefly detail the objectives and current status of each.

- **Raptor Nesting Distribution Model (with R. Y. Conrey and J. Gammonley)** — We used nesting location data to assess suitable nesting habitat for four raptor species in Colorado (golden eagle, bald eagle, prairie falcon, ferruginous hawk). These data come from the CPW SDE SAM Raptor Nesting database. There are 36,388 recorded nest observations in the database, 723 of which are from unique observations of active nests in the last 10 years for our focal species. We used landscape layers relating to land cover classes (linear distance to water features, linear distance to cliffs/bluffs/rocky outcrops, herbaceous grassland, cottonwood, mixed forest, shrubland/scrub-steppe grassland, riverine/riparian, cultivated areas, developed areas, and linear distance to roads), topography (elevation, local elevational difference, and topographic ruggedness index [TRI]), and temperature (degree-days above 5°C). We also included layers that indicate prairie dog range and prairie dog colonies for black-tailed prairie dogs, Gunnison's prairie dogs, and white-tailed prairie dogs. We supply the predicted use surface for each species, wherein red areas are more suitable locations (i.e., Pr[use] ~ 1) and gray areas are less suitable locations (i.e., Pr[use] ~ 0) in Figure 1.

  We have written the results of these analyses as a manuscript and acceptance at the *Journal of Raptor Research* pending revisions.

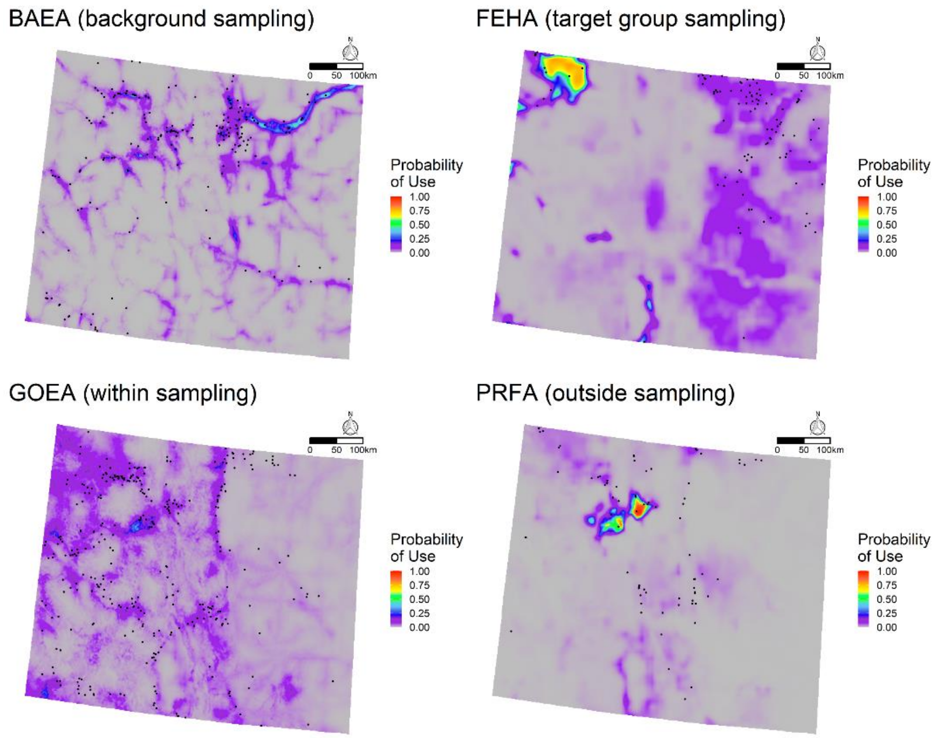

Figure 1. Expected suitable nesting habitat for (A) bald eagle; (B) golden eagle; (C) ferruginous hawk; and (D) prairie falcon in Colorado. White represents likely suitable habitat, black represents unlikely suitable habitat. White points represent observed nest locations.

- **Colorado Bat Distribution Model (with D. Neubaum)** — We compiled expected distribution models and range maps for 13 species of Colorado-resident bats species using location data of radio-tagged bats (see Figure 1 for example, below). A stated goal is to generate baseline expectations for bat distributions for comparative use in the event that white-nose syndrome (*Pseudogymnoascus destructans*) expands its range into Colorado. We have completed analyses and submitted the resulting manuscript for peer-review at the *Journal of Wildlife Management*. Future objectives include evaluating likely species movement corridors using landscape movement models.

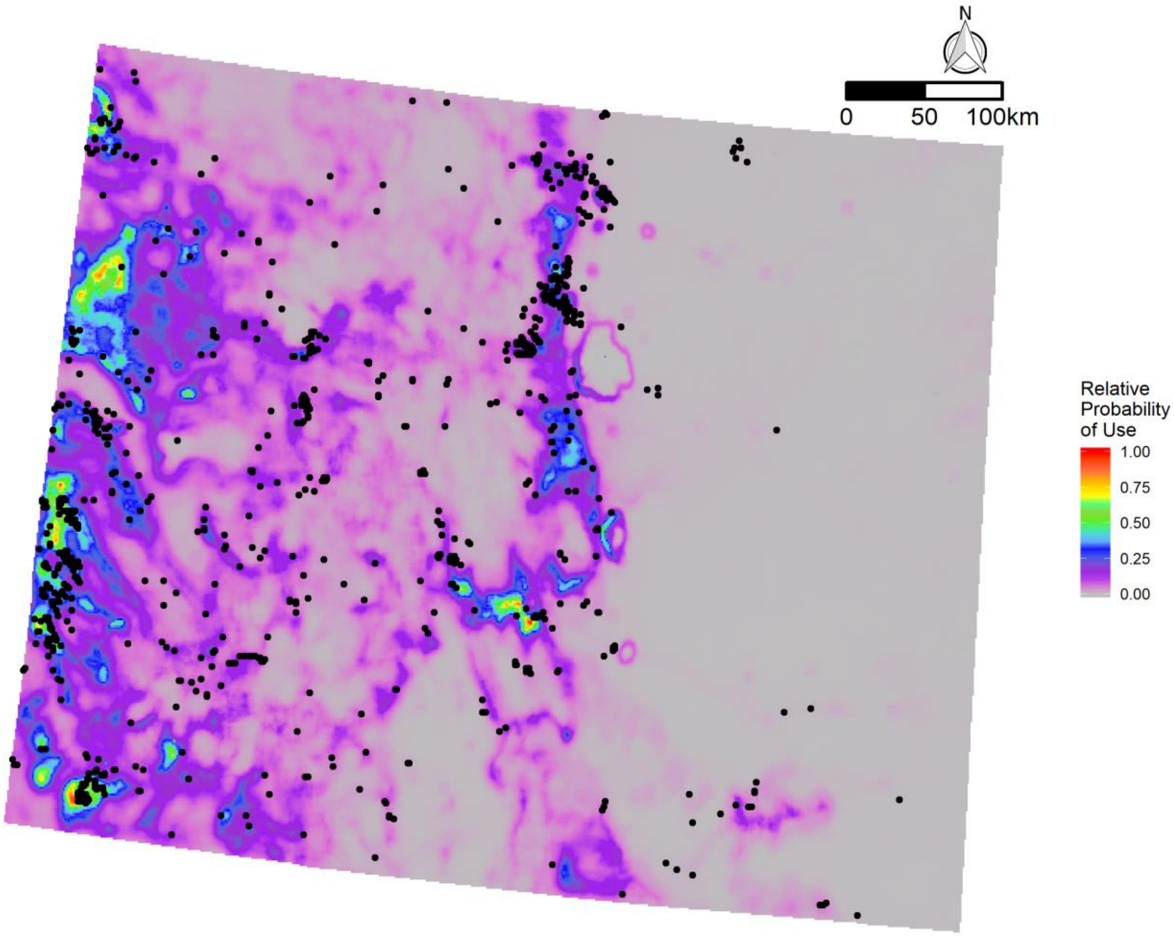

Figure 2. Example bat distribution model for al myotid species across the state. Warm (red) colors represent likely suitable habitat; cool (gray) colors represent likely unsuitable habitat. Black points represent observed locations.

- **Systematic literature review of select raptor home range size (with M. Kocina)** — We systematically searched the open literature for information on HRS for Bald Eagles (*Haliaeetus leucocephalus*), Ferruginous Hawks (*Buteo regalis*), Golden Eagles (*Aquila chrysaetos*), and Prairie Falcons (*Falco mexicanus*). We found 24 articles with HRS estimates and accompanying methodology and demographic information on sampled individuals. Most studies focused on Bald Eagles, followed by Golden Eagles, Prairie Falcons, and Ferruginous Hawks. HRS estimates for the Golden Eagle were the largest and had the greatest associated variance (μ = 8,797 km^2, 95% confidence interval (CI) = 0 – 47,284 km^2). Estimates for Bald Eagle HRS were smaller, with a mean of 2,215 km^2 (95% CI = 0 – 12,472 km^2). Prairie Falcon and Ferruginous Hawk HRS estimates were much smaller, with means of 156 km^2 (95% CI = 0 – 415.22km^2) and 22 km^2 (95% CI = 0 – 96.88 km^2), respectively. HRS estimates varied substantially across period (breeding/nonbreeding), sex, age class, fix type, and estimation method for all species, and points to the importance of accounting for the context of these estimates. The information can be used to inform other efforts to characterize their spatial use. The results of this review were written as a manuscript and have been accepted for publication at *Western North American Naturalist*.

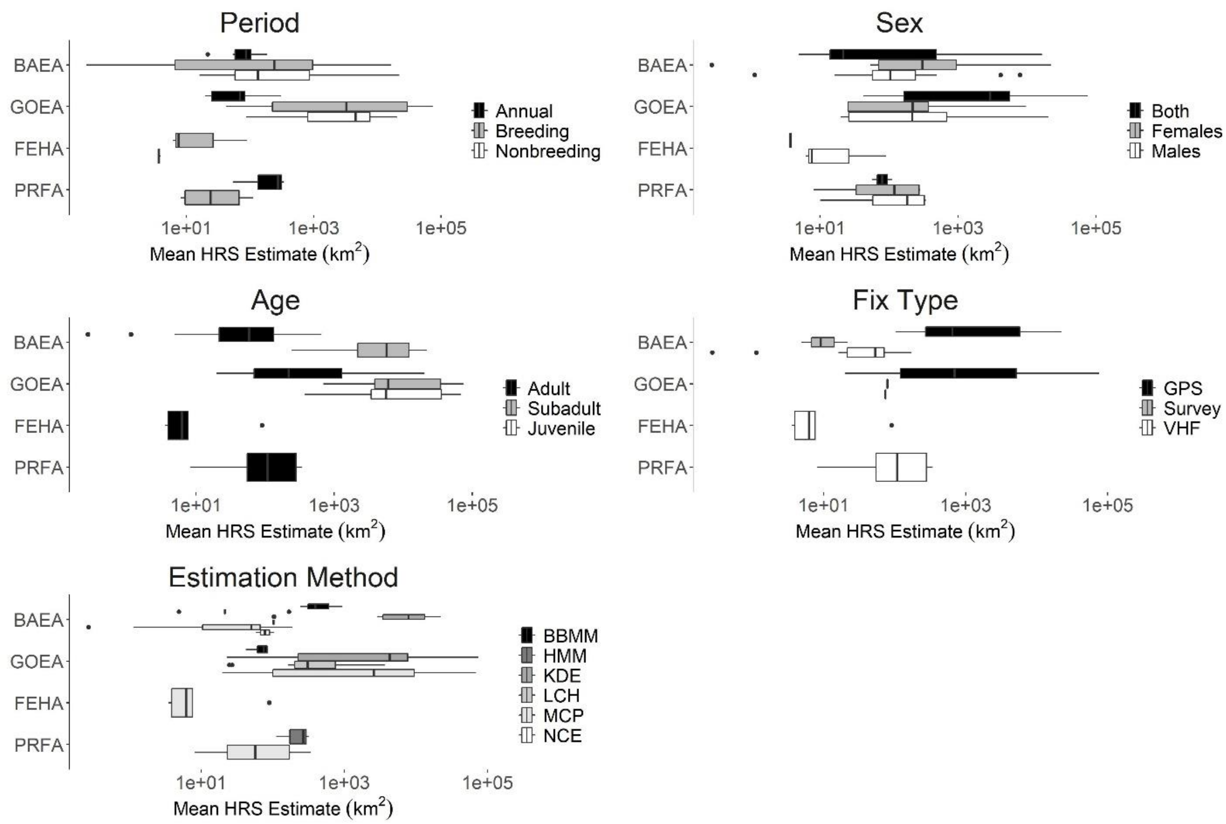

Figure 3. Mean home range size estimates across all studies (shapes) and corresponding standard deviation (error bars) for each domain (species-category-variable combination). Size of shapes indicates the mean number of individuals per study included in the derivation of the mean for that category.

- **Gunnison Sage-grouse Habitat-use Model (with T. Apa, L. Wiechman [USFWS], M. Rice [USFWS], J. Heinrichs [USGS], M. O'Donnell [USGS], C. Aldridge [USGS], S. Oyler-McCance [USGS])** — We worked with members of the U. S. Fish and Wildlife Service and U. S. Geological Survey to develop management-focused habitat-use models (resource selection function, RSF) for Gunnison sage-grouse (*Centrocercus minimus*) populations. We have developed the landscape habitat covariate layers for use in the RSF and have developed the distributional models. We worked with area biologists and wildlife managers to identify which covariates in certain contexts (populations and seasons) are the most useful from a management perspective. The results of this effort have been written in a manuscript and submitted for peer-review to *Wildlife Research*.

- **Generalized avian movement and energetics model (with W. Thogmartin [USGS], M. Hooper [USGS], M. Wildhaber [USGS], and B. West [USGS])** – We aim to generate a continental scale model to inform migration dynamics of mallard-like dabbling ducks across North America and approximate realistic energetic stress-responses to oiling events (e.g., spills). We are building upon a spatially explicit energetics-based model of avian migration (with descriptive manuscript in review at *Ecology and Evolution*) to evaluate the consequences of oiling to stopover time, body condition, refueling capacity, and survivorship during the nonbreeding period of the annual cycle. We have thus far found expected results of decrease survivorship with increasing degree of oiling, and are evaluating the effects of modifying the location and severity of the oil spill. Results of this work will be written in a manuscript and submitted for publication as a peer-reviewed article in the *Journal of Environmental Management*.

- **Black swift breeding phenology (with C. Gunn)** — We analyzed over two decades of breeding phenology and nest success data, collected from 1996 through 2017. We documented dates of first arrival, laying, incubation onset, hatching, and fledging, and determined the intervals from arrival to laying and from laying to incubation, and the durations of incubation and nestling period in each year. All breeding events followed each other closely and showed little chronological change throughout the study. The estimate of nest success for all nest attempts was 77.5%. We have written these results in a manuscript which has been accepted for publication at *The Wilson Journal of Ornithology*.

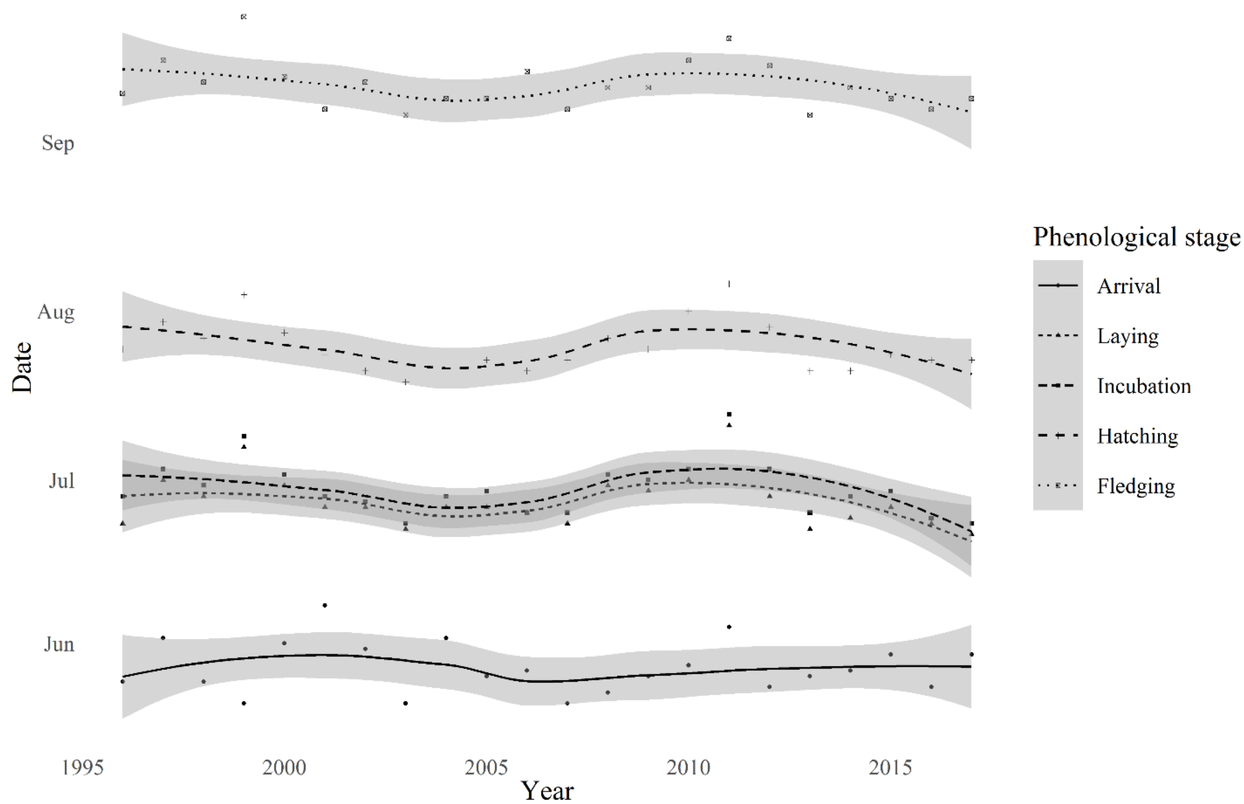

Figure 5. Breeding phenology of Black Swifts at the Box Canyon colony, Ouray, Colorado, 1996-2017. Gray bars indicate the 95% CI.

- **Anthropogenic development around bald eagle nests (with J. Gammonley, R. Conrey) —** We sought to determine the degree of development-related incursion around Bald Eagle nests in the Front Range of Colorado (defined as the counties of Adams, Arapahoe, Boulder, Broomfield, Denver, Douglas, Jefferson, Larimer and Weld). We used nest locations derived from the Colorado Parks and Wildlife (CPW) Spatial Database Engine (SDE) Raptor Nesting Database. We included only "occupied" nests (those for which an observer was confident a nesting attempt was occurring at the time of observation). Bald Eagle nests along the Front Range occur in locations with greater than average development, though we cannot say whether this is from site selection by eagles (actively choosing to build nests in developed areas) or expanding development into previously occupied nesting habitat. Given the nature of annual Bald Eagle nesting behavior (returning to and building on top of previously existing nests), we speculate it is more likely the latter than the former.

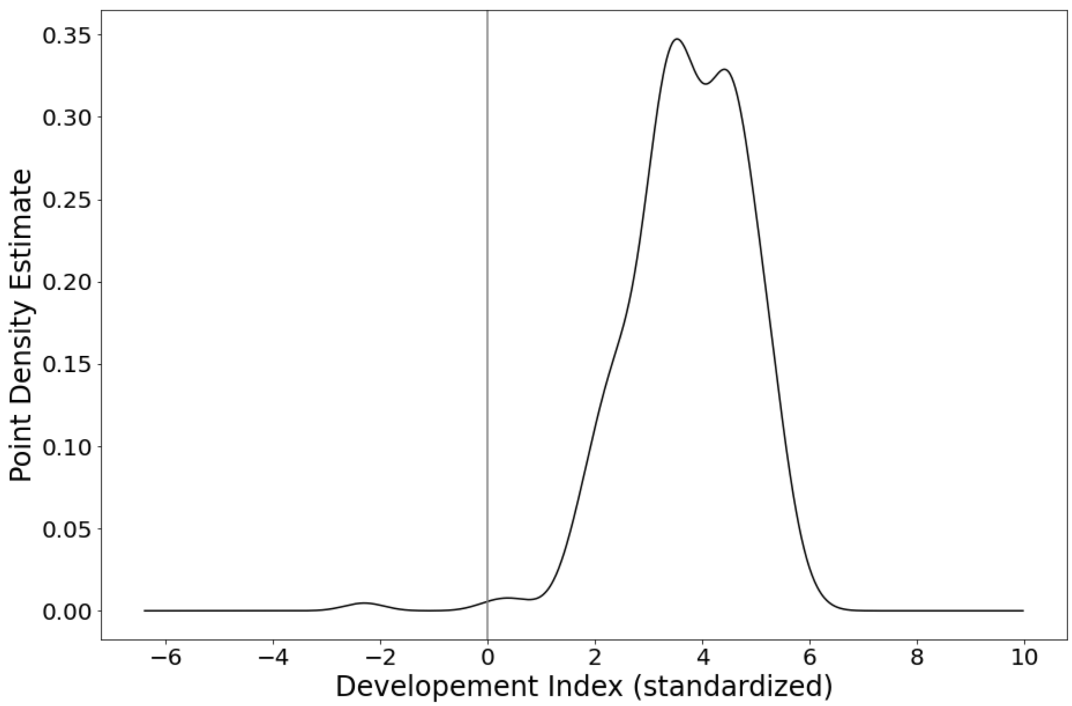

Figure 6. Kernel density plot of development index values at Bald Eagle nests (black line) and across all of the Front Range of Colorado.

- **Various projects related to Big Game management, in concert with Mammals Researchers and members of the Terrestrial Section (with C. Anderson, E. Bergman, and A. Holland) –** Using spatial analyses I assisted in projects related to movement and habitat selection of Big Game in Colorado. Using telemetry data and cluster analysis I was able to identify the seasonal movements of Mule Deer in the Piceance basin (Figure 7) and develop a seasonal habitat buffer for use in a quantification tool for the Colorado Department of Transportation. Using time series data on moose parturition dates I applied a logistic regression analysis in a Bayesian framework to develop a tool to more precisely gauge the true date of birth for moose calves in Colorado (Figure 8). Results of this work were written as a manuscript and published in the journal *Alces*.

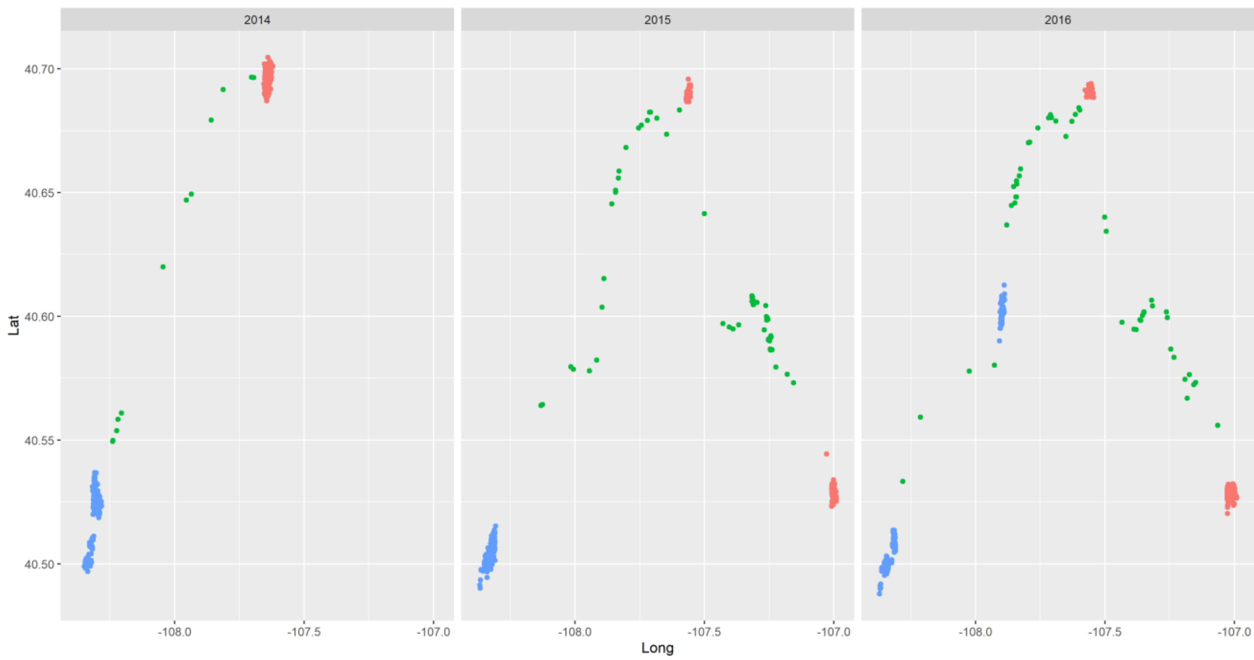

Figure 7. Seasonal delineation of locations of Mule Deer in the Piceance Basin. Blue locations clustered together to represent locations in winter habitat, red represent locations in summer habitat or at stopovers, and green are locations in transition habitat.

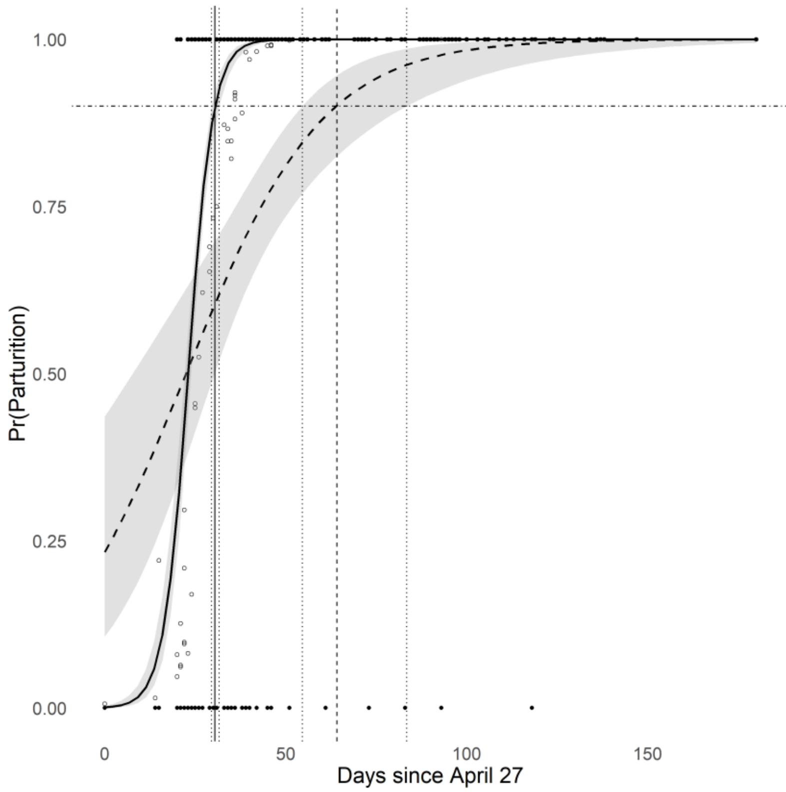

Figure 8. Predicted probability of parturition by date (shown as days since April 27th), modeled with (solid) and without (dashed) including the probability of detection. The horizontal dotted-dashed line indicates a 90% parturition probability. The vertical lines indicate the days on which that 90% parturition probability was estimated to have been achieved for each model (dotted lines represent 95% credible intervals).

Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Avian response to plague management on Colorado prairie dog colonies

Period Covered: January 1 – December 31, 2020

Principal Investigator: Reesa Yale Conrey, reesa.conrey@state.co.us

Project Collaborators: Dan Tripp, Jim Gammonley, Miranda Middleton, Cooper Mark, CPW; Erin Youngberg, Arvind Panjabi, Bird Conservancy of the Rockies; City of Fort Collins Natural Areas and Utilities Programs; Bureau of Land Management (Gunnison and Ca. on City offices); National Park Service Florissant Fossil Beds National Monument; and CPW wildlife managers, biologists, park rangers, and property technicians from Areas 1, 4, 14, and 16.

EXTENDED ABSTRACT

Prairie dogs (*Cynomys* sp.) are highly susceptible to plague, a disease caused by the non-native bacterium *Yersinia pestis,* introduced to the Great Plains of North America in the 1940s–50s (Ecke and Johnson 1952, Antolin et al. 2002). Plague epizootics may have cascading effects on species associated with prairie dog (*Cynomys* spp.) colonies, such black-footed ferrets (*Mustela nigripes*), ferruginous hawks (*Buteo regalis*), and burrowing owls (*Athene cunicularia*). Colorado Parks and Wildlife (CPW) has completed a study of plague management in prairie dogs, in which oral vaccine treatments were compared to placebo baits and insecticidal dusting of burrows (Tripp et al. 2017). Our objective is to quantify the effects of plague and plague management on avian species and mammalian carnivores associated with colonies of black-tailed (*C. ludovicianus:* BTPD) and Gunnison's (*C. gunnisoni*: GUPD) prairie dogs. Working at sites receiving vaccine, placebo, insecticidal dust, and no treatment, we have sampled colonies before, during, and after plague epizootics. We also compared on- and off-colony areas at GUPD sites during 2013-2015, in order to better quantify the effect of GUPD on shrub-steppe communities.

Data collection over seven years has included: avian point counts, summer and winter raptor surveys, burrowing owl surveys and nest monitoring, monitoring of all raptor nests located opportunistically, remote camera data targeting mammalian carnivores, and percent ground cover, visual obstruction, and species composition of vegetation at points, nests, and along randomly located transects. In prior years, we also monitored passerine nests and surveyed for mountain plover (*Charadrius montanus*).

Study areas include BTPD colonies in north-central Colorado and GUPD colonies in western and central Colorado. BTPD study colonies are dominated by short and mid-grasses (especially blue grama *Bouteloua gracilis* and buffalograss *B. dactyloides*) and located in Larimer and Weld counties adjacent to the Wyoming border, managed by the City of Fort Collins. GUPD study colonies are dominated by sagebrush (especially big sagebrush *Artemisia tridentata*) mixed with other shrubs and grasses and located in the Gunnison Basin (Gunnison County), northwest Saguache County, Woodland Park area (Teller County), South Park (Park County), and Baca National Wildlife Refuge (Saguache County). GUPD sites are managed by the Bureau of Land Management, U.S. Forest Service, National Park Service, U.S. Fish and Wildlife Service, and CPW. Study sites were grazed by cattle (and sheep in Baca

NWR) and native grazers, especially prairie dogs, pronghorn (*Antilocapra americana*), jackrabbits (*Lepus* sp.), and cottontails (*Sylvilagus* sp.).

Over a 3-year period starting in fall 2013, plague epizootics occurred over >80% of the BTPD study area. Some colonies, particularly those receiving dust or vaccine, have had increasing prairie dog numbers since initially declining during the peak of the epizootic, while others, especially untreated areas, have continued at severely reduced acreage (Tripp et al. 2017). Precipitation has varied greatly over the course of this study, from slightly dry to very wet, compared to the 30-year average. This plague cycle began during a dry period but peaked during two wet years. In contrast, we observed very little plague activity (two small colonies) at GUPD sites until the 2017 field season, when epizootics began at several colonies. The 2018 season was our first opportunity to collect data on post-plague communities on GUPD colonies.

To summarize the phases of this research project:

- Phase 1 (2013-2015) featured active vaccine research (vaccine, insecticide, and placebo treatments) by CPW Wildlife Health and plague epizootics across much of the BTPD site but almost *no* plague at GUPD sites. We did extensive avian field work at BTPD sites, on and off GUPD colonies, and nest searching at all sites.
- Phase 1.5 (2016) featured the early use of plague vaccine as a management tool for CPW. Plague continued at some BTPD colonies. Because plague research goals could not be pursued at GUPD sites without plague, we discontinued avian work in Woodland Park and Gunnison Basin. We started work on GUPD colonies (extant and extirpated) in South Park, ahead of planned GUPD reintroductions (which then did not happen).
- Phase 2 (2017-2019) featured broader plague management by CPW Terrestrial staff at all our GUPD sites and some BTPD sites. Plague epizootics began in some GUPD sites in Woodland Park, Gunnison Basin, and then Baca NWR (new site in 2017), so we resumed on-colony (but not off-colony) work at GUPD sites. BTPD sites began a post-epizootic growth cycle.
- Phase 3 (2020-?) features less intensive longer-term monitoring (e.g., point counts, vegetation transects, and camera surveys) of species associated with prairie dogs at sites with varying levels of plague management. This will require close collaboration internally and externally to monitor colony boundaries and changes in prairie dog activity caused by plague.

The following results are preliminary. At BTPD colonies, we detected more Brewer's blackbirds (*Euphagus cyanocephalus*), vesper sparrows (*Pooecetes gramineus*), and horned larks (*Eremophila alpestris*) during point counts in active colonies, and more grasshopper sparrows (*Ammodramus savannarum*) and lark buntings (*Calamospiza melanocorys*) in colonies impacted by plague (which intersected with wet years). Grasses were taller and plant cover generally higher following epizootics, which likely contributed to higher densities of species that prefer taller vegetation structure and lower densities of those that prefer shorter stature vegetation. In both summer and winter raptor counts, during which we recorded time spent within colonies, ferruginous hawks showed the strongest preference for foraging on active vs. post-plague colonies, with a use rate six times higher on active colonies. American kestrels (*Falco sparverius*) and golden eagles (*Aquila chrysaetos*) had use rates 2 – 4 times higher on active colonies. In contrast, burrowing owls, which are known to be associated with BTPD colonies (e.g., Butts & Lewis 1982, Tipton et al. 2008) and were by far the most commonly detected raptor in our summer surveys, had use rates ~2.5 times higher on post-plague colonies. Although seemingly counterintuitive, this confirms results from Conrey (2010), who found high densities of burrowing owls nesting on post-plague colonies where small numbers of BTPD occurred. Looking across raptor species, the pattern of higher use of active vs. post-plague colonies was stronger in winter than in summer. Additional analyses of bird data are planned, with the inclusion of covariates related to colony characteristics, weather, vegetation, and for raptors, alternative prey such as lagomorphs.

Badgers and coyotes had 20-30% lower usage of BTPD colonies following plague events. Swift fox showed the opposite pattern, but prairie dog activity had a weaker effect on fox occupancy, and this species may be responding more strongly to coyotes, which prey upon swift fox (Kamler et al. 2003, Karki et al. 2006). Occupancy models containing prairie dog activity had 99.9% of model weight for coyotes and badgers and 82.7% for swift fox. Detection rates for all three species were higher when more cameras were deployed and during August-April, compared to May-July. Coyotes and badgers appear to respond negatively to plague in prairie dogs, which dramatically reduces abundance of an important prey item. Future analyses of camera data will incorporate additional years of data and more covariates and may include multi-species models (allowing coyote-fox interaction) and relative abundance models.

Plague management via vaccine delivery and insecticidal dust can reduce the impact of plague on prairie dogs (Tripp et al. 2017) and their associates. Smaller scale applications within larger BTPD complexes did not eliminate plague but helped to maintain pockets of live prairie dogs and promote population recovery. This mosaic of active and plague-affected areas retains habitat for species associated with colonies. Not surprisingly, species that prey upon prairie dogs or preferentially forage in short stature grasslands are the most likely to benefit from plague management. It will likely take additional years of monitoring to detect potential changes in the avian community caused by different types of plague management, as treated colonies no longer experience extinction events and over time diverge from untreated areas.

We created a time lapse video from footage at two burrowing owl nests (posted publicly February 2020):

- https://www.youtube.com/watch?v=D8eVuO5h59Y
- https://www.facebook.com/104599519602883/videos/618208965577832/?__so__=channel_tab&__rv__=all_videos_card

Progress and completed project components in 2020:

- 2019 was the final year of intensive sampling for this research project, with only camera trapping occurring during 2020.
- We pulled all cameras from the field in May 2020, with a total of 4.17 million photos collected.
- We created a time lapse video from burrowing owl nest footage that was publicly posted in Feb 2020.
- R. Conrey presented this research for a large CPW audience virtually during Conservation Over Virtual Interface Days (COVID).

Plans for 2021 and beyond:

- Cooperate with Terrestrial and Wildlife Health staff and external partners to continue monitoring colony boundaries and prairie dog/plague activity at research sites.
- Rotate among BTPD and GUPD sites over future years, conducting point counts, vegetation, and camera surveys every few years.
    - Sample as many GUPD sites as possible in 2021 (dependent on other priorities and COVID restrictions).
    - Collaborate with Bird Conservancy on point counts at BTPD sites (SPNA & MSR) in 2022.
    - We will track longer-term impacts of different plague management strategies on the community of wildlife associated with prairie dog colonies.
- Continue data analyses and preparation of manuscripts:
    - Changes in grassland bird densities at BTPD sites over two plague and recovery cycles (14+ years), co-authored with Bird Conservancy of the Rockies.
    - Changes in bird density or occupancy at GUPD sites, with comparisons of active vs. plagued sites and on- vs. off-colony sites.
    - Grassland bird nest survival and relationship to plague, weather, carnivore occupancy, and other factors.
    - Site use/occupancy of mammalian carnivores, with comparisons of active vs. plagued sites.

- Site use of raptors, with comparisons of active vs. plagued sites.
- Changes in plant community related to plague, weather, biosolids applications, and other factors.

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Figure 1. Photos from BTPD and GUPD sites in Colorado. a) GUPD consuming experimental bait. b) Ferruginous hawk seen during a winter raptor count. c) Visual obstruction measurement. d) Burrowing owl on BTPD site. e) Coyote and badger photographed by remote camera.

Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Golden eagles in Colorado: monitoring methods and status evaluation

Period Covered: January 1 – December 31, 2020

Principle Investigators: R. Yale Conrey reesa.conrey@state.co.us, K. Aagaard, J. Gammonley

Project Collaborators: Bird Conservancy of the Rockies; U.S. Fish and Wildlife Service; U.S. Forest Service; Bureau of Land Management; National Park Service; Boulder County; other agencies who have submitted nest data; Cornell Lab of Ornithology. CPW Species Conservation Unit, GIS Unit, and Biologists: especially L. Rossi (SCON); J. Thompson (Resource Stewardship); R. Sacco (GIS); A. Estep, M. Sherman, M. Cowardin, L. Carpenter, & Senior Terrestrial Biologists (TERR).

EXTENDED ABSTRACT

Raptor monitoring databases have generated important insights into various aspects of raptor ecology and can provide a sound foundation for management of individual species or within the larger context of managing targeted habitats (Greenwood 2007). CPW has a statewide raptor nest database developed by R. Sacco (GIS Unit), which currently contains records for >11,000 nest locations of 30 species going back to the 1970s. Until recently, the nest database was primarily being used by CPW at a site-specific scale in the oil and gas consultation process (Colorado House Bill 07-1298) and other local-scale land use input. This continues to be an important function of the raptor data, which will be even more prominent in decision making going forward, as Colorado Senate Bill 181 requires annual updates of the raptor data for COGCC (Colorado Oil and Gas Conservation Commission). As part of this research project, the potential of these data to assess raptor populations at regional and statewide scales has been evaluated and field protocols are being optimized to yield more useful information. In addition, we have begun a pilot study in the SE Region to evaluate monitoring methods and population status of golden eagles (hereafter, GOEA), a Tier 1 Species of Greatest Conservation Need (CPW 2015).

The CPW raptor nest database contained nest records for 11,097 locations on 7 January 2021 (Table 1A), having grown from 8,696 locations in 2016 due to increased sampling effort. Although the majority of nest locations (5,799 nests) have an unknown or undetermined status (with no information about occupancy during the past 5 years), this proportion has been reduced from 70% in 2016 to 52% of the total at the end of 2020, due to increased sampling effort, especially at historic nest sites.

In early 2020, Avian Research and Terrestrial staff completed a raptor nest monitoring protocol and revised the nest datasheet, with a goal of standardizing monitoring methods statewide and ensuring that relevant data are reported in fields that can be queried for analysis. Stated priorities are nests visits for Golden Eagle, Bald Eagle, Peregrine Falcon, Prairie Falcon, Ferruginous Hawk, and Swainson's Hawk (Northern Goshawk are actively monitored by USFS), especially to nests that were last checked ≥5 years ago and are losing their "known" 5-year status. The new protocol requests submission of all records, rather than a single annual summary record, including visits to unoccupied alternate nest structures (where birds have built several structures within a territory). For nests that will be monitored multiple times within a season, observers should try to determine when incubation is initiated (laying of the first egg) and hatching and fledging occur. The 2020 updated datasheet provides "unknown" and "not

applicable" options in all relevant fields to discourage leaving fields blank. Nest status has been clarified and expanded into three separate fields for bird occupancy, nest structure, and fate of the nesting attempt. Observations of behaviors, nestling age, and potential disturbances that previously could only be described as "Comments" are now quantified in separate fields that can be queried for analysis.

As of 15 January 2021, not all data have been incorporated into the statewide database and quality control (especially of new data fields added for 2020) is ongoing. When this is completed, our goals are to assess fields left blank or improperly entered, calculate apparent nest success for species with adequate sample size, and summarize potential nest threats. We anticipate that this process will reveal some needed revisions to protocols for 2021. For example, we are proposing an earlier deadline for data submission (1 November rather than 1 December), and we are differentiating between permanent infrastructure and more recent sources of disturbance near nests, as these may have different impacts on breeding raptors.

We completed distribution models using the existing CPW nest database for four priority species: bald eagle, golden eagle, ferruginous hawk, and prairie falcon. We used generalized linear models to identify the relationship between nest locations and explanatory covariates relating to land cover, temperature, topography, and prey distribution. We investigated the effect of differential use of available locations by comparing four different selection frames. A manuscript (Aagaard et al. *in revision*) is being revised for Journal of Raptor Research, which we believe will soon be accepted for publication.

In 2019, we began a new SCTF-funded raptor project that will continue through 2023 and focuses on GOEA in the SE Region of Colorado; this focus was agreed upon by statewide Regional, Terrestrial, and Research staff during a September 2018 meeting. Objectives are to better describe GOEA population status and analyze the cost:benefit ratio of monitoring methods that incorporate detection probability (therefore allowing estimation of abundance and trend), minimize sampling bias (which will also produce improved distribution models), explore use of citizen science (e.g., eBird) data, and estimate productivity at a subset of nests. In April 2019, we piloted a method for aerial raptor nest surveys that allows estimation of detection probabilities, documents non-detections (rather than presence-only), and minimizes road bias. Using a CPW Cessna aircraft, we flew north-south transects as well as one tributary and one canyon route that covered most of Crowley and ~half of Otero County in Area 12. We used double-observer methods and distance sampling, categorizing nest detections into one of three strata (plains, canyon/bluff, or associated with water) and placed into ¼ mile distance bins. We attempted to record UTMs when the plane drew even with the nest. We also recorded bird species and structural characteristics (e.g., intact/dilapidated and tree species) whenever possible, plus time, weather, and altitude.

As a result of these flights, we detected ~80 raptor nest structures, most of which were not previously included in the statewide database, in an area where only three bald eagle nests were being actively monitored. Analyses of detection probability and comparison of efficacy of distance sampling versus double-observer methods are ongoing. Flight plans for 2020 had to be postponed due to COVID-19 restrictions. However, we were able to expand ground-based monitoring of known nests in the SE Region during 2020, including further ground-truthing and observations at nests discovered during 2019 flights. We are currently making plans to fly with fixed wing aircraft and possibly with a helicopter (which can fly more slowly and potentially get more direct line-of-sight) during spring of 2021. We hope to do a follow-up flight over structures detected during transect surveys of the plains and optimize flights over canyons and tributaries (where detection was difficult due to topography and aircraft speed).

Other data sources have potential to contribute to our understanding of Colorado raptors, including eBird, Breeding Bird Survey, and Colorado Breeding Bird Atlas. As we are better able to achieve survey coverage through flights, we hope to further evaluate how citizen science data can be used along with survey data collected by CPW staff, partners, and volunteers in distribution or occupancy modeling.

We hope to continue progress on statewide assessments of raptors in Colorado during 2021 by providing improved data collection and modeling. However, meeting these goals will also require continued articulation of CPW objectives for raptor monitoring and priorities for raptor conservation and management.

Progress and project components completed during 2020:

- Staff and volunteers statewide collected raptor nest data using standardized monitoring methods and a revised data form that will facilitate queries of nest fate, bird behaviors, and potential threats.
- Re-submitted manuscript (Aagaard et al.) on raptor distribution models, which is expected to be accepted to Journal of Raptor Research after we complete the requested revisions.
- Due to COVID-19 restrictions, flights were not possible during 2020.
- Expanded ground-based monitoring of known nests in the SE Region, including further ground-truthing and observations at nests discovered during 2019 flights. Eagle nests were observed multiple times to estimate apparent nest survival and productivity.

Plans for 2021:

- Continue data queries and quality control as the remainder of 2020 raptor nest data are submitted and incorporated into the statewide database. Calculate apparent nest survival rates and summarize threat data.
- Complete revisions of Aagaard et al. manuscript on raptor distribution models for publication in JRR.
- Conduct additional aerial surveys (fixed wing and possibly helicopter) in a different portion of the SE Region to locate previously unreported raptor nests while testing methods that account for detection probability.
- Conduct repeat observations at a subset of golden eagle nests, focusing on the SE Region, in order to estimate nest survival and productivity.

Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Behavioral and demographic patterns of nesting bald eagles along a gradient of human disturbance on the Front Range corridor in Colorado

Period Covered: January 1 – December 31, 2020

Authors**:** Reesa Yale Conrey

Principle Investigators: R. Yale Conrey reesa.conrey@state.co.us, K. Aagaard, J. Gammonley

Project Collaborators: M. Smith and B. Snyder, Bird Conservancy of the Rockies; M. Lockhart, Wildlands Consulting; W. Kendall, Colorado Cooperative Fish and Wildlife Research Unit; U.S. Fish and Wildlife Service; Front Range Cities & Counties; private landowners; M. Sherman, L. Carpenter, L. Rossi, R. Sacco, B. Marette, NE Region staff from Areas 2, 4, and 5, CPW

External funders: Audubon Society of Greater Denver's Lois Webster Fund; U.S. Fish and Wildlife Service Region 6 Migratory Bird Program

EXTENDED ABSTRACT

The bald eagle (*Haliaeetus leucocephalus*) is a Tier 2 species of greatest conservation need in the Colorado State Wildlife Action Plan (Colorado Parks and Wildlife 2015). Historically, bald eagles commonly occurred in northcentral Colorado during migration and winter, but the state was considered to be only a peripheral part of the breeding range (Craig 1979). By the end of the 1970s, there were only three known occupied bald eagle nests in Colorado and none in the Front Range (Craig 1998). Bald eagle populations declined in the early- to mid-20th century due to pesticides (primarily DDT), human persecution, and land conversion. Recovery began with the banning of DDT in 1972 and listing of bald eagles under the newly created Endangered Species Act in 1973.

Bald eagles have recovered from dramatic population declines and were removed from the federal threatened and endangered species list in 2007. However, there is still concern about the status of local and regional populations, and the potential impacts of land use changes on bald eagles. Bald eagles are a high-profile species with strong interest from the public, and along the Colorado Front Range corridor where bald eagles and humans coexist in close proximity, public awareness of bald eagles is high and citizens closely track individual bald eagles and their nests. With a rapidly expanding human population along the Front Range, development (residential, business, energy, etc.) and other forms of land use change regularly create concerns about impacts on bald eagles, and particularly the loss of nest sites. The U.S. Fish and Wildlife Service is currently developing standards for allowing limited take of eagle nests (U.S. Fish and Wildlife Service 2016) and regularly seeks input from Colorado Parks and Wildlife (CPW) on human-eagle issues.

In recent decades, a relatively high concentration of breeding pairs has become established in the Colorado Front Range (Wickersham 2016). Human activity may negatively impact bald eagles at breeding sites or winter roosts (Buehler 2020). CPW and the U.S. Fish and Wildlife Service have recommended disturbance buffer distance and timing restrictions for bald eagle nests and roost sites (U.S.

Fish and Wildlife Service 2007, CPW 2020). However, bald eagles exhibit a wide range of tolerance and response to various human activities and their proximity (Buehler 2020), making it challenging to develop disturbance mitigation recommendations that are both defensible and consistent. We predict that along the northern Front Range, nesting bald eagles that are regularly exposed to human activity are more tolerant of human activities and at closer distances than eagles using nest sites where human activities are limited.

The goal of this study is to better understand current demographics and space use of bald eagles breeding along the northern Front Range, and the impact of human disturbance and changing land use on these measures. We are conducting this project during 2020–2024. Specific objectives include: 1) Quantify changes in land use around bald eagle nests along the northern Front Range over the past three decades. 2) Quantify and compare demography (breeding effort, breeding success, survival) and space use (home range, daily movements) of bald eagles nesting along a gradient from sites with little historical and no new disturbance activity to sites with relatively high historical disturbance levels and significant new disturbance activity during the study. 3) Quantify nonbreeding survival, movements, and space use of bald eagles that breed along the northern Front Range, in relation to anthropogenic features.

The study area includes the Front Range corridor of northcentral Colorado in Adams, Arapahoe, Boulder, Broomfield, Denver, Douglas, Jefferson, Larimer, and Weld counties. This is an area of rapid human population growth (18% growth from 2000 to 2020) and a relatively high concentration of bald eagles throughout the year. Nests are routinely exposed to varying levels of disturbance and most have been closely monitored for multiple years to determine annual occupancy and success.

CPW obtained a statewide land-use and land-cover dataset consisting of five layers which quantified oil and gas development, wind and solar energy development, and residential development between 1970 and 2020 (Sushinsky 2020). We also incorporated roads (Colorado Department of Transportation) and trails (both accessed through CPW's SDE) layers, which were static (without temporal data). Wind, solar, and oil and gas data were available annually, but we summarized them by decade to match the timescale of the other data. We calculated a development index within 20 km^2 of each Front Range bald eagle nest by summing the distance layers (from nests to wind turbines, solar arrays, power lines, oils and gas wells, roads, and trails) and layers related to urbanization (exurban, suburban, urban, and commercial/industrial). We compared the development index near nests ("used") to what is "available" on the landscape. During 1990 to 2020, an increasing number of nests had a higher proportion of residential and commercial/industrial development (urban index) within 20 km^2 (Fig. 1). This suggests that in 2020, bald eagle nests in the northern Front Range were subject to a broad range of anthropogenic impacts. Overall development (residential, commercial, and energy development, plus roads, trails, and powerlines) near nests (Fig. 2) was higher than expected by chance.

In 2020, Bird Conservancy of the Rockies (BCR) continued its Bald Eagle Watch program, where volunteers monitor known bald eagle nests. CPW staff worked with BCR to ensure complete coverage of nests within the study area. BCR and CPW have standardized monitoring protocols that provide detailed information to determine nest activity and fate, as well as habitat features and potential disturbance sources. For all nests, observers determined if the nest was occupied, and whether the nest was destroyed (e.g., by a weather event or a nest tree falling down), failed, hatching success (i.e., at least one egg hatched), and whenever possible, fledging success (i.e., at least one young fledged). Occupied nests were observed multiple times (typically every two weeks or more frequently) to determine nest fate. Preliminary results show that in 2020, 138 known bald eagle nests were monitored in the study area. Of these, 27 nests (19.6%) were destroyed prior to the 2020 nesting season, and an additional 16 nests (11.6%) were unoccupied. Of the 95 occupied nests, 20 nests failed to hatch any nestlings (21%), including four nests that were destroyed during the nesting season. A total of 75 nests produced 150 nestlings, and 72 nests produced 138 fledged young (76% apparent nest success). Of successful nests, 28% produced one fledgling, 55% produced two fledglings, and 17% produced three fledglings.

We are estimating nest survival rates for bald eagles and determining what ecological and anthropogenic covariates are important predictors of nest survival. Prior to beginning this research project, we began a modeling effort using existing data with 163 nest attempts at 86 locations from

2012−2016. An additional 179 nests during those 5 years were not included, typically because the observer did not visit enough times or at the appropriate time to confirm nest fate. Preliminary results suggested that daily nest survival was best modeled by nest stage, maximum temperature in June, and time in season. Other covariates for land cover, roads, distance to water, monthly precipitation, location (Front Range vs. elsewhere), and year were not supported. We did not have enough information to properly analyze the effects of nest substrate (live tree, dead tree, or other) or disturbance (traffic, recreation, etc.). We expect that more of these covariates will be informative in new models built with more recent data, due to larger sample sizes and more frequent nest visits that prioritized collection of the information required for nest survival modeling.

The traditional nest survival model does not incorporate uncertainty in nest initiation or completion dates or nest stage (incubation of eggs vs. chick-rearing). Therefore, B. Kendall, our collaborator at USGS Colorado Cooperative Fish and Wildlife Research Unit, has begun development of a multi-event nest survival model that explicitly incorporates some types of uncertainty. Thus far, he has simulated 50 bald eagle nests with 2 – 3 visits each, calculating two survival parameters (one for each stage). This model produced unbiased estimates and reasonable precision, with higher precision for the nestling stage (which lasts longer) than the incubation stage and when nests were visited more frequently.

As of January 2021, we have 18 solar-powered transmitters using a GSM (Global System for Mobile Communication) platform, in which the transmitter's location is determined and recorded based on its proximity to cell phone towers. These transmitters are smaller and less expensive than transmitters that transmit signals to satellites, and because there are many cell phone towers throughout the study area, we expected GSM transmitters to be very effective. Transmitter data service provided one location every 2 hours during the night, every 15 minutes during the day when the eagle was not moving, and every 4 - 7 seconds when the eagle was flying ("flight mode"), except when voltage declined close to the winter solstice and daytime locations were provided every 5 minutes.

We partnered with a consultant with extensive experience to lead our efforts to trap and mark eagles using methods approved by the CPW Institutional Animal Care and Use Committee. Due to COVID-related problems, transmitter shipment was delayed and we were unable to attempt to capture eagles until July, when only a few active nests with unfledged young remained. We have had only two successes in 18 all-day trapping attempts, but we anticipate that pairs feeding nestlings will be more likely to feed on baits and hope that trapping success will increase in the spring. We attempted to capture one member of a pair of eagles at active nest sites, using baited, padded leg-hold traps (Bloom et al. 2007) adjusted for safe eagle capture; trap sets were under nearly constant observation and field personnel immediately retrieved captured eagles. We marked each captured eagle with a standard U.S. Geological Survey rivet leg band and a GSM transmitter using a break-away backpack style X-harness constructed of Teflon ribbon straps (total weight of the transmitter and harness approximately 70 g: < 2% body mass of an adult male). The harness we used typically breaks down and the transmitter drops off within several years after marking.

We successfully marked an adult female from a successful nest with two nearly-fledged young in Larimer County on 13 July (Fig. 3). We have been tracking her movements since then and have visually confirmed that the eagle appears to be doing well and the transmitter is working properly. In the three months after marking (late summer to fall), the eagle used a core area of about 393.4 km^2, which included several water bodies and a mix of rural, exurban, suburban, and urban development (Fig. 4). The eagle left this core area in September and traveled 447.6 km to southeastern Wyoming, then returned to the core area three days later (Fig 4). Within the core area, the eagle moved an average of 105 km per day (24-hr period), ranging from 1.8 km (18 August) to 344.8 km (12 September).

Following the nesting season, we successfully trapped a female eagle near a nest in Weld County on 18 October. This eagle left the study area and moved to southern Colorado several days after marking. She remained in SW Colorado for 5 weeks before returning to the Front Range, where she was exploring the region between Denver and Fort Collins as of mid-January.

We will continue to annually monitor nesting activity and land use patterns at all known nests through the 2024 nesting season. We will continue to monitor the two eagles we have successfully marked, and we will attempt to capture and mark additional eagles in 2021. Data on marked eagles will be collected at least through 2024. We will analyze data and prepare reports and manuscripts for publication in 2025. Results will be used to model the bald eagle population trajectory and expected impacts of predicted future land use change, and to make recommendations on minimizing and mitigating disturbances near nests. This study will provide a better understanding of this species' tolerance of and adaptability to human activities and land use changes. The results will also improve long-term bald eagle monitoring efforts in Colorado.

Progress and project components completed during 2020:

- Calculated a preliminary development index for eagle nest sites using new data layers.
- Coordinated with partners, including private landowners, cities and counties, to access nest sites.
- Captured and attached transmitters to two adult eagles.
- Adjusted statewide raptor monitoring protocols to facilitate data collection on nest stage, nest fate, and potential sources of disturbance.
- Monitored 95 occupied bald eagle nests on the Front Range with multiple visits per site.

Plans for 2021:

- Continue spatial analysis on land use metrics.
- Tag as many breeding adults as possible (up to 30).
- Continue to evaluate monitoring data and tweak protocols to maximize their usefulness.
- Monitor all occupied bald eagle nests on the Front Range at least every two weeks.
- Continue to evaluate movement data and space use by transmittered birds.

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Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Northern bobwhite response to short-duration intensive grazing on Tamarack State Wildlife Area

Period Covered: January 1 – December 31, 2020

Principal Investigator: Adam C. Behney adam.behney@state.co.us

Project Collaborators: Trent Verquer, Ed Gorman, Jim Gammonley

All information in this report is preliminary and subject to further evaluation. Information MAY NOT BE PUBLISHED OR QUOTED without permission of the author. Manipulation of these data beyond that contained in this report is discouraged.

EXTENDED ABSTRACT

Widespread suppression of historic disturbance regimes have reduced heterogeneity in vegetation communities on which many wildlife rely for various life events and stages. Northern bobwhites require areas of thicker grass cover for nesting within close proximity to more open areas with bare ground and abundant food-producing forbs for brood-rearing and feeding. Altered or eliminated vegetation disturbance has been implicated in the range-wide decline of northern bobwhite populations. Lack of disturbance on state wildlife areas in Northeast Colorado has caused the vegetation to become uniformly dense and tall which is likely not meeting the needs of all parts of the northern bobwhite life cycle. Some type of disturbance is required to reduce the vegetation biomass and create some of the open structure on which bobwhites rely. Grazing represents one of the only options for disturbance at Tamarack State Wildlife Area and other similar riparian areas in northeast Colorado. Whereas unmanaged continuous grazing has been linked to degradation of bobwhite habitat quality, short-duration high-intensity grazing holds promise to reduce the vegetation biomass and rejuvenate the habitat to become more attractive to bobwhites.

The objectives of this project were to assess the efficacy of using short-duration high-intensity grazing as a tool to improve northern bobwhite habitat. We used a randomized block design in which we divided the study site into groups of four plots, one of which was grazed each year over a three year period and one was a control (Fig. 1). Beginning in late winter each year, we captured bobwhites using walk-in traps and deployed necklace-style VHF radio transmitters. We located each radio-marked bobwhite three times per week and determined nest sites by observing birds in the same location on subsequent days. When nests hatched we continued to monitor broods and on day 14 post-hatch we flushed the brood, and weekly thereafter to count chicks and assess brood status. To assess nest and brood site selection, we sampled vegetation at nest and brood sites and four associated random points to represent available habitat around the nest or brood site. The overall goals were to estimate adult, nest, and brood survival as well as nest and brood site selection in relation to grazing treatment and other general habitat characteristics.

Grazing had no effect on nest or brood survival or brood habitat selection. However, northern bobwhites avoided grazed plots for nesting. Percent litter negatively influenced nest survival (Fig. 2). Grass cover was positively related to nest site selection whereas, bare ground was negatively related to nest site selection (Fig. 3). Brood habitat selection was positively related to woody vegetation and negatively related to bare ground (Fig. 4). Vegetation was impacted by grazing immediately after grazing, however, the effects generally disappeared by the end of the growing season (Fig. 5). Forbs were an exception and tended to be more abundant on grazed plots throughout the summer. Overall, northern

bobwhite demographics and habitat selection were not significantly affected by grazing and effects on vegetation were neutral to positive. Spring high-intensity short-duration grazing appears to be ineffective to manage habitat for northern bobwhites in northeastern Colorado.

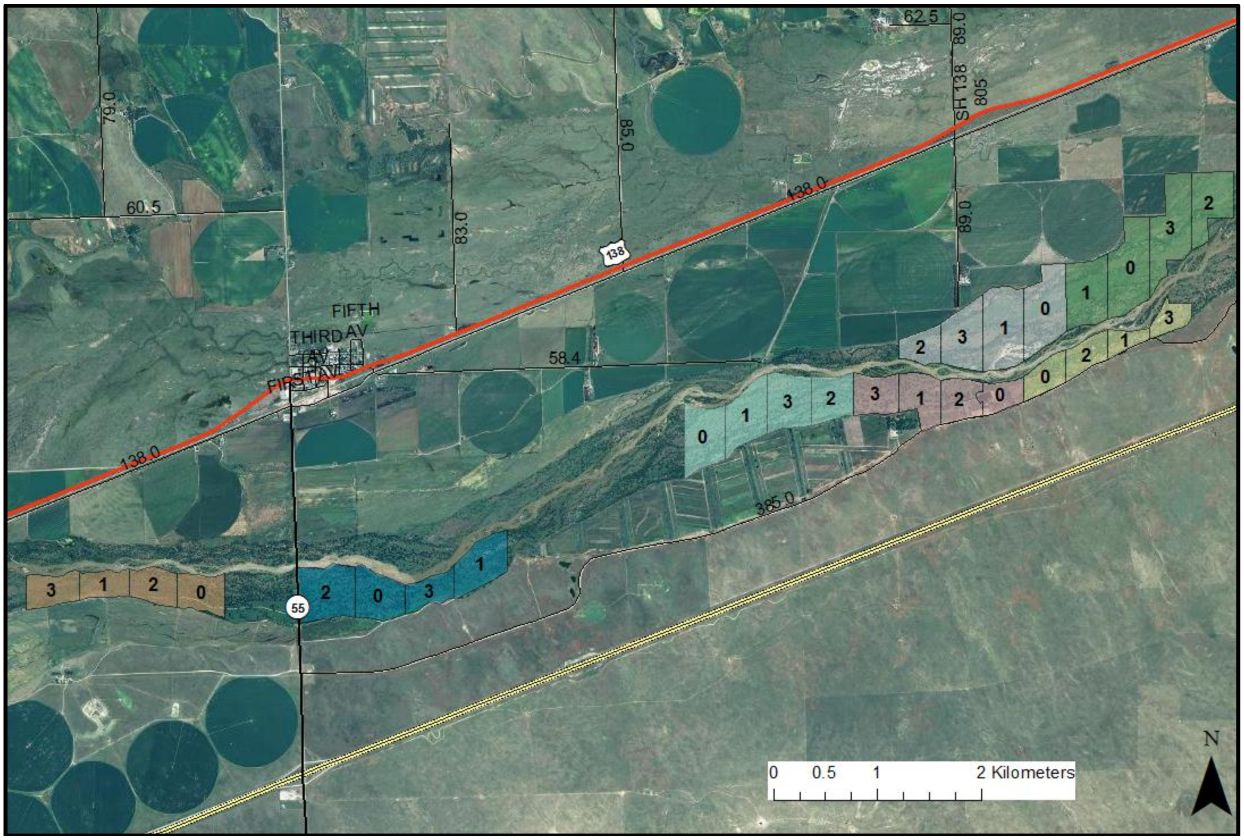

Figure 1. Grazing treatment plot layout for Tamarack State Wildlife Area. Numbers represent the year of treatment, zeros indicate control plots.

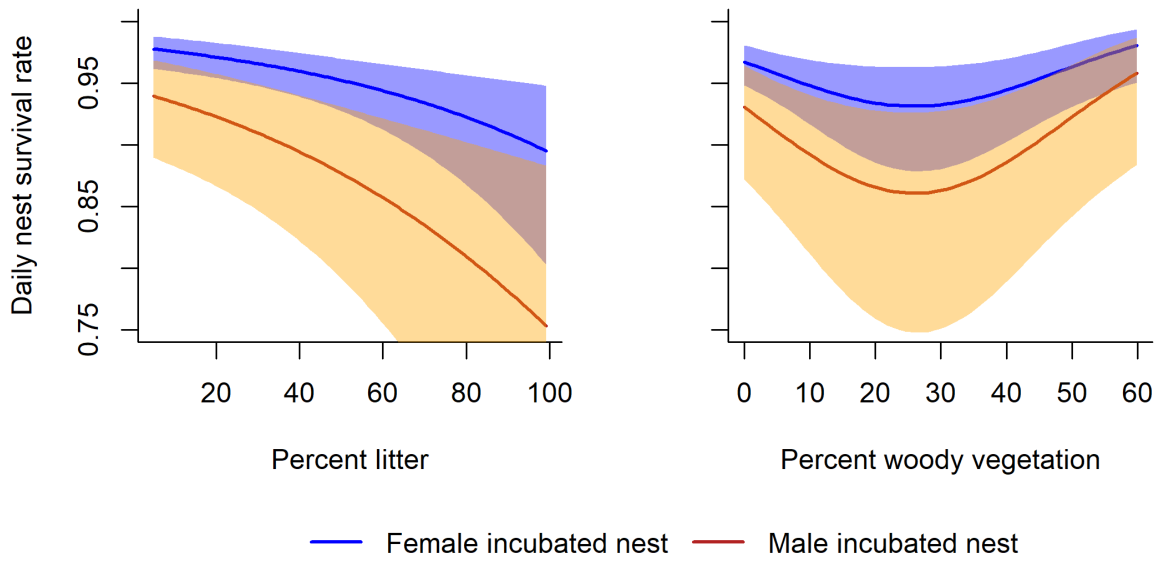

Figure 2. Daily nest survival rates for male and female-incubated nests of northern bobwhites in relation to percent litter and woody vegetation around nests in northeastern Colorado during 2016–2019. Shaded areas represent 90% credible intervals.

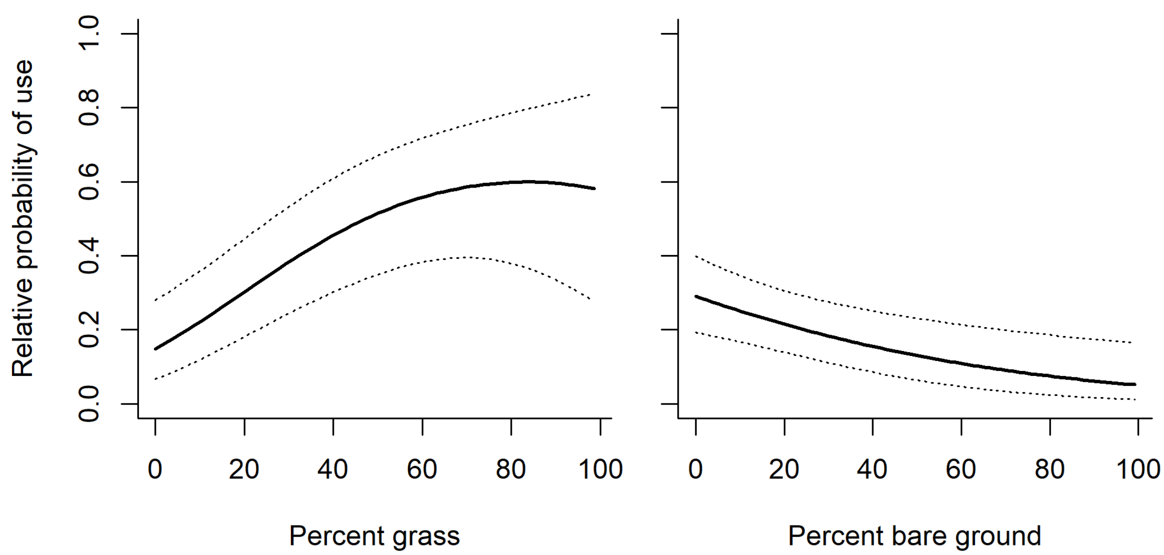

Figure 3. Relative probability of use as a nest site in relation to percent grass and bare ground for northern bobwhites in northeastern Colorado during 2016–2019. Dotted lines represent 90% credible intervals.

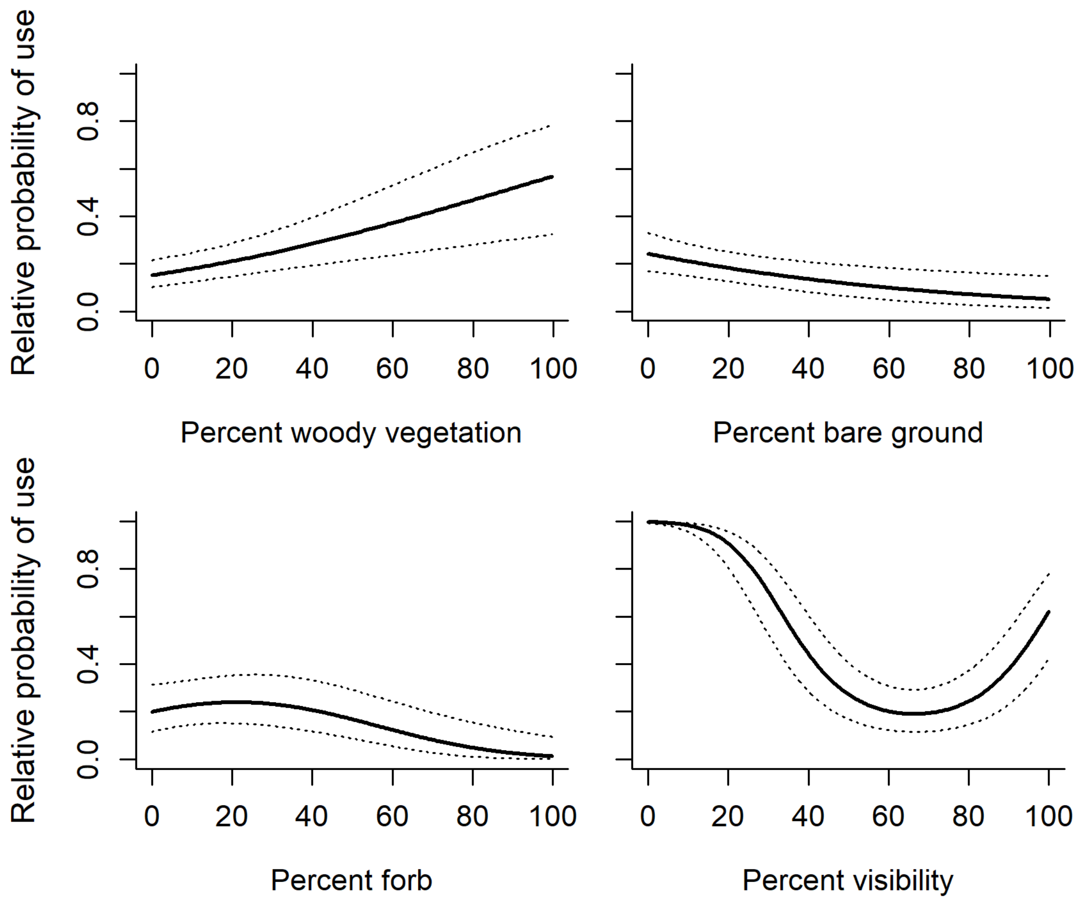

Figure 4. Relative probability of use as brood sites based on percent woody vegetation, percent bare ground, percent forb, and percent visibility for northern bobwhites in northeastern Colorado, 2017–2019.

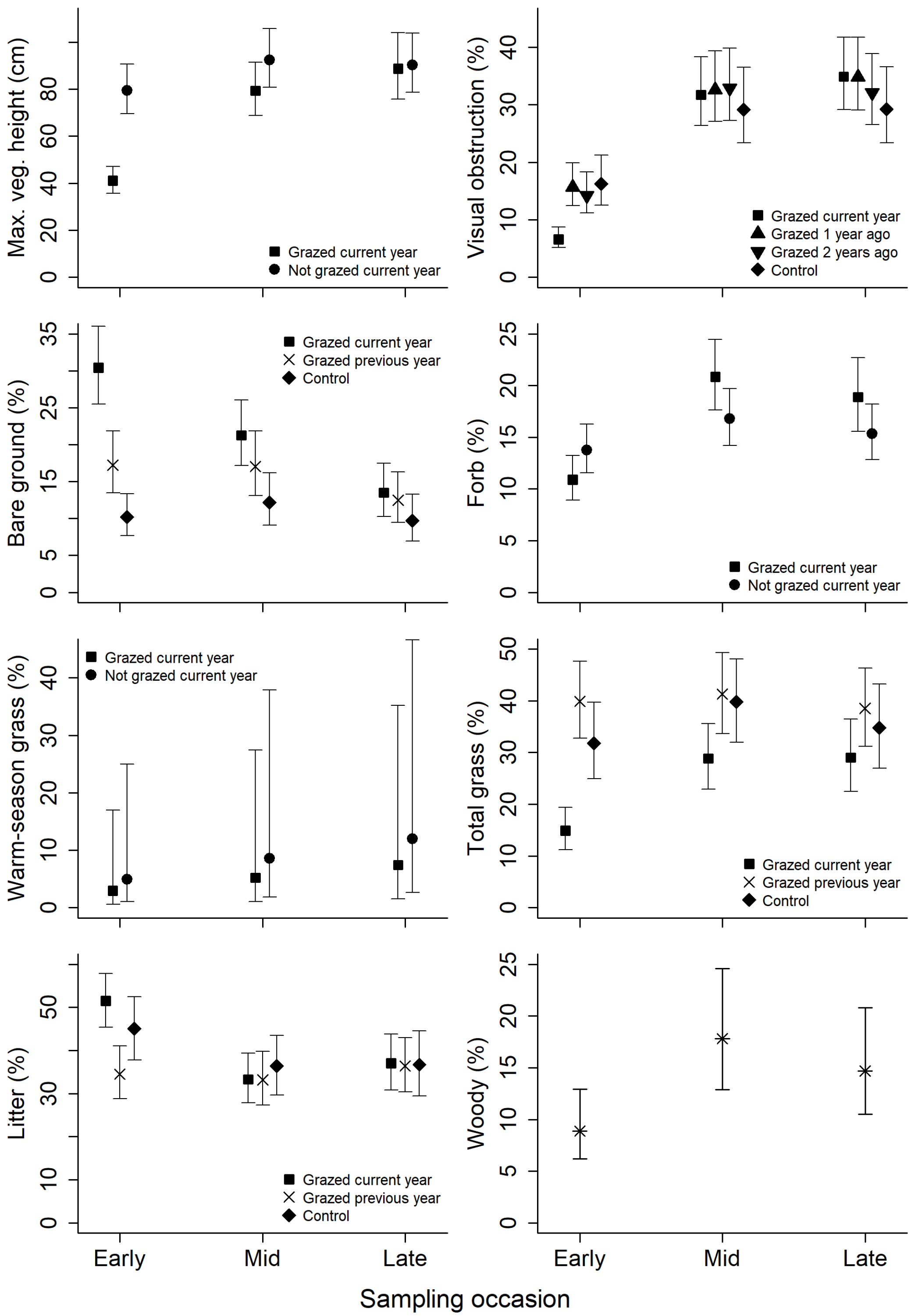

Figure 5. Vegetation characteristics from the most-supported model for each vegetation characteristic sampled at random points throughout grazed and control plots during early (Apr), mid (Jun-Jul), and late (Aug-Sep) sampling occasions in northeastern Colorado, 2017–2019. Error bars represent 90% credible intervals.

Colorado Parks and Wildlife

WILDLIFE WILIDLIFE RESEARCH PROJECT SUMMARY

Nonbreeding season survival and habitat use of northern bobwhite

Period Covered: January 1 – December 31, 2020

Principal Investigator: Adam C. Behney adam.behney@state.co.us

Project Collaborators: Larkin Powell, Joseph Wolske, University of Nebraska-Lincoln

ABSTRACT

Identifying the vital rates to which population growth rate is limited by, or sensitive to, can help guide management actions aimed to affect population size. For bobwhites, some studies have suggested that some populations can be sensitive to adult nonbreeding season survival, especially in northern parts of their range. We have recently completed a research project looking at bobwhite demography during the breeding season but we do not have any information on population characteristics during the nonbreeding season. Therefore, our goals with this project were to estimate survival and assess habitat selection of northern bobwhites during the nonbreeding season. We also assessed whether bobwhites would use artificial structures in areas that seem suitable except for a lack of cover. If we observed bobwhites using artificial structures, it would confirm our suspicion that woody cover limits bobwhite occupancy in those areas.

The first field season ended successfully in March 2020. We deployed 98 transmitters during the first field season across two state wildlife areas and created five individual artificial quail structures. The best survival model estimated constant probability of survival. Nonbreeding season survival (26 weeks) in 2019–2020 was $\hat{S} = 0.235$ (95 % CI: 0.165–0.323) with weekly survival of $\hat{S} = 0.946$ (0.932–0.957). The second and final field season commenced in September 2020 and field work is underway.

Colorado Parks and Wildlife

WILDLIFE RESEARCH PROJECT SUMMARY

Estimates and determinants of duck production in North Park, Colorado

Period Covered: January 1 – December 31, 2020

Principal Investigators: Adam C. Behney adam.behney@state.co.us and James H. Gammonley jim.gammonley@state.co.us

Project Collaborators: Derek Danner, Ella Engelhard, Kris Middledorf, Allicyn Nelson, Brian Sullivan, Carolin Tappe (CPW); Casey M. Setash and David Koons (Colorado State University); Tara Wertz (Arapaho National Wildlife Refuge); Matt Reddy (Ducks Unlimited Inc.)

EXTENDED ABSTRACT

Assessing waterfowl use and productivity throughout the Intermountain West can inform habitat management practices across various land use regimes. The North Platte River Basin (hereafter, North Park) in Jackson County in northcentral Colorado has historically held important breeding and stopover habitat for ducks and is expected to become increasingly important as water demands increase across the state. In 2018, we began a study to examine duck breeding populations and production in North Park, in relation to wetland habitat conditions.

During the 2020 field season, our first objective was to estimate the breeding population of ducks and evaluate the variation in abundance across wetlands. We used independent double observer surveys on riparian areas and dependent double observer surveys on basin wetlands, and surveyed 128 individual wetlands for breeding ducks. The number of indicated breeding pairs was greatest on wetlands with more open water. Summed across all sites, we observed 4,350 total indicated breeding pairs, including 1,360 gadwall (*Mareca strepera*), 516 mallard (*Anas platyrhynchos*), 452 lesser scaup (*Aythya affinis*), and 377 cinnamon teal (*Spatula cyanoptera*). Because mallards, gadwall, cinnamon teal, and lesser scaup were the most commonly detected species, we modeled pair abundance separately for these species in addition to all ducks combined. For all ducks combined and each species except lesser scaup, a model including a linear time trend outperformed models with quadratic, or cubic time trends and the null model. These time trends decreased throughout the survey period. For lesser scaup, the null model outperformed any time trend model. For each species and all species combined, we added vegetation variables to the best time trend model. For all ducks, cinnamon teal, gadwall, and lesser scaup, the best vegetation model included percent of the wetland that was open water, with the number of breeding pairs increasing with open water. For mallards, the best vegetation model included percent of the wetland classified as shrub/scrub and the number of mallard indicated breeding pairs increased with increasing levels of shrub/scrub vegetation.

Our second objective was to assess nesting characteristics of waterfowl. We searched 73 plots across five privately-owned ranches, Arapahoe National Wildlife Refuge, Lake John State Wildlife Area, and Hebron Slough Waterfowl Area within areas that were impacted by flood irrigation.We located 36 nests of five species throughout the 2020 breeding season. A large portion (69.4%) of these nests were located on Arapahoe NWR while 30.6% (*n*=11) were associated with working lands. Thirteen nests successfully hatched at least one duckling, while seven failed due to investigator disturbance. Nest

survival varied by habitat type, with temporary emergent wetlands having the highest 33-day nest survival (0.585, 95% CI: 0.33–0.84) and shrub/scrub habitat having the lowest (0.162, 0.00–0.47).

Our third objective was to estimate duck production across the park, using independent double observer surveys of duck broods. We conducted brood counts at 99 sites including 76 basin wetlands, 13 hay meadows, 6 riparian transects, and 4 irrigation ditch transects from 1-Jul until 4-Sep. We observed broods of 13 duck species. Summed across sites we observed 613 broods including 226 gadwall, 76 cinnamon teal, 51 mallard, and 50 lesser scaup broods. On average, we conducted 3 brood surveys per site. Similar to the analysis for pair counts, we modeled mallards, gadwall, cinnamon teal, and lesser scaup separately in addition to all ducks combined. For all ducks combined and each species separately, the model of brood abundance including date in a quadratic form outperformed models including date in cubic or linear form, as well as the null model. Brood abundance peaked in early August for all species. For each species and all species combined, we added vegetation variables to the best time trend model. For all ducks, mallard, gadwall, and lesser scaup, the best vegetation model included percent of the wetland that was flooded/inundated, with the number of broods increased with percent flooded. For cinnamon teal, the best vegetation model included percent open water; cinnamon teal brood abundance increased with increasing amounts of open water. Duckling:pair ratio for all ducks ranged from 0 to 19 and averaged 1.39 (SD=3.1).

A fourth study objective is to use banding data to obtain demographic estimates and the contribution of North Park ducks to hunting opportunity. In 2020 we banded 1,294 ducks during preseason banding operations (Table 1). As part of a related study at Colorado State University, 34 mallards (4 adult females, 22 adult males, 3 local females, 5 local males) and 19 gadwall (14 adult females, 5 adult males) were captured and banded prior to pre-season banding (22 April through 21 July). Of these, 4 adult female mallards and 13 adult female gadwall were marked with transmitters. During pre-season banding, 24 additional adult female mallards and 1 adult female gadwall were also marked with transmitters.At the time of this report, 97 ducks we banded in 2018, 77 ducks we banded in 2019, and 85 ducks we banded in 2020 (total = 259) had been harvested by hunters and reported to the USGS Bird Banding Laboratory.

We plan to continue annual data collection on this study through 2023.

Table 1. Numbers of ducks banded in North Park during pre-season capture efforts in 2020. LM = local male, LF = local female, HYM = hatch year male, HYF = hatch year female, AHYM = after hatch year male, and AHYF = after hatch year female.

Species[a]	LM	LF	HYM	HYF	AHYM	AHYF	Total
Mallard	20	25	266	200	223	105	839
Gadwall	51	45	20	15	21	60	212
Cinnamon/blue-winged teal[b]	4	4	45	40	2	6	101
Northern shoveler	0	2	19	9	5	5	40
Green-winged teal	0	0	12	6	16	5	39
American wigeon	2	4	8	3	2	3	22
Lesser scaup	1	6	1	0	0	3	11
Northern pintail	1	0	2	5	1	1	10
Canvasback	7	0	0	2	0	1	10
Redhead	1	3	1	1	2	1	9
Ring-necked duck	0	0	0	0	0	1	1
Total	87	89	374	281	272	191	1,294

Scientific names: blue-winged teal (*Spatula discors*), northern shoveler (*S. clypeata*), green-winged teal (*Anas crecca*), American wigeon (*Mareca americana*), northern pintail (*Anas acuta*), canvasback (*Aythya valisineria*), redhead (*Aythya americana*), ring-necked duck (*Aythya collaris*).

[b]We could not reliably distinguish between cinnamon and blue-winged teal for locals and females.

Publications, presentations, workshops and committee involvement by Avian Research staff
January – December 2020

PUBLICATIONS

Aagaard, K., R. Y. Conrey, and J. H. Gammonley. *Accepted*. Spatial analysis of the nesting distribution of four priority raptor species in Colorado. Journal of Raptor Research.

Aagaard, K., E.V. Lonsdorf, and W.E. Thogmartin. *In review*. A continental generalizable avian movement and energetics model. Ecology and Evolution.

Apa, A. D., **K. Aagaard**, M. B. Rice, E. Phillips, D. Neubaum, N. Seward, J. R. Stiver, and S. Wait. *Accepted*. Species distribution models for a threatened species: the Gunnison sage-grouse. Wildlife Research.

Apa, A. D., **J. H. Gammonley**, D. J. Neubaum, E. Phillips, J. P. Runge, N. Seward, S. Wait, and B. Weinmeister. *In revision*. Survival rates of translocated Gunnison sage-grouse. Wildlife Society Bulletin.

Behney, A. C. 2020. Ignoring uncertainty in predictor variables leads to false confidence in results: a case study of duck habitat use. Ecosphere 11:e03273.

Behney, A. C., J. M. Wolske, T. M. Cucinotta, and C. Tappe. 2020. Factors influencing trapping success of northern bobwhites. Wildlife Society Bulletin 44:240-245.

Behney, A. C. 2020. The influence of water depth on energy availability for ducks. Journal of Wildlife Management 84:436-447.

Behney, A. C. *In press*. Rapid assessment of habitat quality for nonbreeding ducks in Northeast Colorado. Journal of Fish and Wildlife Management.

Behney, A. C. *In press*. Benefits of playa buffers as bird habitat. Wilson Journal of Ornithology.

Behney, A. C. *In review*. High-intensity short-duration grazing during spring is not an effective habitat management tool for Northern Bobwhite in Colorado. Condor.

Bergman, E., F. P. Hayes, and **K. Aagaard**. 2020. Incorporating detection probability to refine Colorado moose parturition dates. Alces 56:127-135.

Donnelly, J. P., S. L. King, J. Knetter, **J. H. Gammonley**, V. J. Dreitz, B. A. Grisham, M. C. Nowak, and D. P. Collins. *In press*. Migratory efficiency sustains connectivity across agroecological networks supporting sandhill crane migration. Ecosphere.

Garbowski, M., C. S. Brown, and **D. B. Johnston**. 2020. Soil amendment interacts with invasive grass and drought to uniquely influence aboveground versus belowground biomass in aridland restoration. Restoration Ecology 10.1111/rec.13083.

Garbowski, M., **D. B. Johnston**, and D. S. Baker. *In review*. Invasive annual grass interacts with drought to influence plant communities and soil moisture in dryland restoration. Ecosphere.

Garbowski, M., **D. B. Johnston**, and C. S. Brown. *Accepted*. Cultivars of popular restoration grass developed for drought do not have higher drought resistance and do not differ in drought-related traits from other accessions. Restoration Ecology.

Gunn, C., S. E. Hirshman, and **K. Aagaard**. *In press*. Trends in black swift (*Cypseloides niger*) breeding phenology and success in southwest Colorado, 1996 – 2017. The Wilson Journal of Ornithology.

Johnston, D. B. 2020. Piceance Basin restoration for wildlife. Colorado Parks and Wildlife Technical Report 57, Fort Collins, CO.

Johnston, D. B., and M. Garbowski. 2020. Responses of native plants and downey brome to a water-conserving soil amendment. Rangeland Ecology and Management 73:19-29.

Kircher, A. A., **A. D. Apa**, **B. L. Walker**, and R. S. Lutz. 2020. A rump-mount harness design improvement for greater sage-grouse. Wildlife Society Bulletin 44:623-630.

Kirol, C. P., D. C. Kesler, **B. L. Walker**, and B. C. Fedy. 2020. Coupling tracking technologies to maximize efficiency in avian research. Wildlife Society Bulletin 44:406-415.

Kocina, M, and **K. Aagaard**. *In press*. A review of home range sizes of four raptor species of regional conservation concern. Western North American Naturalist.

Lindstrom, J. M., M. W. Eichholz, and **A. C. Behney**. 2020. Effect of habitat management on duck behavior and distribution during spring migration in Indiana. Journal of Fish and Wildlife Management 11:80-88.

Neubaum, D, and **K. Aagaard**. *In review*. Use of predictive distribution models to describe habitat use by Colorado bats. Ecology and Evolution.

Peterson, R. O., R. L. Beschta, D. J. Cooper, N. T. Hobbs, **D. B. Johnston**, D. Kotter, E. J. Larsen, D. R. MacNulty, K. N. Marshall, L. E. Painter, W. J. Ripple, S. D. W., and E. C. Wolf. 2020. Indirect effects of carnivore restoration on vegetation. *In* D. W. Smith, D. R. Stahler, and D. R. MacNulty, editors. Yellowstone wolves: Science and discovery in the world's first national park. University of Chicago Press, Chicago, IL.

Shyvers, J. E., **B. L. Walker**, S. J. Oyler-McCance, J. A. Fike, and B. R. Noon. 2020. Genetic mark-recapture analysis of winter faecal pellets allows estimation of population size in Sage Grouse *Centrocercus urophasianus*. Ibis 162:749-762. DOI: 10.1111/ibi.12768.

Tabak, M. A., M. S. Norouzzadeh, D. W. Wolfson, E. J. Newton, R. K. Boughton, J. S. Ivan, E. A. Odell, E. S. Newkirk, **R. Y. Conrey**, J. Stenglein, F. Iannarilli, J. Erb, R. K. Brook, A. J. Davis, J. Lewis, D. P. Walsh, J. C. Beasley, K. C. VerCauteren, J. Clune, and R. S. Miller. 2020. Improving the accessibility and transferability of machine learning algorithms for identification of animals in camera trap images: MLWIC2. Ecology and Evolution 10:10374-10383.

Walker, B. L., M. A. Neubaum, S. R. Gorfoth, and M. M. Flenner. 2020. Quantifying habitat loss and modification from recent expansion of energy infrastructure in an isolated, peripheral greater sage-grouse population. Journal of Environmental Management **255**:109819.

Walker, B. L., L. D. Igl, and J. A. Shaffer. 2020. The effects of management practices on grassland birds—Brewer's Sparrow (*Spizella breweri breweri*), Chapter AA *of* D. H. Johnson, L. D. Igl, J. A. Shaffer, and J. P. DeLong, J.P., editors. The effects of management practices on grassland birds: U.S. Geological Survey Professional Paper 1842, 31 p., https://doi.org/10.3133/pp1842AA.

Walker, B. L., and M. A. Schroeder. *Accepted*. Atypical primary molts and plumages in greater sage-grouse: implications for age classification.

PRESENTATIONS, WORKSHOPS, AND COMMITTEES

Aagaard, K. Habitat Selection Modeling Workshop. 05 June, 2020 – Fort Collins, CO. Video-conference.

Aagaard, K. Evaluating spatial patterns of avian and mammal populations. Conservation Over Virtual Interface Days (video-conference), April 29, 2020.

Aagaard, K., A. C. Behney, R. Y. Conrey, and J. H. Gammonley, J. Avian Research Updates. Northeast Region Biology Days, January 28, 2020.

Apa, A. D. Technical support, CPW Northwest region ruffed grouse translocation project. Prepared report: Colorado Parks and Wildlife. 2020. Ruffed Grouse Range Expansion: A summary of the 2016-2019 ruffed grouse trap and transplant plan: Current Creek, Utah to Garfield Creek, Colorado.

Apa, A.D. CPW science support, United States Fish and Wildlife Service Species Status Assessment Science Expert Team for Gunnison sage-grouse.

Apa, A.D. CPW science support, United States Fish and Wildlife Service Gunnison Sage grouse Recovery Team.

Apa, A. D. Faculty Committee member for M. S. degree candidate Alyssa Kircher, University ofWisconsin-Madison. Successfully defended her thesis in July 2020: Kircher, A. A. 2020. Greater Sage-grouse response to surface coal mine disturbance in northwestern Colorado. M. S. Thesis. University of Wisconsin-Madison.

Apa, A. D. CPW science support, CPW Terrestrial greater sage-grouse transplant project.

Apa, A. D., R. E. Barker and R. Scott Lutz. Columbian sharp-tailed grouse demographic response to habitat improvement. Conservation Over Virtual Interface Days (video-conference), April 28, 2020.

Apa, A. D., A. C. Behney, and **K. Aagaard.** CPW Animal Care and Use Committee.

Behney, A. C. Faculty co-advisor for M.S. degree candidate Joseph Wolske, University of Nebraska-Lincoln.

Behney, A. C. Pacific Flyway Study Committee meeting (video-conference), August 24-28, 2020.

Behney, A. C. Duck food availability and habitat use during the nonbreeding season in northeastern Colorado. Conservation Over Virtual Interface Days (video-conference), April 28, 2020.

Conrey, R. Y. Faculty Committee member for M.S. candidate Tyler Michels, University of Colorado Denver. Michels, T. J. 2020. Spatiotemporal patterns of habitat use during incubation by a uniparental shorebird in a heterogeneous landscape. M.S. Thesis, University of Colorado Denver, Denver, CO. Michels, T. J. 2020. Habitat use and species interactions impacting the mountain plover in the High Plains. 2016-2018 Summary Report.

Conrey, R. Y. IMBCR for PLJV (Integrated Monitoring in Bird Conservation Regions for Playa Lakes Joint Venture) Advisory Committee.

Conrey, R. Y., K. Aagaard, J. H. Gammonley, J. DeCoste, W. L. Kendall, L. Rossi, R. Sacco, A. Estep, and M. Smith. CPW's raptor research and monitoring: focus on southeastern Colorado. CSU Pueblo Wildlife Program Seminar, Pueblo, CO, January 30, 2020.

Conrey, R. Y., D. W. Tripp, E. N. Youngberg, and A. O. Panjabi. Plague management on prairie dog colonies maintains habitat for grassland passerines and raptors. Conservation Over Virtual Interface Days (video-conference), April 29, 2020.

Conrey, R. Y. Grassland birds, prairie dogs, plague, and black-footed ferret research and conservation. CSU class site visit: L. Pejchar's Bird Ecology and Conservation, Carr, CO, October 14, 2020.

Gammonley, J. H. Faculty Committee member for Ph.D. candidate Casey Setash, Colorado State University.

Gammonley, J. H. Central Flyway Waterfowl, Webless Migratory Game Bird, and Central Management Unit Dove Technical Committee meetings, North Platte, NE, February 3-6, 2020.

Gammonley, J. H. Central Flyway wing bee, Hartford, KS, February 16-21, 2020.

Gammonley, J. H. Central Flyway Waterfowl Technical Committee and Council meetings (video-conference), August 24-28, 2020.

Garbowski, M., **D. B. Johnston**, and C. S. Brown. Intraspecific variation of root traits under drought and competition. Ecological Society of America annual meeting (video-conference).

Johnston, D. B. Co-advisor for Ph.D. Candidate Magda Garbowski, Colorado State University, Fort Collins.

Johnston, D. B. Cheatgrass seeds and pothole seeding. *in* Society for Range Management annual meeting, Denver.

Johnston, D. B., and C. Andersen. Vegetation and deer responses to pinyon and juniper tree removal. Conservation Over Virtual Interface Days (video-conference), April 28, 2020.

Kendall, W. L., **R. Y. Conrey**, and **J. H. Gammonley**. Multistage nest survival: a hidden Markov approach when age or stage is uncertain. The Wildlife Society Annual Conference (video-conference), November 2020.

Rossi, L., and **R. Y. Conrey**. Raptor updates. Species Conservation Annual Coordination Meeting, Denver, CO, January 23, 2020.

Walker, B. L. The Wildlife Society Rusch scholarship committee member, reviewed and commented on scholarship applications; Cesar Kleberg Award committee member, reviewed and commented on TWS member lifetime achievement nominations.

Walker, B. L. Update on results from Piceance Basin and Hiawatha. Conservation Over Virtual Interface Days (video-conference), April 29, 2020.