

DISSERTATION

BIRD AND MAMMAL RESPONSE TO LARGE-SCALE HABITAT MITIGATION FOR
GAME SPECIES IN THE OIL AND GAS FIELDS OF NORTHWEST COLORADO

Submitted by

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ABSTRACT

BIRD AND MAMMAL RESPONSE TO LARGE-SCALE HABITAT MITIGATION FOR GAME SPECIES IN THE OIL AND GAS FIELDS OF NORTHWEST COLORADO

Habitat alteration to benefit game species has been underway for centuries. These practices are globally widespread and can take diverse forms – e.g., tree reduction to enhance forage for deer in the United States and burning moorlands in Scotland to increase habitat for wading birds. Yet the consequences of these practices for non-targeted animals are poorly understood. My dissertation focuses on the long- and short-term effects of mechanical habitat manipulation on birds and mammal communities in pinyon-juniper forests in the Piceance Basin, Colorado. The Piceance Basin is experiencing an unprecedented level of natural gas development and provides critical habitat for the largest migratory mule deer herd in the United States. Mature pinyon-juniper forest are thought to provide poor forage quality for mule deer, yet allowing natural disturbances in this ecosystem (e.g., wildfire) is incompatible with energy development. This unique set of circumstances has led land managers to use mechanical tree reduction to improve habitat for mule deer in the midst of one of the U.S.'s largest oil and gas fields.

My dissertation is organized as follows. In Chapter 1, I synthesize the global scientific literature on the effects of habitat manipulation intended to enhance habitat for game species on non-target wildlife; in Chapters 2 and 3, I assess the long-term effects of pinyon-juniper removal on bird and mammal communities, respectively; and in Chapter 4, I compare and contrast the effects of mechanical and natural disturbance on bird and mammals in pinyon-juniper woodland.

To quantify and compare studies that measured the effects of game management on non-

targeted animals, I surveyed the global literature and addressed the following research questions: 1) How many studies have investigated the effects of game management strategies on non-target species?, 2) What proportion of these studies document positive, negative or no effect of game management activities on non-target taxa?, and 3) What are the mechanisms underlying these effects? I found surprisingly few studies ($n = 26$) that evaluated the consequences of game management on other taxa. The outcomes of these studies illustrated that, through diverse mechanisms, game management can have either a positive, negative or no effect on non-target taxa. My analysis suggests that the explicit evaluation of the effects of game management on other species is rare but warranted, offering opportunities to advance ecological understanding and conservation of both target and non-target species. I propose a research agenda to fill knowledge gaps and catalyze a conversation about an approach to wildlife management that affects a large fraction of public and private land.

To partially address this research gap, I investigated whether tree removal to enhance habitat for mule deer and increase forage for livestock has altered bird and mammal communities in pinyon-juniper woodlands mechanically disturbed over 40 years ago, relative to sites that had not been mechanically disturbed (reference woodlands). Whether, and how, natural communities recover after human-induced habitat disturbance are critical questions facing ecologists and conservation practitioners. Forested ecosystems in the western U.S. have been the focus of tree reduction efforts for decades, with the intent of improving forage for livestock and economically important wildlife. Yet, the long-term consequences of tree removal on biodiversity are virtually unknown. To assess whether bird communities differ between historically disturbed and reference woodlands, and to determine if these differences are associated with particular habitat characteristics, I conducted bird and vegetation surveys where trees were mechanically removed

by chaining over forty years ago and woodlands that had not experienced large-scale disturbance. I evaluated differences in avian species richness, diversity, community evenness, and used a Bayesian hierarchical approach to compare density between historically disturbed sites and reference sites. I found that tree reduction catalyzes a long-term change from dense pinyon-juniper woodlands to sagebrush scrub, fundamentally altering bird community composition. Disturbed sites were used by fewer species (J-evenness was 0.75 and 0.83 for disturbed and reference sites respectively) and largely dominated by shrubland-obligate birds – e.g., Green-tailed Towhee (*Pipilo chlorurus*); whereas, the most common birds in reference sites were woodland birds – e.g., Mountain Chickadee (*Poecile gambeli*). Further, bird densities were markedly different between disturbed sites and reference sites. Densities of many species were influenced by specific vegetative characteristics, such as mean tree diameter, that could be factored into management decisions.

I assessed whether mammal habitat use differed between reference pinyon-juniper woodlands and stands that were mechanically disturbed by chaining more than 40 years ago using remotely triggered wildlife cameras in historically chained sites ($n = 22$) and reference sites ($n = 22$). My results demonstrate marked differences in habitat use between chained sites and reference sites for most detected mammal species. Bobcat, mountain lion, American black bear, golden-mantled ground squirrel, and rock squirrel all showed a negative response to historically chained sites, indicating long-term effects of tree removal on these species. In contrast, habitat use of chipmunk, mountain cottontail, and coyote did not differ between chained and reference sites. Similar to birds, mammal habitat use of most species was influenced by specific vegetative characteristics, such as proportion of tree cover, which could be factored into management decisions.

Finally, I evaluated the ecological consequences of simulating natural disturbance as habitat mitigation for game species in a landscape undergoing energy development. Specifically, I investigated whether birds and mammals responded differently to mechanical tree reduction and natural disturbance (wildfire) in the Piceance Basin. My research objectives were to evaluate the differences in bird and mammal community composition, bird densities, and mammal habitat use immediately following mechanical tree removal and wildfires. I found little difference in species composition between mechanically disturbed sites and wildfire sites. However, I found marked differences in bird densities and mammal habitat use between mechanically disturbed woodlands and woodlands that were subject to wildfires. For example, wildfires had a strong positive effect on cavity nesting birds (e.g., Hairy Woodpecker, *Picoides villosus*), but a strong negative effect on shrub nesting birds (e.g., Green-tailed Towhee, *Pipilo chlorurus*). Bobcat (*Lynx rufus*) and coyote (*Canis latrans*) habitat use had a positive relationship with wildfire, but mountain lion (*Puma concolor*) habitat use showed a negative relationship with wildfires. No mammal species in my study showed a positive response to mechanical disturbance. I demonstrate that mechanical tree reduction – intended to emulate natural disturbances – has unintended consequences for birds and mammals. Thus, I suggest that future management actions that result in large-scale tree removal should explicitly measure intended and unintended effects on birds, mammals, and other taxonomic groups.

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PREFACE

This dissertation is ordered by chapter, in which each chapter is intended for publication as an article in a peer-reviewed journal. Therefore, formatting, language and tense may differ between chapters. At this time, Chapter 1 is published in *Biological Conservation*, and Chapter 2 is in review at *Journal of Avian Biology*. Chapter 3 will be submitted to *Forest Ecology and Management* or a similar journal, and Chapter 4 is intended for *Ecological Applications* or a similar applied ecological journal. Because all articles will have at least one co-author, I use the plural pronoun “we” throughout. The titles and full authorship for each chapter/manuscript are listed below.

Chapter 1. Improving habitat for game animals has mixed consequences for biodiversity conservation

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Chapter 2. Woodland reduction leads to long-term state change in bird communities

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Chapter 3. Pinyon-juniper removal has long-term effects on mammals

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Chapter 4. Mechanically simulating natural disturbance amidst natural gas development has
mixed consequences for woodland birds and mammals

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CHAPTER ONE

IMPROVING HABITAT FOR GAME ANIMALS HAS MIXED CONSEQUENCES FOR BIODIVERSITY CONSERVATION

INTRODUCTION

In the aftermath of early industrialization, many animal populations declined globally due to habitat loss, overexploitation, and pollution (Leopold 1933, Vitousek et al. 1997, Sotherton 1998). As awareness of this loss reached policy-makers and the public, preserving and improving habitat quality for wildlife, particularly hunted species, became a priority in North America (Leopold 1933), Europe (Phillips 2004) and colonial-ruled countries in Africa (Phillips 2004). In many cases, manipulating natural communities to improve habitat quality for these species has been remarkably successful at reversing population declines among harvested species. For example, at the turn of the century land preservation (e.g., U.S. National Wildlife Refuge system), game laws (e.g., U.S. Lacey Act), and habitat management (e.g., forest restoration) stabilized many populations of declining North American mammals (Leopold 1933, U.S. Fish and Wildlife Service 2006). Similarly, reinstating natural processes (e.g., prescribed fire) in heather moorlands – has restored populations of commonly hunted wading birds throughout the UK (Tharme et al. 2001, Brennan and Kuvlesky 2005, Pack et al. 2013). Habitat altering practices are widely implemented and well funded across the globe. For example, 58% of the land area in Scotland is managed for hunting (HUNT 2015a), hunting estates cover approximately 80% of the Spanish territory (HUNT 2015c), and hunting influences the management of 94% of the land in Slovenia (HUNT 2015b).

More recently, however, both game and non-game species are faced with novel anthropogenic pressures, such as climate change (Parmesan and Yohe 2003), as well as rapid rates of habitat loss and fragmentation from energy development (Northrup and Wittemyer 2013, Jones et al. 2015) and urbanization (McKinney 2002). Due to the synergistic effects of these changes (Foley et al. 2005) and the continued practice of manipulating habitat for game species across private and public lands, we argue that evaluating the effects of game management on biodiversity is warranted.

Hunting and conservation

We recognize that hunting provides diverse and substantial economic (Schulz et al. 2003, PACEC 2006, U.S. Fish and Wildlife Service 2014), social (Mangun 1992, Heberlein et al. 2008), and ecological (Lindsey et al. 2006) benefits, and that habitat management for hunted species has advanced the fields of ecology and conservation biology (Leopold 1933). We are not advocating that hunting be reduced or prohibited on either public or private lands. Nor do we set out to diminish the dedication of the large number of hunter-based special interests groups and state and federal agencies to the conservation of both game and non-game animals (Lebbin et al. 2010, North American Bird Conservation Initiative 2014). Rather, because game management has the potential to have a significant impact on biodiversity by altering habitat structure, food availability and intra- and inter-specific interactions on large tracts of land (Leopold 1933, Arroyo and Beja 2002), we suggest conservationists objectively examine the ecological consequences of the game management paradigm that remains so prevalent.

Funding for game and non-game species

Funding for the management of game species often exceeds spending on all other species, including those that are threatened and endangered. In the U.S., the state of Washington spends

approximately \$18.6 million biennially to maximize hunting opportunities and sustain game animal populations – compared to \$13.3 million on non-game species protection (Anderson and Larson 2013). The state of Minnesota budgeted \$206.2 million in 2014-2015 for game management and the protection of game species (Minnesota Department of Natural Resources 2013), whereas non-game wildlife management is funded through a \$179.8 million budget that is split among parks and zoos (Minnesota Department of Natural Resources 2013). These examples illustrate funding scenarios for game management and non-game management in just two U.S. states; these values are likely to vary substantially among hunted species and regions of the world.

Because traditional funding support for wildlife conservation has come almost exclusively from user fees and taxes on goods for hunting (Mangun 1992); there is an enormous economic incentive for state and federal agencies to manage for game species on public lands (Draycott et al. 2008). Hunting licenses in the U.S. totaled approximately \$790 million in 2013 (US. Fish and Wildlife Service 2013), and the special U.S. excise taxes and duties on hunting gear under the Wildlife Restoration Act generate approximately \$550 million annually (Corn and Gravelle 2013). In 2014, the U.S. distributed \$1.1 billion dollars from these excise tax revenues to state fish and wildlife agencies for fish and wildlife management (U.S. Fish and Wildlife Service 2014). In the UK, hunters spend approximately \$16.3 million annually on hunting licenses and firearm certificates (PACEC 2006), and in Iceland, hunting generates \$83.8 million annually from reindeer licenses alone (Matilainen and Keskinarkaus 2010). Hunting upland game birds in Scotland is reported to contribute \$365 million annually to the Scottish economy (Irvine 2011). Hunting tourism results in approximately \$68.3 million of revenue annually in South Africa, \$27.6 million in Tanzania, \$18.5 million in Zimbabwe and \$12.6 million in

Botswana (Lindsey et al. 2006, Pack et al. 2013). Further, private landowners have an economic incentive to manage their lands specifically for game species, because recreational hunting by paying clients provides important supplemental income (Sage et al. 2005).

Objectives

Although land ownership and funding mechanisms vary (Pack et al. 2013), strategies to increase the populations of hunted species have been implemented for centuries on every continent except Antarctica (Leopold 1933, Redford and Bodmer 1995, Arroyo and Beja 2002, Damm 2008, Pack et al. 2013). Despite the long history, ubiquitous use, and global relevance of these practices, information on the extent of habitat manipulation is largely lacking, making it difficult to quantify the ecological consequences of game management (Arroyo and Beja 2002). We systematically surveyed the scientific literature to evaluate the state of knowledge on this topic. Specifically, we address the following research questions: 1) How many studies have investigated the effects of game management strategies on non-target species?, 2) What proportion of these studies document positive, negative or no effect of various game management activities on non-target taxa?, and 3) What are the mechanisms underlying these effects? We draw on this literature review to identify potential sources of conflict and synergy between game management and biodiversity conservation, and we conclude by discussing priorities for research, policy and practice.

APPROACH

To quantify the number of previous papers on this topic, and the frequency of results that demonstrated positive, negative or no effect of game management on non-target taxa, we searched the scientific and grey literature using multiple combinations of relevant keywords (see Appendix 1 for keywords and search criteria). We limited the scope of our search to empirical

studies that investigated the effects of habitat management for terrestrial game species (native and introduced) on native terrestrial animals. We define direct effects as the direct and unmediated impact a management activity has on the demography or behavior of an individual species or group of species. In contrast, indirect effects of habitat alteration on a species/group are mediated through changes in abundance of another taxa; these can include apparent competition, trophic cascades (predator-prey interactions), or a change in the physical or chemical properties of the habitat by this species/taxa (Moon et al. 2010).

RESULTS AND DISCUSSION

Number and nature of studies

Our examination of the literature found remarkably few articles ($n = 26$) that directly evaluated the effect of game management practices on non-targeted wildlife (Fig. 1.1, Table A1). These articles measured the effects of game management on diverse non-targeted taxa that included birds (81%), mammals (23%), herptiles (4%) and/or arthropods (8%). A total of 43 relationships were reported; 40% of these effects were positive for non-targeted species, 37% were negative, and the remaining 23% found no effect (Table A1). In the following sections we draw on these studies to highlight several mechanisms through which game management affects non-target animals.

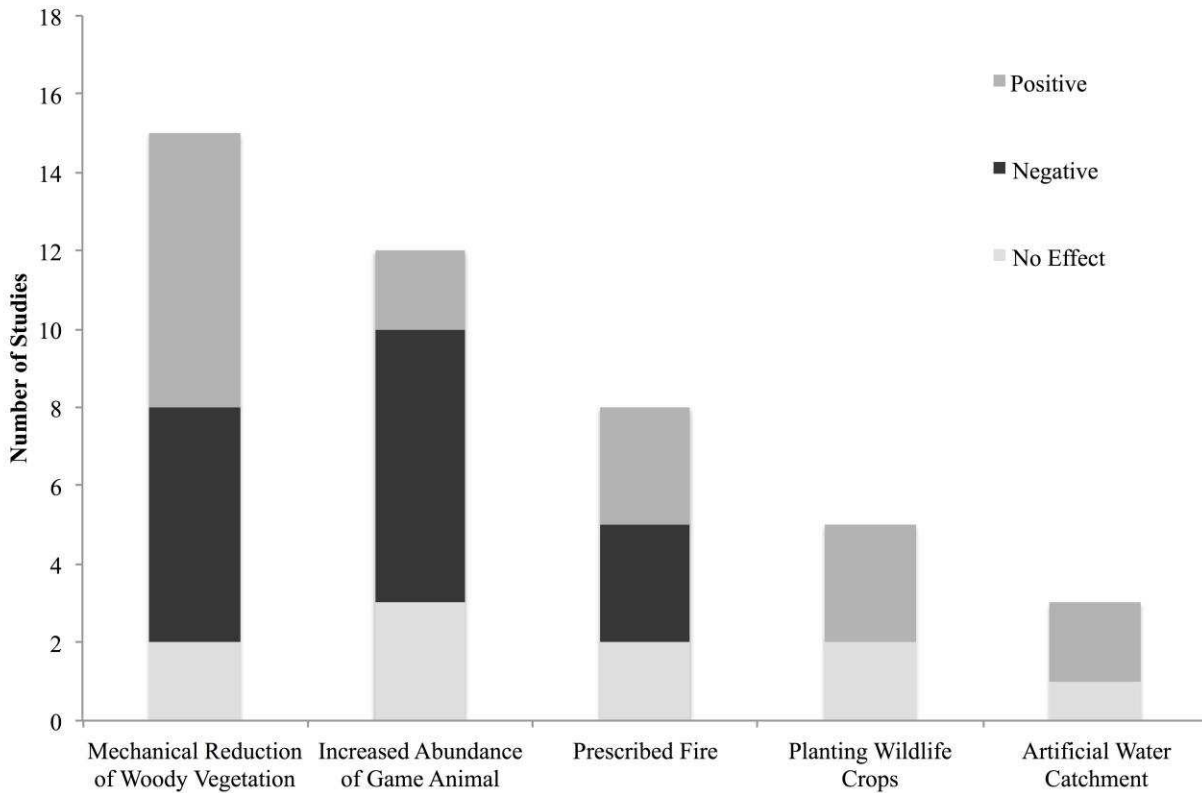


Fig. 1.1 The number of studies that examine the effect of game management on non-target species and met the selection criteria for our review (Appendix 1). The frequency of studies reporting positive, negative or no effect of several types of game management on non-target species are illustrated. See Table A1 for a full list of studies and taxonomic groups.

Positive effects

Managing land for game species has several documented shared benefits for non-targeted species. Many protected areas and the full suite of wildlife they support would have been degraded in the absence of hunting and active land preservation for the benefit of game species (Tharme et al. 2001). In addition, some management practices that closely mimic ecological processes – e.g. prescribed fire and mechanical removal of forest cover as an alternative to natural wildfires – have demonstrated positive effects on animal communities adapted to natural disturbance regimes (O'Meara et al. 1981, Tapper 1999, Arroyo and Beja 2002, Radke et al. 2008).

Previous studies (Table A1) demonstrate that managing specifically for game species can act as an umbrella to conserve habitat for a large number of non-game species (Karl et al. 2005, Hanser and Knick 2011). For example, Hanser and Knick (2011) found that maintaining sagebrush-dominated plant communities as habitat for greater sage grouse (*Centrocercus urophasianus*) in the Western U.S. will likely protect habitat for 13 non-game passerine birds. Similarly, Idaho's Wildlife Management Areas provide valuable habitat for a variety of non-game species – i.e. reptiles, birds and non-game mammals (Karl et al. 2005). These benefits to non-targeted species are likely a function of the broad range of habitats that are protected within those areas (Hanser and Knick 2011), rather than the consequences of specific management practices.

In some cases, habitat alteration to create new vegetation communities that benefit game species – e.g., woodlands converted to grasslands – also benefits species that prefer the new habitat characteristics resulting from the management practice (Table A1). For example, removing shrub species from wetlands in the Great Lakes region of the U.S. maintains high-quality habitat for game birds, such as sharp-tailed grouse (*Tympanuchus phasianellus*), and simultaneously increases the abundance of non-game bird species that require open wetland habitat, such as Le Conte's sparrow (*Ammodramus leconteii*) and sedge wren (*Cistothorus platensis*) (Hanowski et al. 1999).

Artificial supplementation of food and water has also had potential benefits for non-targeted wildlife species (Table A1). Planting game crops – non-agricultural crops that attract game species – is a common tool employed by European farms to increase and diversify farm income through hunting (Sage et al. 2005). Studies in Europe found that farms that planted “game crops” had a positive effect on non-game birds, more so than nearby conventional farms

(McGee 1976, Parish and Sotherton 2004, Caro et al. 2015). Construction of water catchments is a common game-habitat improvement technique throughout the southwestern U.S. (Lynn et al. 2008). In Arizona, native bats, mammalian predators, and rodents were observed using water catchments more often than the game species for which they were designed, such as mule deer (*Odocoileus hemionus*), Gambel's quail (*Callipepla gambeli*), and dove (*Zenaida* spp.) (O'Brien et al. 2006). Equipped with a better understanding of the factors associated with shared benefits of game management for non-targeted species, land managers may be able to strategically implement management practices that account for these factors – an approach that could increase populations of hunted species while also protecting the full suite of biodiversity under their stewardship.

Negative effects

Habitat manipulation to benefit game species can have direct or indirect negative effects on non-targeted species through diverse mechanisms, including competition for resources, trophic cascades, and inter-specific interactions (Table A1). For example, the increased abundance of wild boar (*Sus scrofa*), red deer (*Cervus elaphus*), and aoudad sheep (*Ammatragus lervia*), decreased the available resources for closely related native species of high conservation concern in Spain and across the Iberian Peninsula (Acevedo et al. 2007, Lozano et al. 2007). In the UK, Newson et al. (2012) found that the increase of three commonly hunted deer populations – Reeves' muntjac (*Muntiacus reevesi*), roe deer (*Capreolus capreolus*), and fallow deer (*Dama dama*) – corresponded with substantial declines in the abundance of chiffchaff (*Phylloscopus collybita*), common nightingale (*Luscinia megarhynchos*), willow warbler (*Phylloscopus trochilus*), willow tit (*Poecile montanus*) and song thrush (*Turdus merula*). These five species are associated with dense understory vegetation that was significantly reduced due to browsing

by the ungulates. Similarly, the overabundance of elk at the National Elk Refuge in Wyoming, USA, increased browsing pressure and decreased habitat availability for migratory shorebirds and songbirds that depend on vegetation cover (Matson 2000).

Removing or reducing tree cover and shrub cover has shown to have negative effects on non-target species that prefer woodland and shrubland habitats (O'Meara et al. 1981, Yahner 1984, Gruver and Guthery 1986, Kozicky and Fulbright 1991, Yahner 1993, Brown et al. 2000, Tharme et al. 2001). For example, mosaic-like clear-cutting of forest for ruffed grouse (*Bonasa umbellus*) management in Pennsylvania decreased the abundance of red-eyed vireo (*Vireo olivaceus*) and ovenbird (*Seiurus aurocapilla*) – both forest obligate birds (Yahner 1984, 1993). Although, habitat alteration to create habitat for game species can simultaneously benefit non-targeted species that prefer the novel habitat, these studies demonstrate that habitat alteration can have a negative effect on species that required the habitat that has been lost or altered.

No effect

In some cases game management practices had no significant detectable effect on non-targeted species (Table A1). For example, Radke et al. (2008) found no short-term effect of prescribe fire on lizard abundance in central Texas, and Petersen and Best (1987) found no positive or negative effects from small mosaic-like prescribed fires on non-target bird species that preferred open habitats. In both studies fire was used as a management tool to improve habitat conditions for game species. As aforementioned, creating artificial water catchments and

planting wildlife crops were shown to have positive effects on non-targeted species (McGee 1976, Parish and Sotherton 2004, O'Brien et al. 2006, Caro et al. 2015), but also had no effect on some non-targeted species in studies by Lynn et al. (2006) and Stoate (2002).

RECOMMENDATIONS FOR SCIENCE, POLICY, AND PRACTICE

Land managers are frequently faced with the challenge of managing for both game species and non-game species with limited funding and limited access to information needed to make science-based decisions (Noon et al. 2009). Our examination of the literature suggests that few studies explicitly measure how game management practices affect non-game wildlife. Greater scientific scrutiny of game management practices by game and non-game scientists could provide greater shared benefits to hunters, hunted species, and other biodiversity.

Priorities for Research

To remedy these knowledge gaps, we offer a shortlist of ecologically intriguing and policy relevant questions intended to guide future research on this topic (Table 1.1). In addition to direct effects, it is likely that mechanisms, such as competitive interactions, predator-prey dynamics, trophic cascades, and changes in ecosystem function (Osmond et al. 2004, Levin et al. 2009), are driving the interactions between game management and non-targeted species. Applying principles of community ecology and ecosystem science to game management research provides an unprecedented opportunity to advance science while also building the foundation for well-informed land management practices.

Table 1.1 Priorities for future research: questions that will enhance understanding of the potential unintended consequences of game management practices on non-targeted species.

A research agenda for assessing the consequences of game management on biodiversity

Through what mechanisms does habitat management that increases the abundance of a single game species have direct or indirect effects on non-target species?

Are these mechanisms predictable based on the characteristics of the game and non-game species, and/or the characteristics of the ecosystem?

Does food and water supplementation for game species have broad co-benefits for non-game species, or do some species (e.g. introduced plants or animals) benefit at the expense of others?

Does habitat manipulation designed to mimic natural disturbance (e.g. mechanical clearing or prescribed burns in lieu of wildfire) have differential effects on game and non-game species?

Do the long-term effects of habitat manipulation for game species on non-targeted animal communities differ from measured short-term effects?

Is there a threshold in the extent or intensity of habitat manipulation, which precipitates a state-shift in the community composition of non-game species?

Are hunted species effective surrogate species? Does large-landscape conservation designed to benefit hunted species provide sufficient viable habitat for native non-game species?

A study reporting that habitat management for endangered non-game species provided complimentary benefits for game species suggest that research on this related topic is also warranted (Masters et al. 1996). In western Arkansas, pine-bluestem habitat restoration and red cockaded woodpecker (*Picoides borealis*) management (low intensity prescribed fire) increased preferred forage of white-tailed deer (*Odocoileus virginianus*; Masters et al. 1996). This outcome suggests that there may be additional untapped opportunities for management actions that are mutually beneficial for both games species and species of highest conservation concern.

Finally, obtaining publically available data on funds spent on management activities for particular game and non-game species is often difficult, and these values are likely to vary substantially by management activity, taxonomic group, and region (Mangun 1992, Anderson and Larson 2013). We recommend compiling and comparing these data in regions where both game management and biodiversity conservation are a priority. This, combined with a better understanding of the ecological costs and benefits of managing for hunted species, would enable land managers and society to more fully evaluate public investment in game and non-game management.

Revisiting funding sources for conservation

We suggest that conservationists revisit available funding streams for conservation. Hunters and anglers traditionally pay the user fees and taxes that support wildlife programs. Today, however, there are less people engaged in recreational hunting, as evident in a steady decline of license sales in the U.S. (Mangun 1992, Brown et al. 2000, Enck et al. 2000, Schulz et al. 2003) and throughout Europe (Heberlein et al. 2008). In contrast, an increasingly large number of land users participate in non-consumptive wildlife recreation. For example, in the U.S., 13.7 million people consider themselves “hunters”, compared with the 71.8 million people that consider themselves “wildlife watchers” (U.S. Fish and Wildlife Service 2012). Nearly half (48%) of all Americans participate in an outdoor recreational activity (not including hunting) at least once per year (Cordell 2012). Similarly, participation in non-consumptive wildlife recreation has been steadily increasing in Europe over the last 15 years (Bell et al. 2007). To reflect these national and global trends, one alternative funding stream for wildlife management could be a non-consumptive tax on recreational goods (e.g., the proposed U.S. Teaming with Wildlife Act of 2009). While such a tax may not be viable or desirable in every context,

exploring new ways to diversify the funding stream for conservation could reduce pressure on public and private landowners. In addition to managing for game species, land managers would have additional resources to direct towards the diverse ways that society values natural, intact ecological communities.

Mixed consequences of game management: implications for practice

Previous studies demonstrate that all types of game management have mixed consequences for non-targeted species (Fig. 1.1; Table A1). For example, removing shrub species from wetlands in the Great Lakes region of the U.S. to maintain habitat for sharp-tailed grouse (*Tympanuchus phasianellus*) increased the abundance of non-game bird species that require open wetland habitat, such as Le Conte's sparrow (*Ammodramus leconteii*) and sedge wren (*Cistothorus platensis*), but decreased the abundance of birds that prefer shrubland habitats – i.e. veery (*Catharus fuscescens*), gray catbird (*Dumetella carolinensis*), Nashville warbler (*Oreothlypis ruficapilla*), yellow warbler (*Setophaga petechial*), brown-headed cowbird (*Molothrus ater*) and American goldfinch (*Spinus tristis*). This study and the cumulative findings of our review, demonstrate that game management can have the unintended effect of benefiting some species at the expense of others. Thus, the benefits gained by improving habitat for game species should be weighed against the predicted impacts to the species of greatest conservation

concern in a particular ecoregion (Arroyo and Beja 2002). Incorporating more consistent monitoring of non-target effects into game management projects would help managers detect, and where feasible and appropriate, mitigate for unintended consequences on biodiversity.

CONCLUSION

For decades, the assumption that land management practices that benefit hunted species also positively affect all wildlife species has been relatively unexamined (Johnson et al. 1994). We found that fewer than 30 studies, globally, have addressed this topic, and the direction of the effects they report are not consistent (Appendix 1). In light of limited funding for biodiversity conservation (Primack 2010) and alarming rates of extinction (Pimm et al. 2014), understanding how game management affects other species, and particularly those of conservation concern, is critical. Habitat management intended to benefit hunted species should be designed to experimentally test the consequences of these actions on both game and non-game species. By understanding and acknowledging costs and benefits to diverse species, public and private landowners can more effectively implement management practices that collectively increase populations of hunted species while also protecting the full suite of biodiversity.

CHAPTER 2

WOODLAND REDUCTION LEADS TO LONG-TERM STATE CHANGE IN BIRD COMMUNITIES

INTRODUCTION

Deliberate and large-scale tree removal to increase forage for livestock or economically important wildlife has been common and widespread for centuries (Aro 1971, Lewis et al. 1982, Yahner 1984, Fuschs et al. 2015). These practices are particularly prevalent in the western United States where forested lands are converted to shrub or grasslands to increase forage quantity and quality (Aro 1971, Terrel and Spillett 1975, Evans 1988, Miller and Wigand 1994). Trees are generally removed using either prescribed fire or mechanical techniques (Aro 1971, Miller and Wigand 1994, Redmond et al. 2013). Historically, chaining was the most widely used method to mechanically remove forest cover (Aro 1971). Chaining involved attaching the ends of heavy anchor chains to two bulldozers and dragging the loop of the chain through the trees in a “U” or “J” shaped pattern to uproot trees and shrubs (Aro 1971, Sedgwick and Ryder 1986, BLM 2008). Chaining has been successful at reducing tree cover – killing a majority of older, larger trees in a stand – and increasing herbaceous forage for livestock and economically important wildlife (Aro 1971).

Recent declines in bird and mammal populations dependent on open habitats, such as Greater Sage Grouse (*Centrocercus urophasianus*; Schroeder and Baydack 2001, Knick and Connelly 2011) and Mule Deer (*Odocoileus hemionus*; Bergman et al. 2015), and increased concerns about wildfire has renewed tree reduction efforts. In some cases, chaining is still employed (Redmond et al. 2013), while elsewhere it has been replaced with other mechanical tree removal methods (e.g., hydro ax) that have similar objectives and outcomes (Wästerlund and

Hassan 1995). The scale and intensity of tree removal is expected to increase as land managers are tasked with meeting multiple objectives, including fire prevention and enhancing habitat for hunted species and those of conservation concern in areas subject to rapid urbanization and energy development (Connelly et al. 2000, Redmond et al. 2013, Bergman et al. 2015). Although removing tree cover has been a common land management practice for decades, and is predicted to increase in frequency and intensity (Redmond et al. 2013), the long-term effects of tree removal on bird communities are virtually unknown.

Understanding the consequences of these practices is particularly important in the pinyon-juniper (P-J) ecosystems of the western U.S. During the last half-century, P-J ecosystems have been a major focus of both forest conservation and tree reduction due to their large spatial extent, the ecosystem services they provide, and their natural or human-induced encroachment on shrubland and grassland ecosystems (Tausch and Tueller 1977, Miller and Wigand 1994, Redmond et al. 2013). P-J covers some 40 million hectares, and collectively is the third largest vegetative community in the United States (Romme et al. 2009). Pinyon-juniper woodlands offer valuable resources – supplying food and cover for woodland-dependent wildlife species, food and fuel for humans, and forage for livestock (Schott and Pieper 1987, Romme et al. 2009). However, both pinyon and juniper trees have been expanding into grasslands and shrublands for the past 150 years (Romme et al. 2009). The mechanisms for P-J expansion are not well known, but may include recovery from past natural disturbances, Holocene range expansion, livestock grazing, fire suppression, and the effects of climatic variability and rising atmospheric CO₂ (Miller and Wigand 1994, Romme et al. 2009). Because P-J expansion into grasslands and shrublands reduces forage for livestock and hunted species and decreases the amount of habitat for rare shrubland species – e.g., Greater Sage Grouse and Brewer’s Sparrow (*Spizella breweri*)

– tree reduction and thinning at both the margins and interior of P-J stands is an important component of current land management activities throughout the western U.S. (Aro 1971, Terrel and Spillett 1975, Evans 1988, Miller and Wigand 1994, Bergman et al. 2015).

Large-scale tree removal has the potential to have a variety of ecological consequences for plant and animal communities. Because all or most trees are removed, P-J woodlands are replaced with open grassland and shrubland habitat (Tausch and Tueller 1977). Removing the majority of forest cover is likely to have an impact on the habitat use of forest-dwelling birds that respond to changes in vegetation structure and density (MacArthur and MacArthur 1961). In slow recovering systems, such as P-J ecosystems (Schott and Pieper 1987), these significant changes in vegetative structure may have a lasting effect on bird community composition and habitat use. Further, removing forest cover within contiguous stands of P-J results in forest fragmentation. Fragmentation generally decreases the species richness, diversity and abundance of bird species (Gascon et al. 1999).

Inference about changes in species composition following habitat disturbance are typically drawn from short-term surveys (e.g., 1-3 years; Debinski and Holt 2000), which may or may not predict long-term effects on community dynamics (Stouffer et al. 2011). Previous studies investigating the effects of P-J removal on animal communities have generally occurred within 1-4 years of the initial disturbance (Bombaci and Pejchar 2016). The few studies on the short-term effects of pinyon-juniper removal on animal communities have found that P-J removal has negative consequences on forest-obligate species (Bombaci and Pejchar 2016). For example, habitat use by all bird species was 10x greater in P-J woodlands, and forest-obligate species (i.e. Mountain Chickadee, *Poecile gambeli*; White-breasted Nuthatch, *Sitta carolinensis*; and Black-throated Gray Warbler, *Setophaga nigrescens*) were rarely observed in mechanically

disturbed sites (Sedgwick and Ryder 1986). O'Meara et al. (1981) demonstrated that breeding bird densities were more than double in reference woodland compared to mechanically disturbed areas, and found that mechanically disturbed and reference areas had no breeding birds in common. The abundance of generalist rodent species (e.g., deer mouse, *Peromyscus maniculatus*) have shown to increase immediately following P-J removal (Baker and Frischknecht 1973, O'Meara et al. 1981, Sedgwick and Ryder 1986). However, P-J removal has also had negative consequences for pinyon-juniper specialist, such as pinyon mouse (*Peromyscus truei*; Baker and Frischknecht 1973). Because long-term effects on species, particularly those that are rare and long-lived, can go unseen in short-term studies (Laurance et al. 2002, Laurance et al. 2011), these studies provide the unique opportunity to revisit historically disturbed areas and compare short and long-term changes to natural communities.

Over the long term (40+ years), P-J removal can lead to more perennial grasses, reduced tree cover, and increased shrub cover (Yorks et al. 1994, Redmond et al. 2013). These changes in vegetation structure and cover could have cascading effects on bird communities, which are often considered strong indicators of ecological integrity (Schmiegelow et al. 1997). Birds also provide ecological services such as seed dispersal and pollination and can play an important role in structuring plant communities (Wall 1997, Wunderle Jr 1997, Pejchar et al. 2008, Garcia et al. 2010). Thus, understanding the long-term effects of the widespread practice of tree removal on birds is both ecologically interesting and has important conservation implications.

We investigated whether tree removal to increase forage for livestock and enhance habitat for mule deer has altered bird communities in woodlands mechanically disturbed over 40 years ago, relative to reference woodlands. This study occurred in a P-J ecosystem in the Piceance Basin of northwest Colorado, USA. Our research objectives were to 1) evaluate

differences in species richness, community composition and community evenness between historically disturbed sites and reference sites, 2) compare bird densities between historically disturbed sites and reference sites, and 3) identify the vegetation characteristics associated with differences in bird densities. We predict that species richness, community composition, community evenness, and densities will differ between historically disturbed sites and reference sites, and that the influence of habitat characteristics will vary in their magnitude and direction depending on species life history strategies (i.e. foraging and nesting guilds and habitat preference). Our findings provide greater insight into the long-term consequences of human-induced habitat disturbance for bird communities.

MATERIALS & METHODS

Study site

This study was conducted in the Piceance Basin, in northwest Colorado, U.S.A. on land owned and managed by the Bureau of Land Management. Our study area was bounded by U.S. Highway 40 to the north, Colorado State Highway 139 to the west, the Roan Plateau to the south and Colorado State Highway 13 to the east (Fig. 2.1). Dominant land use activities in the area include oil and gas extraction and domestic livestock grazing (Northrup et al. 2015). Our study area ranges in elevation from approximately 1500 to 2400 m. The topography consists of high plateaus and deeply incised valleys. Precipitation ranges from 30 cm per year at lower elevations to 60 cm per year at higher elevations (Carlson and Cringan 1975). Woodlands are dominated by two tree species, pinyon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*) (Sedgwick 1987). In lower elevations, *J. osteosperma* dominates the overstory, and the understory consists of antelope bitterbrush (*Purshia tridentate*) and mountain mahogany (*Cercocarpus montanus*) (Sedgwick 1987). At higher elevations, *P. edulis* dominates the

overstory, and most of the low elevation grasses and forbs are present in greater proportions, along with arrowleaf balsamroot (*Balsamorhiza sagittata*) and lupine (*Lupinus* spp.). Typical high elevation shrubs include big sagebrush (*Artemisia tridentata*), rabbitbrush (*Chrysothamnus* spp.), *P. tridentata*, *C. montanus*, common chokecherry (*Prunus virginiana*), and Saskatoon serviceberry (*Amelanchier alnifolia*) (O'Meara et al. 1981, Sedgwick 1987).

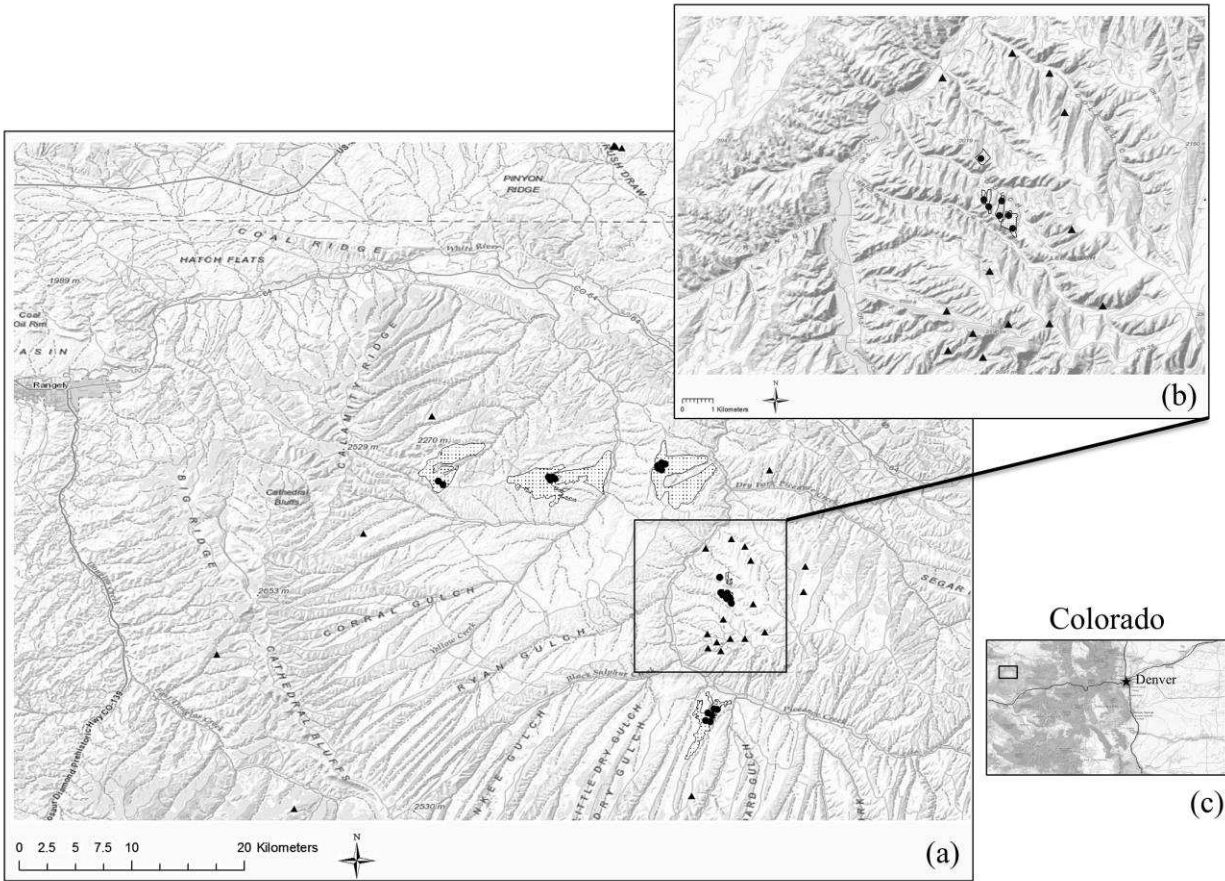


Fig. 2.1. Location of historically disturbed and reference study sites in the Piceance Basin in northwest Colorado, U.S. Stippled polygons represent historically chained (“disturbed”) areas. Black circles mark disturbed sites and black triangles mark reference sites. Map (a) shows the full extent of study area, inset (b) illustrates the distribution of disturbed and reference sites in and around the cluster of smaller disturbed areas, and inset (c) shows the location of the study site within the state of Colorado, USA (Basemap Source: ESRI, Redlands, CA, USA).
Study design

We established sampling sites in historically chained P-J woodlands (“disturbed” sites) and sampling sites in woodlands that were never mechanically disturbed (“reference” sites). We identified areas that had been chained in the 1950’s-1970’s using local knowledge from land managers and aerial imagery. We then visited each location to verify that the site had not been disturbed since initial chaining by prescribed fire, wildfire or mechanical tree removal. We confirmed that these areas had not been disturbed since the initial chaining by observing the presence of many large, decaying, fallen trees and the absence of charred debris indicating fire. Nine historically disturbed areas embedded in a matrix of P-J woodlands and ranging in size from 3 to 1243 ha were identified. Using Geographic Information Systems (GIS), we established 25 sampling sites throughout these disturbed areas. We placed our first sampling site within each disturbed area by picking a random but accessible location in the approximate center of each area. We then placed additional sampling sites in each cardinal direction, such that the sites were at least 250 m apart. Due to the irregular shape of some disturbed areas, some sites were located near undisturbed forest (~35 m). However, only birds detected within the disturbed area were counted. Because we began selecting sites in the smallest disturbed areas first, our design allowed for one site in each of the smallest areas and up to 6 sampling sites in the largest areas (Fig. 2.1a).

Reference sampling sites ($n = 50$) had been previously established across the study area for an ongoing bird-monitoring program. All reference sites were randomly placed on the landscape using GIS and were buffered from all forms of known anthropogenic disturbance (e.g., historically chained areas, energy well pads, roads) by at least 200 m. Each reference site was ground-truthed to verify that it was within P-J woodlands. To ensure a similar sampling effort between the disturbed and reference sites (Magurran 2004), we used stratified random sampling

to select 25 reference sites from this larger set of reference sites using GIS (Fig. 2.1). Because five of the nine disturbed areas were clustered together on the landscape, reference sites were stratified such that 13 sites were randomly selected from the vicinity of the cluster of disturbed areas, and 12 were randomly selected from the overall study area which encompassed the more geographically dispersed disturbed areas (Fig. 2.1a,b).

Bird surveys

Birds were surveyed by conducting 5-minute point counts at each of the 25 disturbed sites and 25 reference sites (Dunn et al. 2006). Each site was surveyed 4 times/year between April-June for two years (2013 – 2014) by 3-5 trained observers. All birds were detected visually or aurally and their distances from the point count station were recorded. All surveys were conducted between 30 minutes after sunrise and no later than 1230, and starting times were rotated among locations and surveys. Surveys were discontinued during periods of fog, rain or high winds (>3 on Beaufort scale).

Vegetation surveys

To assess the relationship between bird densities and habitat characteristics, we measured a variety of vegetative parameters. In 2013, we sampled vegetation in 10x10-m plots, offset from each point count station by ~5 m in a random direction, to determine plant composition and cover (McElhinny et al. 2005). Trees (live and dead) were defined as individuals with a height >1 m (Romme et al. 2009). Species, tree height, crown area, diameter at breast height (DBH), and condition (i.e. live, dead, large-snag broken above 1 m, small-snag broken below 1 m, log, cut stump) were recorded for each tree in the 10x10 m plot following the methods used in Huffman et al. (2012). Percent forest cover was calculated by dividing total area of canopy cover by plot size (100 m²). We estimated an index of forest stand age by dividing the total DBH of all

trees within the plot by the plot size (DBH/plot; modified from McElhinny et al. 2005). In 2013 and 2014, all vegetation was classified into six height classes (0-0.5 m, 0.6-1 m, 1.1-2 m, 2.1-5 m, 5.1-10 m, and >10 m), and percent cover of each height class as well as shrub and grass species richness and percent cover were measured using a 25-m line intercept beginning at each point count station and heading in a random compass direction (Canfield 1941). Differences between years for each vegetation variable were tested using ANOVA in R (R Core Team 2015), and no significant differences were found. Therefore, the two years of vegetation data for each variable at each site were averaged and these values were incorporated into the analyses described below.

Calculating bird species richness and community composition

Bird detections were truncated at 100 m to ensure similar sampling effort and plot sizes among sites and to ensure independence from adjacent sites (Magurran 2004). To account for species-specific detection probabilities we adjusted the count data for each species by dividing the number of detections at each site (C_i) by the median value of the posterior distributions of detection probabilities at each site (p_i) averaged across the four surveys. Posterior distributions of detection probabilities were estimated from the species-specific binomial-mixture models described below. For species that were too rare to estimate a detection probability we used a borrowing rule based on maximum detection distance before truncation (Alldredge et al. 2007) and vegetation strata in which the species most often displays based on field observations and Rodewald (2015). We first grouped all species into 4 general groups: 1) maximum detection distance of ≤ 50 m, 2) maximum detection distance of > 50 m and ≤ 100 m, 3) > 100 m and ≤ 150 m, and 4) > 150 m; and then grouped them into 3 secondary groups: 1) species that generally displays near the ground, 2) species that generally displays in the mid-level vegetation strata, and

3) species that generally displays on the tops of vegetation (Table A2.1). We then borrowed detection information from within groups. Using our adjusted count data, we calculated total species richness (pooled) for disturbed and reference sites using the non-parametric Chao gamma diversity estimator (Chao 1987), mean species richness per sampling site for both disturbed and reference sites, Shannon diversity index (Shannon 1948) and the J-evenness index (Pielou 1966) for disturbed and reference sites using the R packages *vegan* and *BiodiversityR* (Kindt and Coe 2005, Oksanen et al. 2015, R Core Team 2015). We then compared the mean species richness between disturbed and reference sites using ANOVA in R (R Core Team 2015). To compare community composition between disturbed and reference sites we calculated rank abundance distributions/curves using *BiodiversityR* (Kindt and Coe 2005, R Core Team 2015). Rank abundance distributions are commonly used to compare species diversity between assemblages (Magurran 2004). Rank abundance curves clearly display contrasting patterns of species richness and evenness between treatments by plotting the sequence from most to least abundant species along the horizontal axis (Magurran 2004).

Quantifying the effects of historical disturbance and vegetation characteristics on bird density

We used hierarchical open population binomial-mixture models (Kéry and Andrew Royle 2010, Kéry and Schaub 2012) to quantify, 1) the effect of historical disturbance on the abundance of bird species and 2) the effect of vegetation parameters on bird densities. Binomial-mixture models estimate abundance using repeated count data while taking into account imperfect detection (Kéry and Schaub 2012). Thus, they contain more information than simply estimating an occurrence/non-occurrence response, similar to the widely used occupancy modeling framework (MacKenzie et al. 2006, Kéry and Schaub 2012). Again, bird detections

were truncated at a 100 m radius from the point count station – making all surveyed areas 3.1 ha. Therefore, we infer our results as bird density (individuals/3.1 ha).

Modeling the effect of disturbance on bird density

To quantify the long-term effect of tree removal on species density, we let y_{ijk} be the total number of individuals counted at site i , during survey j , in year k (2013 and 2014).

Assuming the population was closed over the course of each year, the observed counts arise as a binomial random variable,

$$y_{ijk} \sim \text{binomial}(N_{ik}, p_{ijk}),$$

where N_{ik} is the total number of individuals available to be counted in year k at site i , and p_{ijk} is the survey specific detection probability. We then modeled our latent variable N_{ik} (Table A2.2) as a Poisson random variable,

$$N_{ik} \sim \text{Poisson}(\lambda_{ik}),$$

where λ_{ik} is the expected abundance at site i for year k . To quantify the influence of historical disturbance on the abundance of bird species, we modeled λ_{ik} as a function of disturbance or non-disturbance at site i using a log link,

$$\log(\lambda_{ik}) = \omega_{g[i]} + x_i\beta.$$

In this expression, our data vector (x_i) was set up so that reference sites were given a 0 and disturbed sites were given a 1. To account for potential spatial dependency we used a multilevel model to included a random effect ($\omega_{g[i]}$) on geographical grouping ($g = 7$). Each historically disturbed area (Fig. 2.1) was given its own group with the exception of the cluster of historical disturbances (Fig. 2.1b) in which they were placed together in a single group. We had a total of five disturbance groups. Reference sites were divided into two groups – the clustering of reference sites (Fig. 2.1b) was placed in one group and the geographically dispersed reference

sites were placed in a second group. We then modeled the group level parameters using a normal distribution and allowing each group to have a common mean (μ_ω) and standard deviation (σ_ω),

$$\omega_g \sim \text{normal}(\mu_\omega, \sigma_\omega^2).$$

This distribution has the effect of drawing the estimates of ω_g toward the mean level (μ_ω), but not entirely – thus, creating a partial-pooling compromise among the estimates (Gelman and Hill 2007). Based on previous analyses (T. Gallo unpublished data) we had reason to believe that observers conducting point count surveys had the greatest influence on the detection probability for all species. Therefore, we modeled the detection probability as a function of the observer conducting the survey at site i , survey j , and year k on the logit scale:

$$\text{logit}(p_{ijk}) = \alpha_o + \alpha_1 \text{observer}_{ijk}$$

Conventional ‘vague’ priors were used for all parameters. Specifically, we assumed

$\beta \sim \text{normal}(0,100)$, $\alpha \sim \text{normal}(0,100)$, $\mu \sim \text{normal}(0,100)$, and $\sigma \sim \text{uniform}(0,100)$.

Modeling the effect of vegetation parameters on bird densities

For those species showing a strong positive or negative response to historically disturbed sites (90% credible intervals not overlapping 0) we developed *a priori* hypotheses for which vegetation parameters may best explain variation in density based on foraging (De Graaf et al. 1985) and nesting guilds (Degraaf and Wentworth 1986) and habitat preference (Rodewald 2015) (Table 2.1, Table A2). To be cautious of over parameterizing our model we chose no more than 4 vegetation covariates per species. We then used the same hierarchical open population binomial-mixture model (Kéry and Andrew Royle 2010, Kéry and Schaub 2012) described

above to quantify the effect of habitat characteristics on species abundance. However, in this model X'_i represents a matrix of continuous predictor variables scaled to have mean 0 and standard deviation 1 (Gelman et al. 2008):

$$\log(\lambda_{ik}) = \omega_{g[i]} + X'_i\beta$$

We tested for correlations among covariates that appeared in the same model to ensure that no covariates were highly correlated ($|r| > 0.7$).

Table 2.1. *A priori* model formulation for each species based on vegetation parameters that were hypothesized to best explain variation in bird density. *Shrub*, *tree*, *grass*, and *bare ground* refer to proportion of shrub, tree, grass, and ground cover, respectively. *Shrub richness* refers to shrub species richness, *dbh* refers to the mean tree diameter/100 m², and *snag* refers to the number of standing dead trees.

Species	Model
Brewer's Sparrow	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_1tree_i + \beta_2shrub\ richness_i + \beta_3grass_i$
Cassin's Finch	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_2tree_i + \beta_3dbh_i + \beta_4ground_i$
Chipping Sparrow	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_1tree_i + \beta_2shrub\ richness_i + \beta_3grass_i$
Dark-eyed Junco	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_2grass_i + \beta_3ground_i + \beta_4tree_i$
Green-tailed Towhee	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_1tree_i + \beta_2shrub\ richness_i + \beta_3grass_i$
Juniper Titmouse	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_2tree_i + \beta_3dbh_i + \beta_4snag_i$
Mountain Bluebird	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_2tree_i + \beta_3dbh_i + \beta_4snag_i$
Mountain Chickadee	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_2tree_i + \beta_3dbh_i + \beta_4snag_i$
Morning Dove	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_1tree_i + \beta_2shrub\ richness_i + \beta_3grass_i$
Plumbeous Vireo	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_2tree_i + \beta_3dbh_i + \beta_4ground_i$
Rock Wren	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_2grass_i + \beta_3ground_i + \beta_4tree_i$
Spotted Towhee	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_2grass_i + \beta_3ground_i + \beta_4tree_i$
Vesper Sparrow	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_2grass_i + \beta_3ground_i + \beta_4tree_i$
Violet-green Swallow	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1shrub_i + \beta_2tree_i + \beta_3dbh_i + \beta_4snag_i$
White-breasted Nuthatch	$\log(\lambda_{ij}) = \omega_{g[i]} + \beta_1tree_i + \beta_2dbh_i + \beta_3log_i + \beta_4shrub_i$

Model fitting and estimation

Posterior distributions of model coefficients were estimated using Markov chain Monte Carlo (MCMC) methods implemented in JAGS using the *rjags* package in R (Plummer et al. 2006, R Core Team 2015). Models were run with 3 chains of 200,000 iterations each, a thinning

rate of 1, and a burn-in rate of 50,000. We checked for model convergence by visually inspecting the trace plots of MCMC samples, and by checking that the Gelman-Rubin diagnostic statistic for each parameter was <1.1 (Gelman and Rubin 1992). Due to both a small sample size and the desire to suggest reasonable management recommendations, we chose local-scale vegetation covariates that could be readily incorporated into land management practices. We also used relatively simple models to examine main effects alone, without considering additive effects or interactions. To assess the relative influence of disturbance and vegetation parameters on species density, we compared the posterior distribution of model coefficients and examined overlap of their distributions with 0.

RESULTS

Bird richness, diversity, and community composition

We observed a total of 37 bird species in historically disturbed sites and 56 bird species in reference sites (Table A2.3). Chao gamma diversity estimates were 38.32 (SE = 4.62) and 71 (SE = 12.59) for chained and reference sites respectively. Mean species richness per site differed significantly between chained (11 SD = 3.61) and reference sites (15.48 SD = 3.37; $F(1,48) = 20.61$, $P < 0.001$). Shannon-Wiener indices for disturbed and reference sites were 2.52 and 3.33, respectively, and the J-evenness indices were 0.72 for disturbed sites and 0.84 for reference sites.

The rank abundance distributions (Fig. 2.2) indicated that disturbed sites were dominated by fewer species and most of those species were shrubland-obligate species. Sixty five percent of the total detections in disturbed sites were represented by five species – Blue-gray Gnatcatcher (*Poliioptila caerulea*), Mountain Bluebird (*Sialia currucoides*), Brewer’s Sparrow, and Chipping Sparrow (*Spizella passerine*). In contrast, the relative proportion of bird species in reference sites was somewhat more even, and these sites were characterized by more woodland birds. The

five most dominant species in reference sites were Spotted Towhee, Black-throated Gray Warbler, *Empidonax* flycatcher (*Empidonax* sp.), Plumbeous Vireo (*Vireo plumbeus*), and Blue-gray gnatcatcher. These species combined represented 35% of the total detections in reference sites (Fig. 2.2).

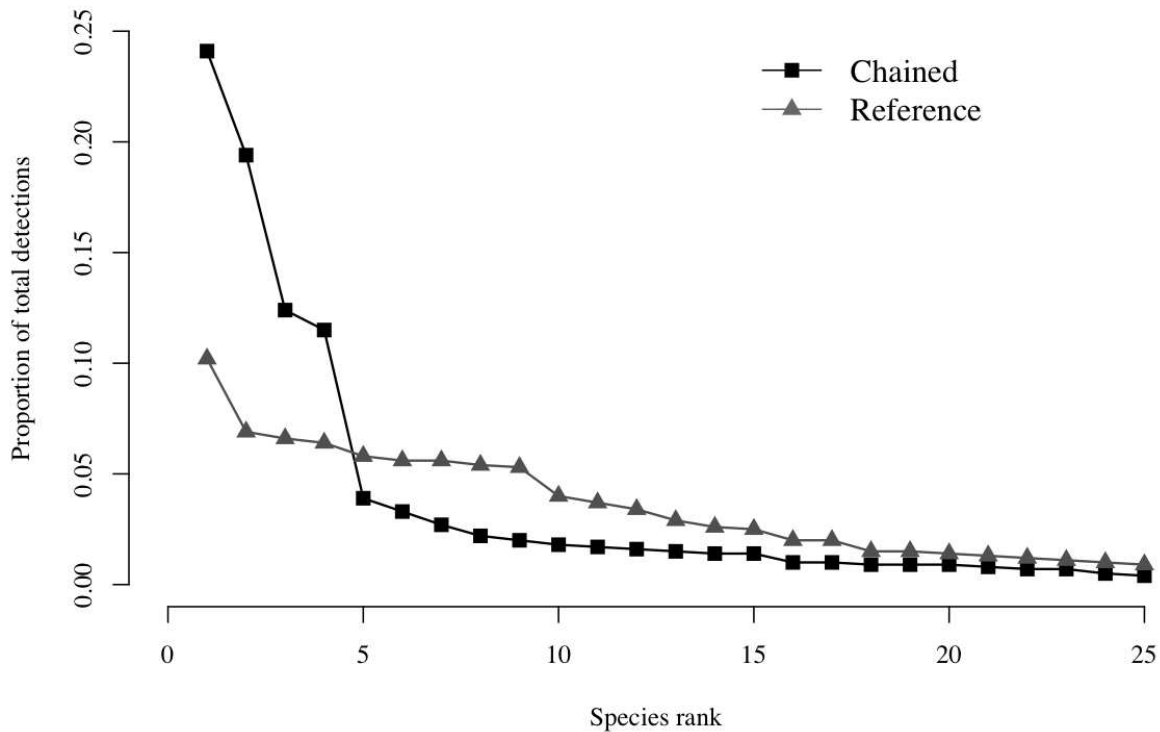


Fig. 2.2. Rank abundance curves for bird species in historically disturbed sites and undisturbed reference sites in the Piceance Basin, CO.

Bird densities in disturbed and reference sites

We were able to appropriately fit our hierarchical open population binomial mixture model to 21 species (Table 2.1). American Robin (*Turdus migratorius*), Cassin’s Finch (*Haemorhous cassinii*), Dark-eyed Junco (*Junco hyemalis*), Juniper Titmouse (*Baeolophus ridgwayi*), Mountain Chickadee, Mourning Dove (*Zenaida macroura*), Plumbeous Vireo,

Spotted Towhee, Vesper Sparrow (*Pooecetes gramineus*), Violet-green Swallow (*Tachycineta thalassina*), and White-breasted Nuthatch all demonstrated a strong long-term negative response to historical habitat disturbance. In contrast, Brewer’s Sparrow, Chipping Sparrow, Green-tailed Towhee, Mountain Bluebird (*Sialia currucoides*), and Rock Wren (*Salpinctes obsoletus*) showed strong positive responses to disturbance. Ash-throated flycatcher (*Myiarchus cinerascens*), Black-throated Gray Warbler, Broad-tailed Hummingbird (*Selasphorus platycercus*), *Empidonax* flycatcher, and Northern Flicker (*Colaptes auratus*) showed little to no response to historical disturbance (Table 2.2).

Table 2.2. Median posterior distributions values and proportion of the posterior distribution that lies below and above 0 for the open population binomial-mixture model used to assess the long-term effects of habitat disturbance on the density of bird species in the Piceance Basin in northwest Colorado, USA. Species are organized by the direction that habitat disturbance affected density.

Direction of effect	Species	Treatment coeff.	Prop.< 0	Prop.> 0
<i>Negative</i>	American Robin	-1.16	0.92	0.08
	Cassin's Finch	-1.55	1	0
	Dark-eyed Junco	-2.62	0.92	0.08
	Juniper Titmouse	-1.28	0.93	0.07
	Mountain Chickadee	-0.69	0.94	0.06
	Mourning Dove	-1.66	1	0
	Plumbeous Vireo	-3.11	1	0
	Spotted Towhee	-0.72	0.98	0.02
	Vesper Sparrow	-3.2	0.95	0.05
	Violet-green Swallow	-1.32	0.99	0.01
	White-breasted Nuthatch	-1.13	1	0
<i>Positive</i>	Brewer's Sparrow	2.5	0	1
	Chipping Sparrow	0.79	0.03	0.97
	Green-tailed Towhee	0.64	0.01	0.99
	Mountain Bluebird	1.55	0	1
	Rock Wren	1.31	0.02	0.98
<i>No effect</i>	Ash-throated Flycatcher	0.46	0.19	0.81
	Black-throated Gray Warbler	-0.43	0.81	0.19
	Broad-tailed Hummingbird	-0.47	0.74	0.26
	<i>Empidonax</i> flycatcher	0.39	0.13	0.87
	Northern Flicker	-0.76	0.82	0.18

Vegetation parameters influencing bird densities

Most measured habitat characteristics differed between disturbed and reference sites (Table 2.3). Disturbed sites had a higher mean percent cover of shrubs and grasses and a greater number of logs present. Reference sites had a higher mean percent cover of trees and a greater overall number of snags. However, proportion of bare ground and shrub species richness was similar across site types, and neither disturbed nor reference sites had trees over 10 m tall (Table 2.3).

Table 2.3. Summary statistics of vegetative parameters (means and 95% confidence intervals) in historically disturbed sites and reference sites in northwestern Colorado.

Parameter	Disturbed	95% Confidence Intervals		Reference	95% Confidence Intervals	
		Lower	Upper		Lower	Upper
Mean shrub cover	0.27	0.21	0.33	0.15	0.09	0.21
Mean grass cover	0.11	0.09	0.13	0.09	0.05	0.13
Mean proportion of bare ground	0.36	0.30	0.42	0.34	0.20	0.48
Mean forest cover	0.03	0.01	0.05	0.28	0.16	0.40
Mean cover of height class 0-0.5 m	0.13	0.11	0.15	0.09	0.05	0.13
Mean cover of height class 0.6-1 m	0.1	0.08	0.12	0.03	0.01	0.05
Mean cover of height class 1.1-2 m	0.1	0.08	0.12	0.05	0.03	0.07
Mean cover of height class 2.1-5 m	0.04	0.02	0.06	0.18	0.10	0.26
Mean cover of height class 5.1-10 m	0	0.00	0.00	0.1	0.06	0.14
Mean cover of height class >10 m	0	0.00	0.00	0	0.00	0.00
Mean DBH/plot (cm)	0.07	0.03	0.11	0.79	0.48	1.10
Average tree height (m)	1.02	0.55	1.49	3.12	1.90	4.34
Average snag height (m)	0	0.00	0.00	1.12	0.69	1.55
Mean number of logs	3.24	1.93	4.55	2.2	1.34	3.06
Mean number snags above 1.37 m	0	0.00	0.00	0.36	0.22	0.50
Mean number of shrub species	2.8	2.25	3.35	2.28	1.38	3.18

The influence and direction of effect of habitat characteristics on bird densities varied by species (Fig. 2.3). We found that proportion of shrub cover had a positive influence on Spotted Towhee, Green-tailed Towhee, and Mountain Chickadees; whereas, Rock Wren, Chipping Sparrow, Mourning Dove, Violet-green Swallow, Cassin's Finch, and Plumbeous Vireo showed a negative response to proportion of shrub cover. Increased tree cover had a positive influence on Spotted Towhee, White-breasted Nuthatch, Juniper Titmouse, Dark-eyed Junco, and Plumbeous Vireo; yet Brewer's Sparrow and Mountain Bluebird had a negative response to increased tree cover. Mountain Bluebird and Vesper Sparrow had a positive response to bare ground. Dark-eyed Junco, Green-tailed Towhee, Chipping Sparrow, and Mourning Dove all showed a positive response to proportion of grass cover; but Rock Wren showed a negative response to proportion of grass cover. Mountain Bluebird had a negative response to large snags (snags over 1.37 m), and Plumbeous Vireo had a negative response to increased bare ground. We were unable to fit our vegetation model to American Robin due to lack of model convergence.

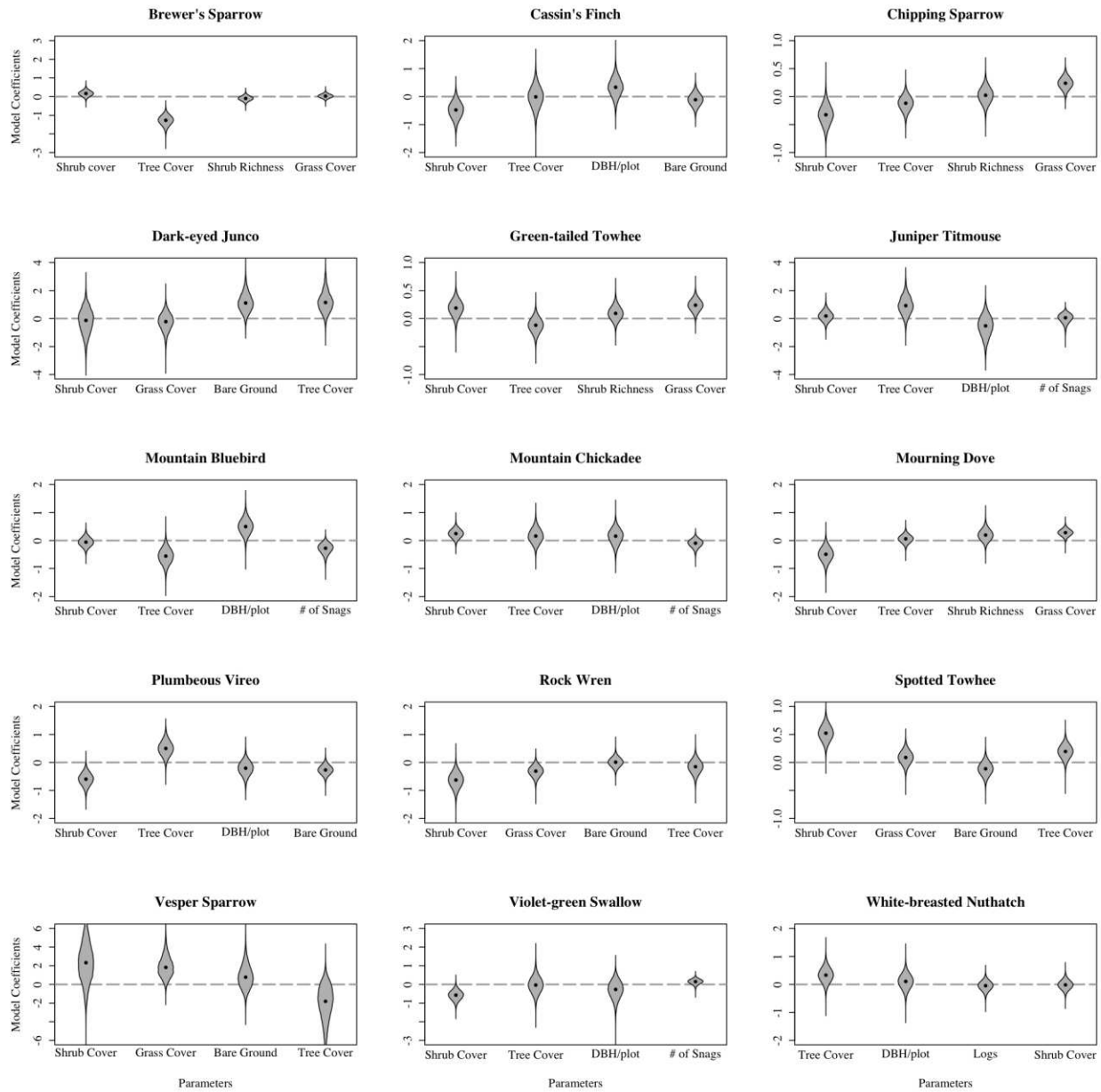


Fig. 2.3. Posterior distributions of vegetation covariates for bird species that experienced a strong long-term response to tree removal in the Piceance Basin, CO. Dashed line indicates coefficient value of 0. Dot in the center of each distribution represents the median posterior value.

DISCUSSION

Anthropogenic habitat disturbance is a major threat to natural communities, yet the consequences of these practices are often best understood only over short time periods (Gill 2007). We examined the long-term effects of large-scale tree removal on bird communities in the Piceance Basin of northwest Colorado. Our findings demonstrate marked differences in bird community composition and species densities between historically disturbed sites and reference woodlands. Our results, combined with those of previous studies (O'Meara et al. 1981, Sedgwick and Ryder 1986) indicate a persistent state change in these human-disturbed landscapes from a woodland bird community to one dominated by a shrubland bird community. Without any evidence that the bird community is returning to its original condition four decades after disturbance, it is possible that these natural communities may have been fundamentally altered (Redmond et al. 2013).

We found higher species diversity and species richness in reference woodlands, and these results are consistent with studies conducted in the first few years after tree removal at the same sites (O'Meara et al. 1981, Sedgwick and Ryder 1986). O'Meara et al. (1981) also found that recently disturbed areas were dominated by shrubland-obligate species (e.g., Brewer's Sparrows and Green-tailed Towhees), and they showed that woodland-obligate species, such as Mountain Chickadees, White-breasted Nuthatches and Black-throated Gray Warblers, were most common in reference sites. Our rank-abundance distributions demonstrate similar differences in community composition forty years after disturbance. Specifically, historically disturbed sites were dominated by a smaller number of species and these dominant species were shrubland-obligate birds, while woodland-obligate birds dominated reference woodlands.

Species were differentially affected by disturbance based on habitat preference. Similar to studies conducted shortly after the initial disturbance (O'Meara et al. 1981, Sedgwick and Ryder 1986), we found tree removal had negative effects on the densities of White-breasted Nuthatch, Mountain Chickadee, Juniper Titmouse, and Violet-green Swallow. In 1976, after the initial disturbance and the four subsequent years following, Mourning Dove, *Empidonax* flycatcher, Mountain Chickadee, Plumbeous Vireo, and Black-throated Gray Warbler had higher densities in the reference woodlands (Sedgwick and Ryder 1986), whereas, Green-tailed Towhee and Brewer's Sparrow had higher densities in disturbed sites. We found similar results >40 years later, with the exception of *Empidonax* flycatcher and Black-throated Gray Warbler. *Empidonax* flycatchers are air sallying birds and require plant structural diversity for pursuing insects from perches (Sabo and Holmes 1983). Although taller trees and snags were initially removed during tree removal activities, plant structural diversity may have sufficiently recovered to a state beneficial to air sallying birds. Black-throated gray warblers are woodland-obligate birds and choose nest sites within the canopy of trees (Guzy and Lowther 2012). However, in P-J systems, Black-throated Gray Warblers have been show to forage on big sagebrush up to 23% of the time (Guzy and Lowther 2012). Big sagebrush was the dominate shrub species in historically disturbed sites, therefore, it is unsurprising that Black-throated Gray Warblers are utilizing these sites for foraging.

Higher densities of some cavity nesters (e.g., Juniper Titmouse), tree nesters (e.g., Plumbeous Vireo) and bark gleaners (e.g., White-breasted Nuthatch) (Table A2, Fig. 2.3) were positively associated with tree cover and trees with larger DBH. These results are consistent with previous studies demonstrating that tree cover and larger DBH trees are important habitat variables for these nesting and foraging guilds (Airola and Barrett 1985, Li and Martin 1991).

Tree cover in disturbed sites was extremely low (mean percent cover of vegetation 5.1-10 m and >10 m were both 0%, and mean percent cover of vegetation 2.1-5 m was 4%; Table 2.3), and the trees that had regenerated since disturbance were relatively small compared to reference sites (average tree height = 1.02 m and mean DBH/plot = 0.07 cm; Table 2.3). Because pinyon and juniper are both slow growing tree species (Tausch and Tueller 1977), it may take decades or even centuries for these trees to recover to a state preferred by cavity nesters, tree nesters, bark gleaners, and canopy gleaners.

Shrubland-obligate birds such as Green-tailed Towhee and Brewer's Sparrow all responded positively to disturbance, but differently to individual vegetative characteristics. Higher densities of Green-tailed Towhee were associated with increased shrub and grass cover, which is unsurprising as Green-tailed Towhees tend to nest and forage amid dense shrubs and scattered grasses (Dobbs et al. 2012). Brewer's Sparrow densities had positive response to increased shrub cover, but densities declined with increased shrub species richness. Brewer's Sparrows are closely associated with large open habitats dominated by big sagebrush (Wiens and Rotenberry 1981). It is possible that the spatial extent of big sagebrush in historically disturbed areas have a stronger influence on Brewer's Sparrow's habitat selection than local shrub species richness or composition (Petersen and Best 1987).

The vegetation parameters that influenced densities of ground nesting birds were complex. Proportion of bare ground had a positive influence on Dark-eyed Junco densities, yet had no effect on Rock Wren or Vesper Sparrow. Further, Dark-eyed Junco and Vesper Sparrow showed a negative long-term response to tree removal, yet Rock Wren showed a positive long-term response to tree removal. In general, older P-J woodlands have scarce understory vegetation (Tausch and Tueller 1977), which tends to increase the proportion of bare ground. In our study

area, however, both references and chained sites had similar proportions of bare ground. Areas where P-J woodlands have been reduced or removed tend to have more perennial and annual grass and forb cover (Aro 1971), which could be used as cover from predation and nesting material for ground nesting birds (Martin and Roper 1988). However, Rock Wren responded negatively, Vesper Sparrow responded positively, and Dark-Eyed Junco showed no response to increased grass cover. It is possible that measuring parameters at smaller, species-specific scales (e.g., nest site selection) could further explain patterns in ground-nesting bird densities.

Consistent with our findings, long-term state-changes in vegetative communities after P-J removal have been observed in the Colorado Plateau (Redmond et al. 2013). However, succession in P-J ecosystems after disturbance can sometimes take decades (Tausch and Tueller 1977, Schott and Pieper 1987). Although Tausch and Tueller (1977) and Schott and Pieper (1987) are careful to state that not all P-J woodlands follow the same model of succession, they do emphasize that tree regrowth can take up to 50-60 years after mechanical disturbances. Thus, recovery from a mechanical disturbance in a slow recovering system may not be evident within 40-50 years. However, historically disturbed sites in our study area averaged only 4% cover of P-J over 2 m tall, contained no trees over 5 m tall, and were dominated by shrub species, indicating that recovery to a woodland system even after 40 years may have only just begun, or alternatively, this community may continue to persist in an alternate shrubland-dominated state. Our study reinforces the value of multi-decadal ecological monitoring (Lindenmayer and Likens 2009), and suggests careful consideration should be given to the long-term consequences of deliberate habitat disturbance for natural communities.

Our response metrics were limited to bird community composition, richness and density within a disturbed patch. Future research should evaluate the long-term effects of disturbance on the fitness of avian populations on the landscape level to more fully understand bird population dynamics in a slow recovering system. Further, to our knowledge, there are no studies that have investigated the effect of deliberate and large-scale tree removal on bird communities in a chronological time series from initial disturbance decades ago until present day. Chaining as a means of tree reduction was used less frequently after the 1970's in P-J ecosystems (Romme et al. 2009, Redmond et al. 2013), making it difficult to measure changes in bird and plant communities using a space for time substitution for long-term data. Finally, our findings from northwest Colorado may or may not be relevant to tree reduction or other forms of deliberate habitat disturbance in other regions or ecosystems. The broad geographical distribution of P-J woodlands supports a wide variety of habitat types and bird communities (Balda and Masters 1980). These systems also can vary by soil type, precipitation, and elevation, which could affect rates and patterns of regeneration after disturbance (Schott and Pieper 1987, Romme et al. 2009, Tausch et al. 2009). Therefore, we urge caution in generalizing our results to other woodland ecosystems, and encourage monitoring the effects of disturbance on diverse taxa wherever tree removal is employed as a form of habitat improvement or mitigation.

The relationships we identified between bird densities and specific habitat characteristics could be factored into management decisions to account for the long-term effects of tree removal on avian communities. Modern tree removal techniques (e.g., hydroax; Wästerlund and Hassan 1995) can be highly selective in the trees that are removed and the vegetation cover that is left in place (Wästerlund and Hassan 1995, Crow and van Riper 2010). We found mean DBH/plot (a proxy for forest age) was the most significant vegetative parameter influencing habitat use of

bark gleaning birds. Therefore, leaving larger older trees within the treated areas would maintain habitat for bark gleaning birds. Tree cover had a significant positive influence on habitat use by upper canopy gleaners and tree nesters; therefore, selectively leaving patches of intact tree cover could mitigate the negative effects of tree removal on canopy gleaners and tree nesting birds. We emphasize, however, that tree reduction projects have diverse goals and intended outcomes. Some objectives (e.g. creating habitat for Greater Sage Grouse, which cannot tolerate standing trees; Baruch-Mordo et al. 2013) may not be compatible with our recommendations for maintaining habitat quality for other groups of birds.

Anthropogenic disturbance in the form of tree reduction to increase forage for livestock and enhance habitat for mule deer had long-term ecological consequences for bird communities in our study region. Thus, we suggest that future management actions that result in large-scale tree removal should explicitly measure intended and unintended effects on birds and other taxonomic groups. The hierarchical binomial-mixture model that we used can be applied more broadly to predict effects on plants and animals based on individual habitat traits. This information could be used to adaptively guide management decisions, taking into account long-term costs and benefits to species of both conservation and/or economic value. Given the magnitude of anthropogenic impacts on natural ecosystems, which increasingly include actions intended to enhance habitat quality for particular species, it is imperative that we understand the long-term effects of deliberate habitat disturbance on diverse communities.

CHAPTER 3

PINYON-JUNIPER REMOVAL HAS LONG-TERM EFFECTS ON MAMMALS

INTRODUCTION

Removing or reducing tree cover to benefit cattle or game species has been a common forest management practice for decades (Aro 1971, Yahner 1984, Redmond et al. 2013, Bergman et al. 2015). The scale and intensity of these tree removal practices is expected to increase as technology advances and land managers are tasked with meeting multiple objectives, including fire prevention and enhancing forage for livestock and shrub-dependent wildlife in areas undergoing urbanization and energy development (Connelly et al. 2000, Redmond et al. 2013, Bergman et al. 2014). Although this practice is widespread and pervasive in some forest ecosystems, little is known about the long-term effects of tree removal on wildlife communities (Bombaci and Pejchar 2016).

During the last half-century, pinyon-juniper (P-J) woodlands have been a major focus of both forest conservation and tree reduction efforts (Redmond et al. 2013). Collectively, P-J woodlands are the third largest vegetative community in the United States, covering over 40 million hectares (Romme et al. 2009). P-J woodlands are also an important source of food and cover for woodland-dependent wildlife species and forage for livestock (Schott and Pieper 1987, Romme et al. 2009). However, both pinyon and juniper trees have been expanding into grasslands and shrublands for the past 150 years (Romme et al. 2009). Pinyon-juniper expansion into grasslands and shrublands reduces forage for livestock, increases fuel for wildfires and reduces habitat for economically important or rare species that depend on open habitats, such as

mule deer (*Odocoileus hemionus*) and Greater Sage Grouse (*Centrocercus urophasianus*) (Terrel and Spillett 1975, Evans 1988). Tree removal and reduction has thus become a common tool for forest and wildlife management in P-J ecosystems (Aro 1971, Bergman et al. 2014).

Historically, a mechanical technique known as chaining was the most common method for removing tree cover (Aro 1971, Miller and Wigand 1994, Redmond et al. 2013). Between 1950 – 1964, 1.2 million hectares of P-J were removed via chaining in the U.S. (Box et al. 1966). Chaining uproots trees and shrubs by dragging heavy anchor chains between two bulldozers across the forested landscape (Aro 1971, Sedgwick and Ryder 1986, BLM 2008). Chaining is an efficient method for killing older, larger trees over a large area, and increasing herbaceous forage for livestock and economically important wildlife (Aro 1971). Because chaining removes all or most trees, P-J woodlands are replaced with open grasslands and shrubland habitats (Tausch and Tueller 1977). Over time, the removal of P-J cover can shift the plant community from a woodland dominated system to a shrub dominated system (Redmond et al. 2013). Specifically, P-J removal can lead to more perennial grasses, reduced tree cover, and increased shrub cover (Yorks et al. 1994, Romme et al. 2009, Redmond et al. 2013). Although the use of chaining has tapered off since the 1970's (Romme et al. 2009, Redmond et al. 2013), it has been replaced with other mechanical tree removal methods (e.g., hydro ax) that have similar objectives and outcomes (Wästerlund and Hassan 1995).

Regardless of the method employed, tree removal and subsequent changes to the plant community may fundamentally alter habitat use by mammals (Andr et al., 1994), particularly in slow recovering systems, such as P-J woodlands (Schott and Pieper 1987). Mammals contribute to important ecological processes such as seed dispersal and nutrient cycling, and small and

medium-sized mammals are food sources for predatory birds and large carnivores (Holechek 1981). Thus, understanding the long-term effects of the widespread practice of P-J removal on mammalian species is both ecologically interesting and has important conservation implications.

The short- and long-term effects of tree removal on medium and large-size mammals in temperate and arid forest systems remain poorly understood (Crooks 2002), and most studies of small mammals evaluated effects only shortly after tree removal. Habitat use of generalist rodent species (e.g., deer mouse, *Peromyscus maniculatus*) increased immediately following chaining (Baker and Frischknecht 1973, O'Meara et al. 1981, Sedgwick and Ryder 1986). Specifically, O'Meara et al. (1981) and Sedgwick and Ryder (1986) demonstrated that a greater number of least chipmunks (*Tamias minimus*) were caught in chained sites than in undisturbed reference sites, but found a decrease in the total abundance of a pinyon-juniper specialist – pinyon mouse (*Peromyscus truei*). Sedgwick and Ryder (1986) also found that golden-mantled ground squirrels (*Callospermophilus lateralis*) were caught less often in chained sites than in reference sites. In New Mexico, lagomorphs preferred chained sites to reference sites immediately following disturbance (Kundaeli and Reynolds 1972, Howard et al. 1987). With the exception of Howard et al. (1987), these studies occurred in the few years following tree removal. Inference about changes in species composition immediately following habitat disturbance (e.g., 1-3 years) may or may not predict long-term effects on community dynamics (Stouffer et al. 2011). Although long-term effects on species can go undetected in short-term studies (Laurance et al. 2002; Laurance et al. 2011), these baseline studies provide the unique opportunity to revisit historic mechanically disturbed sites and compare short and long-term changes to mammal habitat use.

We investigated whether tree removal intended to increase forage for livestock and mule deer has long-term consequences for other mammal species. We hypothesize that habitat use will differ between historically disturbed sites and reference sites depending on the species. Using remotely-triggered wildlife cameras, we compared mammalian habitat use in P-J woodlands chained greater than 40 years ago to habitat use in intact reference woodlands in the Piceance Basin of northwest Colorado, USA. Our research objectives were both to evaluate differences in habitat use between historically chained and reference sites, and to identify the vegetation characteristics that influenced habitat use in our study area. Where mammalian habitat use is associated with particular vegetative characteristics, we suggest incorporating these characteristics into future forest management practices, with the objective of minimizing impacts on species of conservation concern.

MATERIALS & METHODS

Study site

We examined mammal habitat use in historically chained and reference P-J woodlands on public land managed by the Bureau of Land Management in the Piceance Basin of northwestern Colorado, U.S.A. Dominant land uses in this P-J ecosystem include oil and gas extraction and domestic livestock grazing (Northrup et al. 2015). The area is topographically diverse and consists of high plateaus and deeply incised valleys. Our sampling sites ranged from approximately 1500 – 2400 m in elevation. Woodlands are dominated by pinyon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*) (Sedgwick 1987). The dominant shrubs are antelope bitterbrush (*Purshia tridentata*), mountain mahogany (*Cercocarpus montanus*), big

sagebrush (*Artemisia tridentata*), rabbitbrush (*Chrysothamnus* spp.), common chokecherry (*Prunus virginiana*), and Saskatoon serviceberry (*Amelanchier alnifolia*) (O'Meara et al. 1981; Sedgwick 1987).

Site selection

To assess habitat use of mammals in chained and reference woodlands, we established sampling sites in areas that historically underwent tree removal (“chained sites”) and P-J woodlands that had not been mechanically disturbed (“reference sites”). We identified areas that had been chained in the 1950’s – 1970’s using local knowledge from land managers and aerial imagery. We visited each historically chained location to confirm that no additional disturbance (i.e., prescribed fire, wildfire, or mechanical tree removal) had occurred since the initial disturbance. This was evidenced by the presence of many large, decaying, fallen trees and the absence of charred debris. Nine chained areas ranging in size from 3 – 1243 ha were identified. Using Geographic Information Systems (GIS), we established 22 sampling sites throughout these areas. We placed our first sampling site within each chained area by picking a random location in the approximate center of an accessible portion of the area. We then placed additional sampling sites in each cardinal direction such that all sites were at least 250 m apart. Because we began allocating sites in the smallest chained areas first, our design allowed for one site in each of the smallest areas and up to 6 sampling sites in the largest areas (Fig. 3.1).

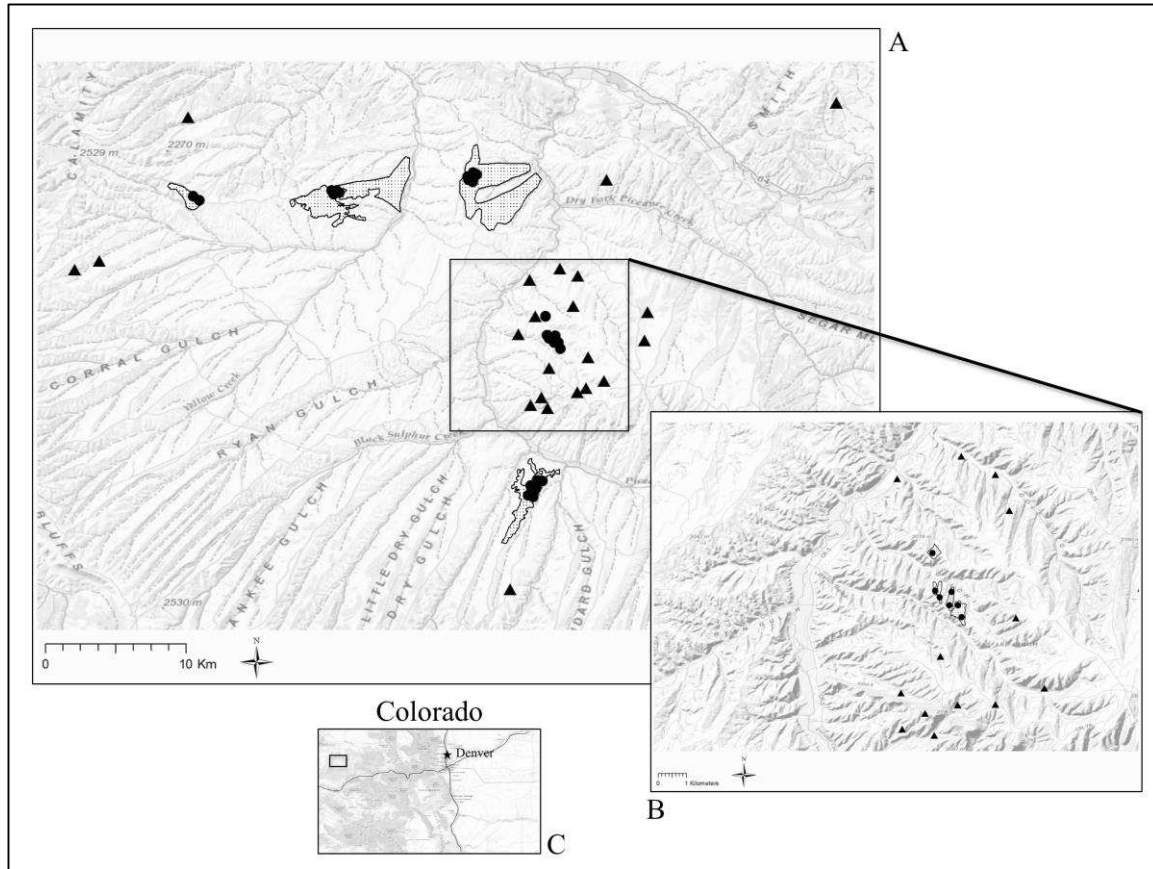


Fig. 3.1. Location of historic treatment and reference study sites in the Piceance Basin in northwestern Colorado, U.S. Stippled polygons represent historically chained areas. Black circles mark chained sites and black triangles mark reference sites. Map (A) shows the full extent of study area, inset (B) illustrates the distribution of chained and reference sites in and around the cluster of small chained areas, and inset (C) shows the area of the study site within the state of Colorado, USA (Basemap Source: ESRI, Redlands, CA, USA).

Reference sampling sites ($n = 50$) had been previously established across the study area for an ongoing wildlife-monitoring program. All reference sites were randomly placed on the landscape using GIS and were buffered from disturbed areas (e.g., other areas of tree removal or prescribed fire sites) by at least 250 m. Each reference site was ground-truthed to verify that it was within P-J woodlands. We randomly selected 22 reference sites for this study from the larger pre-existing set of reference sites using GIS to maintain a consistent sampling effort between the chained and reference sites (Magurran 2004). Because five of the nine historically chained areas

were spatially aggregated on the landscape, we stratified reference sites such that 16 sites were randomly selected from locations near the large cluster of chained areas (Fig. 3.1B). The remaining 6 sites were geographically dispersed across the study area (Fig. 3.1A). This design ensured that all reference sites were within 15 km of the historically chained areas (Fig. 3.1).

Wildlife camera trapping

To quantify mammal habitat use, we placed one unbaited remotely-triggered camera at each sampling site. To maximize detection probability, we established all cameras on wildlife trails, livestock trails, or unmaintained roads (e.g., two-tracks) situated within a 100 m radius of each sampling point. We used three models of cameras: Reconyx PC800 ($n = 30$; Reconyx, Holmen, WI, USA), Cuddeback Attack ($n = 10$; Cuddeback Digital, De Pere, WI, USA), and Cuddeback Capture ($n = 4$; Cuddeback Digital, De Pere, WI, USA). We allocated sampling effort evenly within each site type (chained or reference) among the three camera models. We deployed cameras from April – June 2014, and each camera collected relative-activity data for each species (photos per day) for 149 days following deployment. We checked each camera approximately every two weeks to change batteries, replace data cards, and to download photographs.

Volunteer observers were trained to identify species within each photo. Two observers viewed each photo, and all animals captured in the photographs were identified to species. To ensure consistency across species identifications, two of the authors (TG and LS) resolved all conflicting identifications between volunteers. Any animal that could not be identified to species was excluded from the analysis, with the exception of two chipmunk species. Least chipmunk and Uinta chipmunk (*Tamias umbrinus*) are difficult to consistently and accurately differentiate from wildlife camera photographs; therefore they were collectively recorded as “chipmunk”. The

Colorado Parks and Wildlife Photo Warehouse was used to store and manage data and to generate species occurrence results (Ivan and Newkirk 2015). Camera trap data were standardized to include a 30-second quiet period between subsequent photos to ensure a similar trigger rate among camera types.

Vegetation surveys

To assess the relationship between habitat use and vegetation characteristics, we measured a variety of vegetative parameters (Table 3.1). We sampled vegetation parameters in a 10x10-m plot offset by ~5 m from each sampling point in a random compass direction (McElhinny et al. 2005). We recorded crown area and diameter at breast height (DBH) for each tree following the methods used in Huffman et al. (2009). Trees (live and dead) were defined as >1 m in height (Huffman et al. 2012). We calculated percent tree cover by dividing total crown area by plot size (100 m²), and estimated an index of forest stand age by dividing the total DBH of all trees within each plot by plot size (DBH/plot; modified from McElhinny et al. 2005). Percent cover of shrubs was measured using a 25-m line intercept beginning at each sampling point and heading in a random compass direction (Canfield 1941). At each site a 1-m² quadrat was placed at a random location on each side of the 25-m line. Percent cover of grass was calculated by averaging the proportion of grass cover between the two 1-m² quadrats for each site (Elzinga et al. 1998).

Table 3.1. Summary statistics of vegetative parameters (means and 95% confidence intervals) in historically chained sites and reference sites in northwestern Colorado.

Parameter	Chained	95% Confidence Intervals		Reference	95% Confidence Intervals	
		Lower	Upper		Lower	Upper
Mean shrub cover	0.27	0.21	0.33	0.15	0.09	0.21
Mean grass cover	0.11	0.09	0.13	0.09	0.05	0.13
Mean tree cover	0.03	0.01	0.05	0.28	0.16	0.4
Mean DBH/plot (cm)	0.07	0.03	0.11	0.79	0.48	1.1

Statistical analysis

We used the following Bayesian hierarchical binomial-mixture models, parameterized by Royle and Dorazio (2008) and Gelman and Hill (2007), to address our two specific research objectives: 1) compare mammal habitat use between historically chained sites and undisturbed references sites, and 2) determine the importance of grass, shrub, and tree cover, and forest age (DHB/plot) for explaining potential differences in mammal habitat use among chained and reference sites. Binomial-mixture models are traditionally used to estimate the number of individuals in a population using repeated count data and taking into account imperfect detection (Kéry and Schaub 2012). Alternatively, we exchanged the number of individuals of a given species with the relative activity (detections per day) of a given species. Thus, allowing us to estimate a more appropriate calculation of habitat use than estimating an occurrence/non-occurrence response, similar to the widely used occupancy modeling framework used for camera trap data of unmarked individuals (MacKenzie et al. 2006, O'Connell and Bailey 2011, Kéry and Schaub 2012).

Treatment model formulation. To quantify the long-term effect of chaining on mammal habitat use, we let y_{ij} be the total number of photos counted each day j , at site i . These observed counts would arise as a binomial random variable,

$$y_{ij} \sim \text{binomial}(A_i, p_i),$$

where A_i is the total number of species-specific detections counted at site i , and p_i is the site specific detection probability. We then modeled the latent variable A_i as a Poisson random variable,

$$A_i \sim \text{Poisson}(\lambda_i),$$

where λ_i is the expected number of detections at site i . To quantify the effect of historical disturbance on mammal habitat use, we modeled λ_i as a function of disturbance or non-disturbance at each site using a log link,

$$\log(\lambda_i) = \omega_{g[i]} + \beta_1 \text{chain}_i.$$

Our data vector was set up so that β_1 was the model coefficient for the chained sites. Our sampling sites varied in size and were clustered geographically within our study area. Therefore, to account for potential spatial dependency within spatial grouping we used a multilevel model parameterized by Gelman and Hill (2007) to include a random effect $\omega_{g[i]}$ on geographical grouping. Each chained area (Fig. 3.1) was given its own group with the exception of the spatially aggregated chainings (Fig. 3.1B) in which they were placed together in a single group. We had a total of five chaining groups. Reference sites were divided into two groups – the clustering of reference sites (Fig. 3.1B) was placed in one group and the geographically

dispersed reference sites (Fig. 3.1A) were placed in a second group. We then modeled the group level parameters using a normal distribution and allowing each group to have a common mean (μ_ω) and standard deviation (σ_ω),

$$\omega_g \sim \text{normal}(\mu_\omega, \sigma_\omega^2).$$

This distribution allows the estimates of ω_g to draw toward the mean (μ_ω), but not completely – creating a partial-pooling compromise among the estimates (Gelman and Hill 2007). Detection radius varied by camera type, so we modeled the detection probability (p) as a function of camera type (*camera*) at each site on the logit scale:

$$\text{logit}(p_i) = \alpha_0 + \alpha_1 \text{camera}_i$$

Vague priors were used for all parameters. Specifically, we assumed $\beta \sim \text{normal}(0,100)$, $\alpha \sim \text{normal}(0,100)$, $\mu \sim \text{normal}(0,100)$, and $\sigma \sim \text{uniform}(0,100)$.

Covariate model formulation. To address our second objective of evaluating the effect of percent cover of grass, shrub, and tree and forest age on habitat use for each species, we used a similar Bayesian hierarchical binomial-mixture model as previously described. However, in this model our independent variables of percent cover of grass (*grass*), shrubs (*shrub*), and trees (*tree*) and forest age (*dbh*) are continuous variables scaled to have mean 0 and standard deviation 1 (Gelman et al. 2008):

$$\log(\lambda_i) = \omega_{g[i]} + \beta_1 \text{grass}_i + \beta_2 \text{shrub}_i + \beta_3 \text{tree}_i + \beta_4 \text{dbh}_i$$

We tested for correlations among covariates to ensure that no covariates were highly correlated ($|r| > 0.7$). Again, we assumed $\beta \sim \text{normal}(0,100)$, $\alpha \sim \text{normal}(0,100)$, $\mu \sim \text{normal}(0,100)$, and $\sigma \sim \text{uniform}(0,100)$.

Model fitting and estimations. Posterior distributions of model coefficients were estimated using a Markov chain Monte Carlo (MCMC) algorithm implemented in JAGS using the *rjags* package in R (Plummer et al. 2006, R Core Team 2015). JAGS models were run with 3 parallel chains for each model with a minimum of 100,000 iterations to a maximum of 300,000 iterations (depending on species) and the first 50,000-100,000 (depending on iterations) were discarded as burn-in. Model convergence was assessed by checking that the Gelman-Rubin diagnostic statistic for each parameter was <1.1 (Gelman and Rubin 1992) and visually inspecting the trace plots of MCMC samples. Due to both a small sample size and desire to suggest reasonable management recommendations, we chose to use relatively simple models to examine the main effect of each vegetation characteristic, without considering additive or interactive effects. To assess the relative influence of covariates on habitat use, we compared the posterior distributions of model coefficients, and examined overlap of posterior distributions with 0.

RESULTS

A total of 21 mammal species were photographed across both site types (Table A3). Eighteen species were photographed at chained sites and 19 species were photographed at references sites. We were able to appropriately fit the treatment model to seven species – bobcat (*Lynx rufus*), mountain lion (*Puma concolor*), coyote (*Canis latrans*), American black bear (*Ursus americanus*), mountain cottontail (*Sylvilagus nuttallii*), golden-mantled ground squirrel, rock squirrel (*Otospermophilus variegatus*), and chipmunk (*Tamias* sp.). We found that bobcat, mountain lion, American black bear, golden-mantled ground squirrel, and rock squirrel had a

strong negative response to chaining more than 40 years after tree removal (Fig. 3.2). Chipmunk, mountain cottontail, and coyote showed little to no response to chaining; and we found no evidence of a strong positive response to chaining for any species (Fig. 3.2).

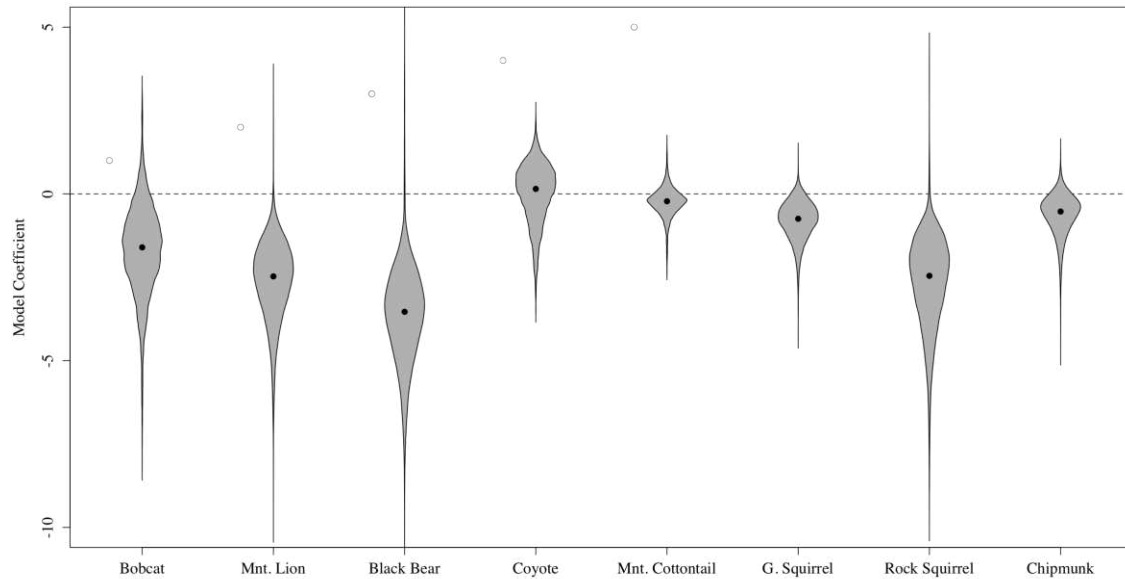


Fig. 3.2. Posterior distributions of model coefficient values for the effect of chaining on mammal habitat use in the Piceance Basin in northwestern CO, USA. Black dots represent median posterior coefficient values.

Most vegetation characteristics differed between chained and reference sites. Chained sites had a higher mean percent cover of shrubs and grasses, and reference sites had a higher mean percent cover of trees and larger mean DBH/100m² (Table 3.1). The influence of these vegetation characteristics on mammalian habitat use differed in both direction and strength, and varied by species (Table 3.2). Grass cover had a strong positive effect on mountain cottontail and golden-mantled ground squirrel habitat use, but had a strong negative effect on bobcat and mountain lion habitat use. Shrub cover had a strong positive effect on bobcat habitat use, but a strong negative effect on rock squirrel habitat use. Increased tree cover had a strong positive effect on habitat use by mountain cottontail and chipmunk, but a strong negative effect on habitat

use by coyote. Forest age had a strong positive effect on mountain lion habitat use, but a strong negative effect on bobcat and golden-mantled ground squirrel habitat use. We were unable to fit our vegetation model to American back bear due to a lack of model convergence.

Table 3.2. Median posterior coefficient values (Coeff.) and proportion (Prop.) of posterior distributions of vegetation characteristics that are below or above 0 for mammal species in the Piceance Basin, Northwest CO, USA. Covariates with posterior distributions largely below or above 0 had a strong effect on mammal habitat use. Posteriors with >90% of the distribution below or above 0 are indicated with bold italic.

	Grass Cover			Shrub Cover			Tree Cover			Forest Age		
	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0
Bobcat	-0.33	0.96	0.04	0.26	0.06	0.94	0.07	0.34	0.66	-0.72	1.00	0.00
Mountain Lion	-0.78	0.95	0.05	0.34	0.16	0.84	0.39	0.12	0.88	0.50	0.02	0.98
Coyote	0.05	0.33	0.67	-0.01	0.54	0.46	-0.67	1.00	0.00	-0.17	0.75	0.25
Mountain Cottontail	0.17	0.00	1.00	0.01	0.43	0.57	0.13	0.07	0.93	-0.02	0.60	0.40
Golden-mantled Ground Squirrel	0.53	0.00	1.00	0.03	0.42	0.58	0.11	0.32	0.68	-0.26	0.91	0.09
Rock Squirrel	0.12	0.39	0.61	-0.67	0.91	0.09	-0.19	0.60	0.40	0.45	0.17	0.83
Chipmunk sp.	-0.36	0.79	0.21	-0.39	0.79	0.21	0.73	0.04	0.96	-0.52	0.89	0.11

DISCUSSION

Removing or reducing tree cover to benefit livestock or game species is widespread throughout the western U.S. (Aro 1971, Redmond et al. 2013, Bergman et al. 2015), yet little is known about the long-term effects of tree removal on wildlife communities (Bombaci and Pejchar 2016). We examined the long-term effects of pinyon-juniper removal on mammalian habitat use in the Piceance Basin in northwestern Colorado. We found marked differences in habitat use between historically chained sites and reference woodlands for some species. Bobcat, mountain lion, American black bear, golden-mantled ground squirrel all used historically chained sites far less than reference sites more than 40 years after chaining.

In many cases vegetation characteristics helped explain variation in habitat use between chained sites and reference sites. We found that both mountain lions and bobcats were negatively influenced by an increase in grass cover. Bobcats preferred areas with high shrub cover, but were less likely to use areas that had mature P-J trees. Whereas, mountain lions were more likely to use areas with mature P-J trees. In previous studies, mountain lions and bobcats preferred a wide variety of habitat characteristics, but mountain lions have been shown to avoid large open areas in summer months (Koehler and Hornocker 1991, Dickson and Beier 2002). Further, mountain lions have shown to utilize mature woodlands during summer months for protective covering and areas to stalk prey (Koehler and Hornocker 1991). Mean DBH/plot was significantly smaller in chained sites, and these sites were dominated by big sagebrush and other low-statured vegetation (38% combined grass and shrub cover). Therefore, mountain lions may utilize historically chained sites less than woodlands because of their spatial extent and lack of protective cover. In Arizona, bobcats often used shrubland habitats but avoided large extensive grasslands, and they showed no strong preference for any single habitat type (Lawhead 1984).

Bobcats and mountain lions have large home ranges that encompass a wide assortment of habitat types (Schonewald-Cox et al. 1991). Therefore, it is likely that habitat characteristics at varying scales influence habitat use of both mountain lions and bobcats.

With respect to small mammals, we were only able to detect a small subsection of the small mammal community using wildlife cameras. However, the responses differed among those species we did observe. Chipmunk and mountain cottontails showed no long-term response to tree removal. However, rock squirrel and golden-mantled ground squirrel showed a negative response to P-J removal. These results, greater than 40 years after the initial chaining, are similar to those from studies conducted in the first few years after chaining occurred (Sedgwick and Ryder 1986). Rock squirrel showed a strong negative response to increased shrub cover; however, golden-mantled ground squirrel showed a strong negative response to mature P-J trees. Both species showed a strong negative response to chaining, yet they both favored vegetation characteristics defining chained sites (increased shrub cover and decreased DBH/plot, Table 3.2) – making it difficult to explain why golden-mantled ground squirrel and rock squirrels showed a strong negative response to P-J removal in our study area. Both species are known to utilize forested and open-brushy habitats if there are rock outcroppings for shelter (Oaks et al. 1987, Bartels and Thompson 1993). Both species also place the entrances of their burrows at the base of rock outcroppings (Oaks et al. 1987, Bihl and Smith 1998). Rugged terrain and large rocky outcroppings create a difficult work space for large machinery (i.e., bulldozers) to operate, and managers applying chaining treatments to an area often avoided rugged and rocky areas (McKenzie et al. 1984). Therefore, golden-mantled ground squirrel and rock squirrel may not use

areas where P-J was historically removed because of specific vegetation characteristics, but instead because suitable habitat structures (e.g., rock outcroppings) may not exist in areas that were chained.

We found that habitat use of certain mammal species was associated with specific vegetative characteristics. These relationships could be factored into management decisions to account for the long-term effects of P-J removal on mammal communities. For example, modern tree removal techniques (e.g., hydroax; Wästerlund and Hassan 1995) can be highly selective such that some trees and other vegetation can be retained (Wästerlund and Hassan 1995; Crow and van Riper 2010). We found that forest age (DBH/plot) had a positive influence on habitat use of mountain lions. Medium and large-bodied carnivores prefer a matrix of habitats (Koehler and Hornocker 1991, Dickson and Beier 2002), and have large home ranges that encompass a wide variety of habitat characteristics (Schonewald-Cox et al. 1991). Therefore, creating large-scale matrices of older trees within the treated areas might maintain habitat for medium and large-bodied carnivores. However, these actions would have to be done at a large scale, and may not be feasible with all tree reduction efforts. It is important to note that tree reduction projects have diverse goals and intended outcomes. Some objectives (e.g., creating habitat for greater sage grouse, which cannot tolerate standing trees; Baruch-Mordo et al. 2013) may not be compatible with the above recommendations for enhancing habitat quality for mammal species.

An important shortcoming of our analysis is that our metric of habitat use only accounts for detections of individual species and does not include density and abundance of mammal species in areas where P-J was historically removed. Assessing the long-term effects of P-J removal on the density of various animal populations should be a future priority to better understand the effects of mechanical tree removal on mammals in a slow recovering system.

Further, landscape-level characteristics may have a greater influence on mammal habitat use than fine scale vegetation parameters (Schonewald-Cox et al. 1991). Therefore, it may be important to incorporate landscape-scale attributes hypothesized to influence mammal habitat use, such as habitat characteristics at differing spatial scales or landscape connectivity metrics (see Kindlmann and Burel 2008).

We chose to use relatively simple habitat use models to examine main effects alone, without considering additive effects or interactions between vegetation, environmental, and anthropogenic characteristics. It is likely that interactions between vegetation parameters, landscape-scale features, or anthropogenic factors, such as intensity of energy development or size of mechanical disturbances have an influence on mammal habitat use. Future research should test for these potential interactions and additional parameters; and tree removal efforts in human dominated landscapes should explicitly account for these interactions in future management plans.

Finally, the use of chaining tapered off significantly after the 1970's (Romme et al. 2009, Redmond et al. 2013), making it difficult to measure changes in mammal communities over a chronological scale for this particular tree removal method. Additionally, the broad geographical distribution of P-J ecosystems encompasses a wide variety of soil types, precipitation, and elevation, which could affect rates and patterns of forest regeneration after disturbance (Tausch and Tueller 1977, Schott and Pieper 1987, Romme et al. 2009). Therefore, we urge caution in generalizing our results to other regions.

Tree reduction to increase forage for livestock and enhance habitat for game species had long-term consequences for mammal species, particularly mid-large sized carnivores, in our study region. Thus, if we are to maintain viable populations of diverse native species, future

management actions that include tree removal should explicitly measure intended and unintended effects on mammal communities. The hierarchical binomial-mixture model that we used can be applied more broadly to camera trap data to predict habitat use for non-target mammals based on individual habitat characteristics. This information could be used to adaptively guide forest management decisions, accounting for the costs and benefits of tree removal for different animal species. Given the magnitude of current anthropogenic pressure on ecosystems, it is critical that we understand the long-term effects of deliberate woodland reduction on native biodiversity.

CHAPTER FOUR

SIMULATING NATURAL DISTURBANCE AS HABITAT MITIGATION FOR ENERGY DEVELOPMENT HAS MIXED CONSEQUENCES FOR WOODLAND BIRDS AND MAMMALS

INTRODUCTION

Manipulating ecosystems to improve habitat quality and mitigate anthropogenic activities such as energy development is increasingly common. Global demand for energy production is predicted to increase by 40% in the next 20 years (International Energy Agency 2009), and this trend is likely to have profound effects on terrestrial wildlife. Oil shale deposits in the western U.S. hold particular promise and the exploitation of this resource is well underway (Carlson and Cringan 1975, Bartis et al. 2005). Development of oil and natural gas impacts wildlife by 1) habitat fragmentation resulting from the creation of extensive systems of roads and pipelines, 2) habitat loss due to drill pads and other energy infrastructure, 3) eliciting behavioral changes, particularly avoidance, due to development-related activity, and 4) inviting further fragmentation, resource extraction, and direct mortality of wildlife through increased human access to wild lands (Northrup and Wittemyer 2013). Understanding and mitigating for the impacts of energy development on wildlife will be a major challenge for ecologists in the coming decades (Northrup and Wittemyer 2013).

Assumed impacts from roads, habitat fragmentation, and introduction of invasive species due to energy development have led to large-scale habitat mitigation to sustain wildlife populations. This is particularly evident in the Piceance Basin in northwestern Colorado, U.S.A., which is currently experiencing an unprecedented level of natural gas development and provides

critical habitat for the largest migratory Mule Deer herd in the United States (Johnston 2009). Mature pinyon-juniper woodlands are considered poor foraging habitat for Mule Deer due to a lack of forbs and shrubs that sustain Mule Deer populations during winter months (Bartmann 1983). A common Mule Deer management objective is to increase the quantity and quality of preferred forage by reducing forest canopy cover (Bartmann 1983). This unique set of circumstances has lead land managers to seek mechanical means of improving Mule Deer habitat in the midst of one of the U.S.'s largest oil and gas fields.

Mechanical tree removal methods are strongly preferable to prescribed or natural fire in oil and gas fields, although fire has historically been an important part of the natural disturbance regime in pinyon-juniper ecosystems. Large fires would have occasionally altered forest structure and created openings that contain forbs, shrubs, and grasses (Romme et al. 2009). Fires in these systems are typically wind-driven, occurring in hot dry conditions, and resulting in the mortality of most trees at scales that can range from small groups of trees to areas that are hundreds of hectares in size (Floyd et al. 2004; Huffman et al. 2012). Large-scale fires reduce tree canopy and, if frequent enough, these ecosystems may convert to grasslands or shrublands (Miller & Tausch 2000). However, human driven processes such as livestock grazing, reduction of natural fire regimes, and climate change have caused encroachment, expansion, and persistence of woodlands into historically shrub-dominated areas (Miller and Tausch 2000, Romme et al. 2009, Huffman et al. 2012). Recent declines in bird and mammal populations dependent on open habitats as well as the risk of increased fuel loads from the encroachment of woody vegetation have given new justifications for emulating natural disturbances in land management activities (Baruch-Mordo et al. 2013, Redmond et al. 2013, Bergman et al. 2015).

In early 2013, pinyon-juniper woodlands in the Piceance Basin in northwestern Colorado, USA, were mechanically removed using a rotary masticator (Fig. 4.1A) as part of a collaborative project between XTO Energy, the Bureau of Land Management, Colorado Parks and Wildlife and Colorado State University to study the effects woodland removal has on the productivity of Mule Deer forage. This form of tree reduction is very selective and can potentially mimic the physical characteristics of a natural fire (Wästerlund and Hassan 1995). Rotary masticators chop and mulch standing vegetation close to the ground with minimal soil disturbance (Fig. 4.1B). However, mastication treatments leave a shallow amount of woody material and do not significantly change soil chemistry levels (Busse et al. 2014). Whereas, fire changes the chemical composition of soils, ultimately having an effect on plant recolonization and regeneration (Certini 2005).



Fig. 4.1. Tree reduction methods in pinyon-juniper ecosystems across the western United States: A) type of machinery used for reducing tree cover in pinyon-juniper woodlands, B) fine mulch layer left after mechanical tree reduction, and C) a representation of the habitat alteration approximately six months after pinyon-juniper removal in the Piceance Basin, northwest Colorado, USA (Photo credit: A) Jason Tack, B) Jason Tack, and C) Sara Bombaci).

Although mechanical disturbances may be meant to mimic natural disturbances (Long 2009), differences in vegetative characteristics or forest regeneration could have differing effects on the birds and mammals that have evolved to rely on particular vegetation structure and density (MacArthur and MacArthur 1961, Andr et al. 1994, Hobson and Schieck 1999). Birds and mammals can play an important role in structuring plant communities by providing ecological services such as seed dispersal (Wall 1997, Wunderle Jr 1997, Garcia et al. 2010) and

maintaining nutrient cycling (Holechek 1981), yet relatively little research has examined the effects of mechanical habitat disturbance as a form of habitat mitigation on bird and mammal communities (Bombaci and Pejchar 2016, Gallo and Pejchar 2016).

Because wildfires burned approximately 3,200 hectares throughout the Piceance Basin during the same time frame as mechanical tree removal occurred, we had the unique opportunity to investigate whether birds and mammals responded differently to mechanical and natural disturbance in the same area and on the same time-scale. Specifically, we evaluated bird and mammal community composition, bird densities, and mammal habitat use in the two years following mechanical disturbance and fires. Our research objectives were to 1) evaluate differences in bird and mammal species richness, community composition and community evenness between mechanically and naturally disturbed sites and undisturbed reference sites, 2) compare how mechanical and natural disturbance influence bird densities and mammal habitat use, and 3) identify important environmental and anthropogenic characteristics associated with differences in bird densities and mammal habitat use among sites. Our findings provide insight into whether mechanical disturbance can emulate natural processes to improve habitat for wildlife in areas undergoing rapid land use change.

MATERIALS & METHODS

Study Site

This study was conducted in a pinyon-juniper ecosystem in the Piceance Basin in northwest Colorado, U.S.A. (Fig. 4.2). Pinyon-juniper woodlands in this region consist of pinyon pine (*Pinus edulis*) and Utah juniper (*Juniperus osteosperma*), and dominate shrubs are antelope bitterbrush (*Purshia tridentata*), mountain mahogany (*Cercocarpus montanus*), big sagebrush (*Artemisia tridentata*), rabbitbrush (*Chrysothamnus* spp.), common chokecherry (*Prunus*

virginiana), and Saskatoon serviceberry (*Amelanchier alnifolia*) (O'Meara et al. 1981, Sedgwick 1987). The topography consists of high plateaus and deeply incised valleys, and elevation ranges from approximately 1800 to 2400 m. Dominant land use activities in the area include oil and gas extraction and domestic livestock grazing (Northrup et al. 2015). In January 2013, approximately 444 hectares of pinyon-juniper woodlands were mechanically removed within 147 mastication treatments (Fig. 4.1C). Between April and September 2012, 13 naturally-ignited wildfires burned approximately 3200 hectares in the Piceance Basin region before being suppressed by wildfire crews— offering a unique opportunity to quantify and compare the effects of mechanical disturbance and a natural disturbance (wildfires) on bird and mammal communities in the same region and over the same time period.

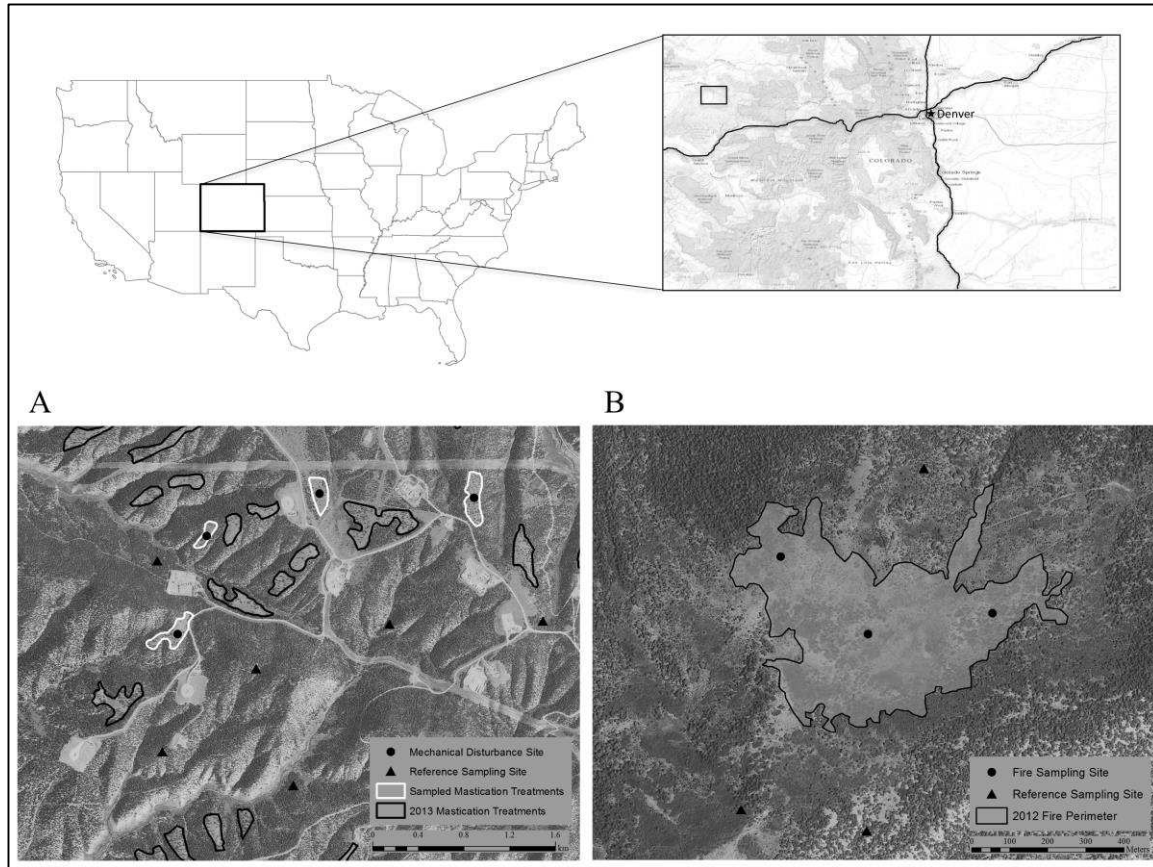


Fig. 4.2. Study area location in the Piceance Basin in northwestern Colorado, USA. Map (A) shows a representation of selected mechanical disturbance sites and reference sampling sites, and map (B) shows a representation of fire and fire references sampling site selection.

Study Design

Treatment and Reference Sites. – To compare bird densities and mammal habitat use between mechanically disturbed sites and wildfire sites, we established 25 sampling sites within each disturbed site type ($n = 50$). Due to the geographical extent of the fires and the clustering of the mechanically disturbed sites, we paired each disturbed site with an undisturbed reference site ($n = 50$; Fig. 4.2).

Mechanically Disturbed Sites. – We randomly selected 25 out of the 147 mastication treatments (ranging from 0.8-4.5 hectares) and established one sampling point in the approximate center of each treatment (“mechanically disturbed sites”; Fig. 4.2A). All selected mechanically

disturbed sites were no less than 450 m apart. Using geographical information system (GIS), we randomly placed 25 reference sites in the vicinity of the mechanically disturbed sites (Fig. 4.2A). Reference sites were buffered from all mechanically disturbed sites, including mastication treatments that were not sampled, by at least 250 m (Fig. 4.2A). Each reference site was ground-truthed to verify that it was within undisturbed pinyon-juniper woodlands.

Fire Sites. – Using data and aerial imagery obtained from the White River BLM office in Rio Blanco County, Colorado, we identified 9 accessible wildfires that ranged from 3.2-2072 hectares in size. We placed an initial sampling site within each fire area by choosing a random but accessible location in the approximate center of each area (“fire sites”; Fig. 4.2B). We then placed additional sampling sites in each cardinal direction from the initially selected site, such that the sites were at least 250 m apart (Fig. 4.2B). Because we began allocating sites in the smallest fire first, our design allowed for one site in the smallest fire area and up to 9 sites in the largest fire area for a total of 25 sites. Each fire site was paired with a reference site located in adjacent unburned pinyon-juniper woodland and buffered by 250 m from the fire’s edge. Paired reference sites were established in one of two ways depending on the location and accessibility of the fire. One scenario was that sites were established by starting at a randomly chosen fire sampling site, surveyors then walked in a random direction towards the fire’s edge and placed the first site 250 m or greater from the fire’s edge within the undisturbed landscape. Under this scenario, all subsequent reference sites were then placed ≥ 250 m apart continuing in the same direction away from the fire’s edge. In the second scenario a surveyor started at a fire site within a burned area and walked in a random direction towards the fire’s edge. One paired reference site was then placed >250 m from the fire’s edge within the undisturbed landscape (Fig. 4.2B). This was then performed for each sampling site within that fire area.

Bird Surveys

To compare bird densities among disturbed (mechanical and fire) and reference sites we conducted 5-minute individual observer-point counts at each of the 100 sampling sites (Dunn et al. 2006). Each site was surveyed 4 times/year between April-June for two years (2013 - 2014) by 3-4 trained observers. All birds were detected visually or aurally and their distance from the point count station was recorded. All surveys were conducted between 30 minutes after sunrise and no later than 1230, and surveys were discontinued during periods of fog, rain or high winds (>3 on Beaufort scale). To account for observer and temporal bias, observers and starting times were rotated throughout the survey season. Due to the small size of some mechanically disturbed and fire areas, some sampling sites were located near undisturbed forest (~35 m). However, only birds detected within the disturbed areas were recorded.

Mammals Surveys

To compare habitat use of mammals among mechanically disturbed, fire and reference sites we randomly selected 22 of the 25 sites where birds were sampled in each site type ($n = 88$ mammal sampling sites). We placed one unbaited remotely-triggered camera at each of these sites. To maximize detection probability, all cameras were located on wildlife trails, cattle trails, or unmaintained roads (e.g., two-track roads) set within a 100 m radius of each sampling point. We used Reconyx P800 ($n=64$; Reconyx, Holmen, WI, USA), Cuddaback Attack ($n=13$; De Pere, WI, USA), and Cuddaback Capture ($n=11$; De Pere, WI, USA) cameras. We allocated sampling effort evenly within each site type among the three camera models. All cameras were deployed between April and June 2014 and each camera collected relative-activity data (photos

per species per day) for 149 days following deployment. In order to ensure camera function, change batteries, and replace data cards, we checked each camera approximately every two weeks and downloaded photo data.

Volunteer observers were trained to identify species within each photo. Two observers viewed each photo and all animals were identified to species. To ensure consistency across species identifications, two authors (TG and LS) resolved all conflicting identifications between volunteers. Any animal that could not be identified to species was excluded from the analysis, with the exception of two chipmunk species, Least chipmunk (*Tamias minimus*) and Uinta chipmunk (*Tamias umbrinus*), which were difficult to differentiate by photograph and therefore collectively identified as Chipmunk. Photograph storage and management, species identifications, and generation of species occurrence data, were all facilitated by the Colorado Parks and Wildlife Photo Warehouse (Ivan and Newkirk 2015). Camera data was standardized *post hoc* to include a 30-second quiet period between subsequent photos to ensure trigger rate between camera types was the same.

Vegetation Surveys

To assess the relationships between bird density and mammal habitat use and habitat characteristics, we measured a variety of vegetative parameters. In 2013, we conducted surveys in a 10x10-m plot, offset from each point count site by ~5 m in a random direction (McElhinny et al. 2005). Species, tree height, crown area, diameter at breast height (DBH), and condition (i.e. live, dead, snag broken above 1 m) were recorded for each tree in the 10x10 m plot following the methods used in Huffman et al. (2012). Trees (live and dead) were defined as individuals with a height >1 m (Huffman et al. 2012). We estimated an index of forest stand age by dividing the total DBH of all trees within the plot by the plot size (DBH/100 m²; modified from McElhinny et

al. 2005). Percent tree cover was calculated by dividing total area of canopy cover within plot by plot size (100 m²). In 2013 and 2014, shrub, grass and forb species percent cover were measured using a 25-m line intercept beginning at each point count site and heading in a random compass direction (Canfield 1941).

Differences between years for each vegetation characteristic were tested using ANOVA in R (R Core Team 2015), and no significant differences were found. Therefore, the two years of vegetation data for each variable at each site were averaged and these values were incorporated into the analyses described below. We further tested for differences in each vegetation characteristic between each reference type (mechanical and fire) using a two-sample *t*-test and found no significant differences. Therefore, the two sets of reference sites ($n = 50$) were not differentiated in our statistical analyses.

Statistical Analysis

Bird diversity and species richness. – Bird detections were truncated at 52 m (radius of smallest mechanically disturbed area) to ensure similar sampling effort and plot sizes at each site and to ensure independence from adjacent sites (Magurran 2004). We calculated pooled species richness for each site type using the non-parametric Chao gamma diversity estimator (Chao 1987), and Shannon diversity index (Shannon 1948) for each site type using the R packages *vegan* and *BiodiversityR* (Kindt and Coe 2005, Oksanen et al. 2015, R Core Team 2015). To account for species-specific detection probabilities we adjusted our bird count data for each species by dividing the total number of detections at each site (C_i) by the median value of detection probabilities at each site (p_i) averaged across the four surveys. Median values were calculated from the posterior distributions of detection probabilities estimated from the species-specific binomial-mixture models described below. We used a borrowing rule based on

maximum detection distance and the vegetation strata where the species generally displays to adjust count data for species that were too rare to estimate detection probability (Alldredge et al. 2007, Rodewald 2015). Species were first categorized into 4 general groups: 1) maximum detection distance of ≤ 50 m, 2) maximum detection distance of > 50 m and ≤ 100 m, 3) > 100 m and ≤ 150 m, and 4) > 150 m. We further grouped species into 4 secondary groups: 1) species generally displays near the ground, 2) species generally displays in the mid-level vegetation strata, 3) species generally displays on top of vegetation, and 4) species generally displays in the air. Detection probabilities were then shared within groups (Table A4.1). We compared mean species richness at each site between mechanically disturbed, fire, and reference sites using ANOVA with a *post hoc* Tukey's honest significant difference (HSD) test in R (R Core Team 2015).

Bird and mammal community composition. – To compare community composition of birds between disturbed and reference sites we used our adjusted count data and calculated the Chao dissimilarity index between mechanically disturbed, fire, and reference sites using *vegan* (Oksanen et al. 2015; R Core Team 2015). The Chao dissimilarity index is a probabilistic approach that considers species abundance, and is commonly used in place of the classic Jaccard and Sørensen incidence-based similarity indices when count data is available (Chao et al. 2005). To compare community composition of mammals between mechanically disturbed, fire, and reference sites we calculated the Jaccard's dissimilarity index ($1 - C_J$) for binary presence/absence data using *vegan* (Oksanen et al. 2015; R Core Team 2015). Jaccard's similarity index (C_J) is a common method for describing the similarity in community

composition between two sites or treatments (Magurran 2004). In both the Chao and Jaccard dissimilarity indices, treatments that have no species in common are given an index of 1, and treatments that have identical community composition are given an index of 0.

Quantifying the effect of disturbance and habitat characteristics on bird density and mammal habitat use. – We used hierarchical open population binomial-mixture models (Kéry and Andrew Royle 2010, Kéry and Schaub 2012) to quantify the effect of mechanical disturbance and habitat characteristics on bird densities. Binomial-mixture models estimate abundance using repeated count data while taking into account imperfect detection (Kéry and Schaub 2012). Thus, they contain more information than simply estimating an occurrence/non-occurrence response, similar to the widely used occupancy modeling framework (MacKenzie et al. 2006, Kéry and Schaub 2012). Bird detections were truncated at a 52 m radius from the point count station – making all surveyed areas 0.85 ha. Therefore, we infer our results as bird density (individuals/ha).

We used a single-season hierarchical binomial-mixture model, parameterized by Royle and Dorazio (2008), to address our objectives of comparing habitat use between disturbance and reference sites for each mammal species. These models are traditionally used to estimate the number of individuals in a population (Kéry and Schaub 2012). However, we exchanged the number of individuals for a given species with the daily activity (photos per day) of that species. Thus, allowing us to estimate a more robust calculation of habitat use compared to an occurrence/non-occurrence response commonly implemented for camera trap data of unmarked individuals (MacKenzie et al. 2006, O'Connell and Bailey 2011, Kéry and Schaub 2012).

Modeling the effects of mechanical disturbance and fire on bird density. – To quantify the effect of mechanical disturbance and fire on the density of each bird species, we let y_{ijk} be the total number of individuals counted at site i , during survey j , in year k (2013 and 2014). We assumed closure over the course of each year; therefore, observed counts would arise as a binomial random variable,

$$y_{ijk} \sim \text{binomial}(N_{ik}, p_{ijk}),$$

where N_{ik} is the total number of individuals available to be counted in year k at site i , and p_{ijk} is the survey specific detection probability. We then modeled our latent variable N_{ik} (Table A4.2) as a Poisson random variable,

$$N_{ik} \sim \text{Poisson}(\lambda_{ik}),$$

where λ_{ik} is the expected density at site i for year k . To quantify the influence of disturbances on the density of bird species, we modeled λ_{ik} as a function of disturbance type at site i using a log link,

$$\log(\lambda_{ik}) = \omega_{g[i]} + X_i\beta.$$

Our data matrix (X_i) was set up as a design matrix so that effect size could be compared between disturbance types. To account for potential spatial dependency that is not accounted for by the disturbance type and the Poisson assumption, we used a multilevel model (Gelman and Hill 2007) to include a random effect ($\omega_{g[i]}$) on geographical grouping ($g = 11$). Mechanically disturbed sites and their associated references sites were spatially aggregated and were put into a single group (Fig. 4.1A). All sampling sites within each fire area were combined with the paired

references sites at that fire and placed into a single-group (Fig. 4.1B). We modeled the group level parameters using a normal distribution and allowing each group to have a common mean (μ_ω) and standard deviation (σ_ω),

$$\omega_g \sim \text{normal}(\mu_\omega, \sigma_\omega^2).$$

This distribution creates a partial-pooling effect by drawing the estimates of ω_g toward the mean level (μ_ω), but not entirely (Gelman and Hill 2007). Based on previous analyses (T. Gallo unpublished data), we had reason to believe that observers conducting point count best explained variation in detection probability for all species. Therefore, we modeled the detection probability as a function of the observer conducting the survey at site i , survey j , and year k on the logit scale:

$$\text{logit}(p_{ijk}) = \alpha_o + \alpha_1 \text{observer}_{ijk}$$

Conventional ‘vague’ priors were used for all parameters. Specifically, we assumed $\beta \sim \text{normal}(0,100)$, $\alpha \sim \text{normal}(0,100)$, $\mu \sim \text{normal}(0,100)$, and $\sigma \sim \text{uniform}(0,100)$. One fire area containing 3 fire and 3 reference sampling sites was removed from our bird density analysis because this area was inaccessible in 2013.

Modeling the effect of vegetation and environmental parameters on bird density. – For those species showing a strong positive or negative response to mechanical disturbance and/or fire (90% credible intervals not overlapping 0) we used a similar hierarchical open population binomial-mixture model (Kéry and Andrew Royle 2010, Kéry and Schaub 2012) to quantify the effect of habitat characteristics on bird density.

$$\log(\lambda_{ik}) = \omega_{g[i]} + X_i' \beta$$

However, in this model our independent variables (X'_i) are continuous variables scaled to have mean 0 and standard deviation 1 (Gelman et al. 2008). We chose vegetation, environmental, and anthropogenic covariates that we hypothesized may best explain variation in bird density based on life history strategies and behaviors of each species, such as foraging (De Graaf et al. 1985) and nesting guilds (Degraaf and Wentworth 1986) and habitat preference (Rodewald 2015) (Appendix A4.1). Our vegetation covariates included proportion of shrub, grass, tree cover, and bare ground, forest age, and number of snags (standing dead or dying tree) within plot. Environmental and anthropogenic covariates included elevation calculated from a digital elevation model, size of disturbed area, and well pad density (number of natural gas well pads located within 1km of the point count station (Toms et al. 2005); see Northrup et al. (2015) for natural gas well pad digitization methods). Each of these covariates was calculated using ArcGIS 10.3 (Esri, Redland, CA, USA). All covariates were tested for correlation and no highly correlated covariates ($|r| > 0.7$) appeared in the model.

Modeling the effects of mechanical and natural disturbance on mammal habitat use. – To quantify the effects of mechanical disturbance and fire on mammal habitat use, we let y_{ij} be the total number of photos counted each day j , at site i ,

$$y_{ij} \sim \text{binomial}(A_i, p_i),$$

where A_i is the total number of species-specific detections counted at site i , and p_i is the site specific detection probability. We then modeled the latent variable A_i as a Poisson random variable,

$$A_i \sim \text{Poisson}(\lambda_i),$$

where λ_i is the expected number of occurrences at site i . To quantify the effect of mechanical disturbance and fire on mammal habitat use, we modeled λ_i as a function of disturbance type at each site using a log link,

$$\log(\lambda_i) = \omega_{g[i]} + X_i\beta_i.$$

Again, our data matrix (X_i) was set up as a design matrix so that the effect size of each disturbance type could be compared. To account for potential spatial dependency we again used a multilevel model to include a random effect $\omega_{g[i]}$ on geographical grouping. We used the same geographical groupings described above, and modeled the group level parameters using a normal distribution and allowing each group to have a common mean (μ_ω) and standard deviation (σ_ω),

$$\omega_g \sim \text{normal}(\mu_\omega, \sigma_\omega^2).$$

Detection radius varied by camera type, so we modeled the detection probability as a function of camera type (*camera*) at each site on the logit scale:

$$\text{logit}(p_i) = \alpha_o + \alpha_1 \text{camera}_i$$

Vague priors were used for all parameters. Specifically, we assumed $\beta \sim \text{normal}(0,100)$, $\alpha \sim \text{normal}(0,100)$, $\mu \sim \text{normal}(0,100)$, and $\sigma \sim \text{uniform}(0,100)$. All sites were included in the mammal analyses.

Modeling the effect of vegetation and environmental parameters on mammal habitat use.

– To assess the effects of vegetation and environmental parameters on mammal habitat use, we used a similar hierarchical binomial-mixture model as previously described. However, in this

model X'_i represents continuous variables scaled to have mean 0 and standard deviation 1 (Gelman and Hill 2007). We chose vegetation, environmental, and anthropogenic covariates that we hypothesized to be important predictors of mammal habitat use.

$$\log(\lambda_i) = \omega_{g[i]} + X'_i\beta$$

The vegetation covariates included proportion of shrub, grass, and tree cover, and forest age. Environmental covariates were in relation to each camera location and included elevation, size of the disturbed area, distance to the nearest road for each site, and number of natural gas well pads within 500 m (general home range size of sciurid species; Harris and Leitner 2004) of each site (well pad density). We tested for correlations among covariates to ensure that no covariates were highly correlated ($|r| > 0.7$).

Model fitting and estimations. – Posterior distributions of model coefficients were estimated using Markov chain Monte Carlo (MCMC) methods implemented in JAGS using the *rjags* package in R (Gelman et al. 2008). ‘Vague’ priors were used for all model parameters. Bird density models were run with 3 parallel chains of 200,000 iterations each, a thinning rate of 1, and a burn-in rate of 50,000. Mammal habitat use models were run with 3 parallel chains and a minimum of 100,000 iterations to a maximum of 300,000 iterations (depending on species) and the first 50,000-100,000 (depending on iterations) were discarded as burn-in. Model convergence was assessed by checking that the Gelman-Rubin diagnostic statistic for each parameter was <1.1 (Plummer et al. 2006, R Core Team 2015) and visually inspecting the trace plots of MCMC samples. Due to both, a small sample size and the desire to suggest reasonable management recommendations, we chose to use relatively simple models to examine main effects alone, without considering additive effects or interactions. Model fit was assessed using post predictive

checks described in (Kéry and Schaub 2012). To assess the relative influence of covariates on bird density and mammal habitat use, we compared the posterior distributions of model coefficients and examined overlap of posterior distributions with 0.

RESULTS

Bird Community Composition and Species Richness

A total of 56 bird species were detected across all site types during 2013-2014 (Table A4.3). Thirty-five bird species were observed in mechanically disturbed sites, 41 species in fire sites, and 51 species in reference sites. Audubon's Yellow-rumped Warbler (*Setophaga coronata*) was detected in both disturbance types, but absent from reference sites. Bewick's Wren (*Thryomanes bewickii*), MacGillivray's Warbler (*Geothlypis tolmiei*), Northern Harrier (*Circus cyaneus*), Orange-crowned Warbler (*Oreothlypis celata*), Red-tailed Hawk (*Buteo jamaicensis*), Western Tanager (*Piranga ludoviciana*), and White-throated Swift (*Aeronautes saxatalis*) were only detected in reference sites. Clark's Nutcracker (*Nucifraga columbiana*), Common Nighthawk (*Chordeiles minor*), House Finch (*Haemorhous mexicanus*), Lark Sparrow (*Chondestes grammacus*), Pinyon Jay (*Gymnorhinus cyanocephalus*), Say's Phoebe (*Sayornis saya*), Vesper Sparrow (*Pooecetes gramineus*), Virginia's Warbler (*Leiothlypis virginiae*), Western Meadowlark (*Sturnella neglecta*), and Western Wood Pewee (*Contopus sordidulus*) were never detected in mechanical disturbance sites; and Black-chinned Hummingbird (*Archilochus alexandri*), Bushtit (*Psaltiriparus minimus*), Cooper's hawk (*Accipiter cooperii*), Dark-eyed Junco (*Junco hyemalis*), Hermit Thrush (*Catharus guttatus*), Lesser Goldfinch (*Spinus psaltria*), Ruby-crowned Kinglet (*Regulus calendula*), and Western Scrub Jay (*Aphelocoma californica*) were never detected in fire sites. Bullock's Oriole (*Icterus bullockii*), Canyon Wren (*Catherpes mexicanus*), Lincoln Sparrow (*Melospiza lincolnii*), and Western Bluebird (*Sialia*

mexicana) were only detected in fire sites. No species was only detected in mechanical disturbance sites.

Across both years, bird community composition between all three site-types had Chao dissimilarity indices lower than 0.5 (Table 4.1). Chao gamma diversity estimates were 38.76 (SE = 6.24) and 46.12 (SE = 4.75) for mastication treatments and fire sites respectively and 44.12 (SE = 4.75) for reference sites. Mean species richness did not differ between disturbance types or reference (Table 4.1). Shannon-Wiener indices for mechanically disturbed and fire sites were 2.97 and 2.6, respectively and 2.74 for reference sites.

Table 4.1. Indices comparing bird (Chao index) and mammal (Jaccard binary index) community composition, and bird species richness (Tukey’s HSD test) in mechanically disturbed, fire and undisturbed reference sites in the Piceance Basin, Colorado.

Taxa	Treatment	Dissimilarity Index		Tukey’s HSD test		
			Difference	Lower CI	Upper CI	p-value
<i>Birds</i>	Reference - Mastication	0.14	-0.44	-2.42	1.54	0.86
	Reference - Fire	0.09	-1.08	-3.06	0.9	0.4
	Mastication - Fire	0.16	-0.64	-2.62	1.34	0.72
<i>Mammals</i>	Reference - Mastication	0.31				
	Reference - Fire	0.35				
	Mastication - Fire	0.25				

Effect of Mechanical Disturbance and Fire on Bird Densities

We were able to appropriately fit both open population binomial-mixture models to 12 bird species (Fig. 4.3). Blue-gray Gnatcatcher (*Polioptila caerulea*), Brewer’s sparrow (*Spizella breweri*), *Empidonax* flycatcher, Green-tailed Towhee (*Pipilo chlorurus*), Spotted Towhee (*Pipilo maculatus*), and White-breasted Nuthatch (*Sitta carolinensis*) all showed a strong negative response to fire sites; and Hairy Woodpecker (*Leuconotopicus villosus*) and Lark Sparrow (*Chondestes grammacus*) showed a strong positive response to fire sites. However, these species showed little to no response to mechanical disturbance. Mourning Dove (*Zenaida*

macroura) had a strong negative response to both disturbance types, and Mountain Bluebird (*Sialia currucoides*) had a strong positive response to both disturbance types. Broad-tailed Hummingbird (*Selasphorus platycercus*) and House Wren (*Troglodytes aedon*) showed little to know response to either disturbance type.

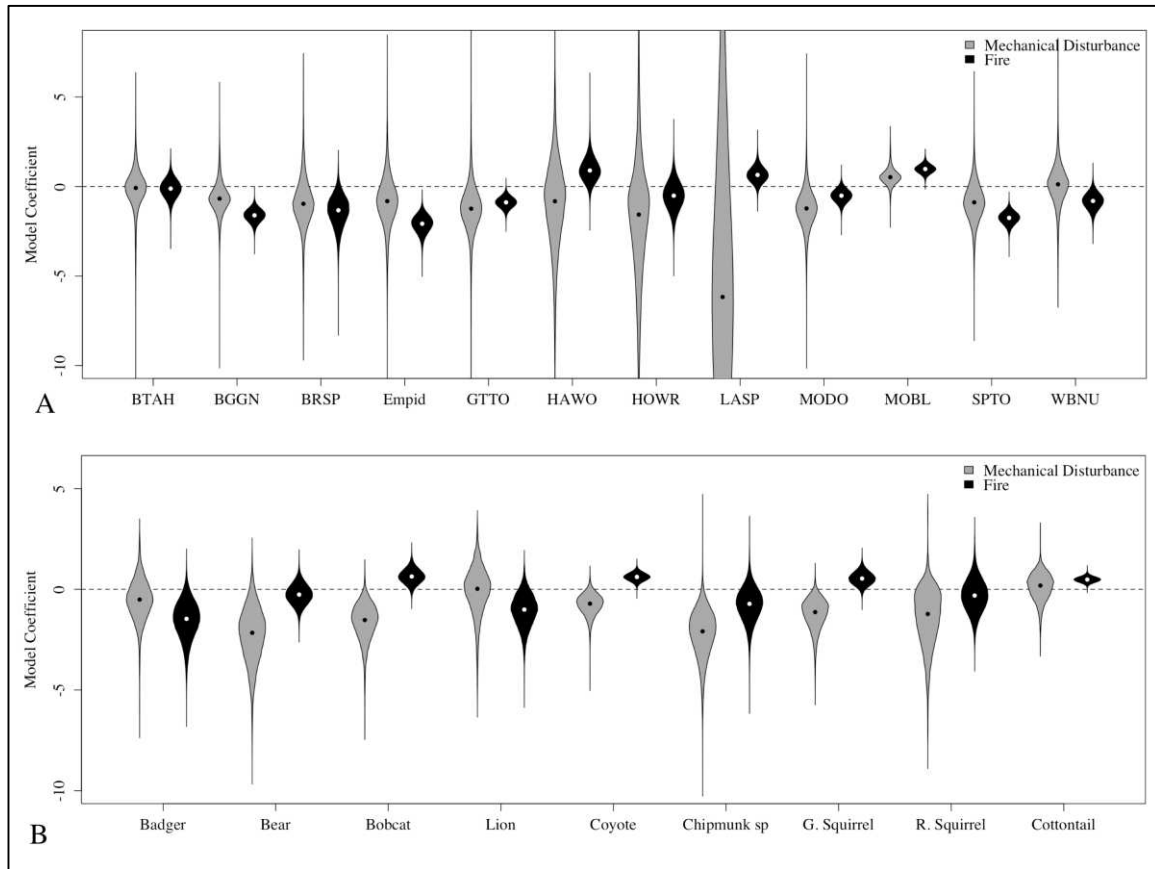


Fig. 4.3. Posterior distributions of model coefficient values for the effect of mechanical disturbance and fire on A) bird density and B) mammal habitat use in the Piceance Basin, CO. Dashed line indicates coefficient value of 0. Dot in the center of each distribution represents the median value. Abbreviations are: BTAH, Broad-tailed Hummingbird; BGGN, Blue-gray Gnatcatcher; BRSP, Brewer’s Sparrow, Empid, *Empidonax* flycatcher; GTTO, Green-tailed Towhee; HAWO, Hairy Woodpecker; HOWR, House Wren; LASP, Lark Sparrow; MODO, Mourning Dove; MOBL, Mountain Bluebird; SPTO, Spotted Towhee; and WBNU, White-breasted Nuthatch.

Influence of vegetation and environmental parameters on bird densities

Most measured vegetation characteristics differed to some degree between mechanical disturbance sites, fire sites, and reference sites (Table 4.2). The magnitude and direction of effect of vegetation and environmental parameters on bird densities differed by species (Table 4.3). The greatest drivers of habitat use for Blue-gray Gnatcatcher were proportion of shrub cover (+) and size of disturbed area (-). Brewer's Sparrow density was most influenced by tree cover (+) and forest stand age (-), and *Empidonax* flycatcher density was most influenced by well pad density (+) and elevation (-). Elevation had the greatest positive relationship with Green-tailed Towhee density, and size of disturbed area had the greatest negative relationship. Hairy Woodpecker density was most influenced by number of snags (+) and grass cover (-). The greatest drivers of Lark Sparrow density use was tree cover (+) and well pad density (-). Mountain Bluebird density was most influenced by proportion of bare ground (+) and shrub cover (-). The greatest driver of Morning Dove density was grass cover (+), and the greatest driver of Spotted Towhee density was shrub cover (+) and number of snags (-). White-breasted Nuthatch density was most influenced by tree cover (+) and size of disturbed area (-).

Table 4.2. Vegetation parameters (means and 95% CI's) in mechanically disturbed, fire and undisturbed sites in the Piceance Basin, Colorado.

Parameter	Mech. Dist	95% Confidence Intervals		Fire	95% Confidence Intervals		Reference	95% Confidence Intervals	
		Lower	Upper		Lower	Upper		Lower	Upper
Mean shrub cover	0.30	0.22	0.39	0.11	0.06	0.16	0.36	0.27	0.46
Mean grass cover	0.08	0.06	0.10	0.09	0.06	0.13	0.10	0.07	0.12
Mean proportion of bare ground	0.23	0.17	0.29	0.54	0.47	0.61	0.35	0.30	0.40
Mean forest cover	0.07	0.04	0.09	0.05	0.02	0.09	0.26	0.20	0.33
Mean DBH/plot (cm)	0.06	0.01	0.12	0.24	0.11	0.37	0.72	0.52	0.92
Mean number snags	0.84	0.00	1.78	3.18	0.84	5.53	0.45	0.22	0.67

Table 4.3. Median posterior distribution values and proportion of the posterior distribution that lies below and above 0 from the hierarchical open population binomial-mixture model used to quantify the effects of finer-scale vegetation characteristics (A), course-scale woodland characteristics (B), and environmental and anthropogenic (B) covariates on bird densities in the Piceance Basin in northwest Colorado, USA. *Italic bolding* indicates the most influential positive and negative covariates for each species.

A. Species	<i>Shrub cover</i>			<i>Grass cover</i>			<i>Proportion of bare ground</i>		
	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0
Blue-gray Gnatcatcher	0.32	0.00	1.00	0.20	0.03	0.97	0.00	0.49	0.51
Brewer's Sparrow	0.52	0.01	0.99	0.08	0.37	0.63	-0.03	0.54	0.46
<i>Empidonax</i> flycatcher	-0.15	0.81	0.19	-0.16	0.90	0.10	-0.19	0.90	0.10
Green-tailed Towhee	0.32	0.00	1.00	0.12	0.20	0.80	-0.02	0.55	0.45
Hairy Woodpecker	-0.50	0.94	0.06	-1.09	1.00	0.00	-0.18	0.73	0.27
Lark Sparrow	-1.74	0.99	0.01	0.69	0.01	0.99	-0.04	0.54	0.46
Mountain Bluebird	-0.33	0.99	0.01	0.33	0.00	1.00	0.46	0.00	1.00
Morning Dove	0.22	0.11	0.89	0.37	0.00	1.00	-0.10	0.73	0.27
Spotted Towhee	0.66	0.00	1.00	-0.09	0.77	0.23	0.13	0.16	0.84

	White-breasted Nuthatch	-0.38	0.99	0.01	-0.21	0.93	0.07	-0.33	0.99	0.01
B.	Species	<i>Tree cover</i>			<i>Forest stand age</i>			<i># of snags</i>		
	Blue-gray Gnatcatcher	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0
	Brewer's Sparrow	0.40	0.00	1.00	0.00	0.50	0.50	-0.06	0.67	0.33
	<i>Empidonax</i> flycatcher	0.91	0.01	0.99	-2.34	1.00	0.00	0.19	0.19	0.81
	Green-tailed Towhee	-0.09	0.70	0.30	0.19	0.11	0.89	0.05	0.38	0.62
	Hairy Woodpecker	0.26	0.15	0.85	-0.34	0.91	0.09	-0.06	0.69	0.31
	Lark Sparrow	-0.15	0.65	0.35	0.01	0.49	0.51	0.29	0.03	0.97
	Mountain Bluebird	0.82	0.03	0.97	-1.32	1.00	0.00	0.52	0.25	0.75
	Morning Dove	-0.02	0.54	0.46	0.09	0.32	0.68	0.19	0.02	0.98
	Spotted Towhee	0.02	0.47	0.53	0.07	0.39	0.61	-0.07	0.67	0.33
	White-breasted Nuthatch	0.34	0.02	0.98	0.14	0.22	0.78	-0.40	0.99	0.01
		0.42	0.00	1.00	-0.25	0.94	0.06	0.04	0.37	0.63
C.	Species	<i>Elevation</i>			<i>Disturbance size</i>			<i>Well pad density</i>		
	Blue-gray Gnatcatcher	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0
	Brewer's Sparrow	-0.04	0.59	0.41	-0.89	1.00	0.00	0.03	0.40	0.60
	<i>Empidonax</i> flycatcher	-0.77	1.00	0.00	-0.69	1.00	0.00	0.45	0.00	1.00
	Green-tailed Towhee	0.63	0.00	1.00	-0.61	0.99	0.01	-0.28	0.94	0.06
	Hairy Woodpecker	-0.07	0.58	0.42	-0.76	0.93	0.07	0.15	0.27	0.73
	Lark Sparrow	0.53	0.13	0.87	-0.33	0.93	0.07	-4.07	1.00	0.00
	Mountain Bluebird	0.30	0.01	0.99	0.09	0.21	0.79	0.13	0.12	0.88

Morning Dove	-0.19	0.79	0.21	0.05	0.39	0.61	-0.01	0.52	0.48
Spotted Towhee	0.22	0.05	0.95	-0.11	0.68	0.32	0.03	0.39	0.61
White-breasted Nuthatch	0.24	0.08	0.92	-0.97	0.98	0.02	0.15	0.11	0.89

Mammal Community Composition

Twenty-six mammal species, including human (*Homo sapien*), domestic dog (*Canis lupus familiaris*), domestic cattle (*Bos taurus*), and domestic sheep (*Ovis aries*) were detected by camera traps across all sites. Eighteen mammal species were observed in mechanical disturbance sites, 17 species in fire sites, and 26 species in reference sites (Table A4.4). Pronghorn (*Antilocapra americana*) and Gray Fox (*Urocyon cinereoargenteus*) were detected in reference sites and fire sites, but were not detected in mechanical disturbance sites. Chipmunk sp., Mountain Lion (*Puma concolor*), and Long-tailed Weasel (*Mustela frenata*) were detected in reference sites and mechanical disturbance sites, but were not detected in fire sites; and Striped Skunk (*Mephitis mephitis*), Western Spotted Skunk (*Spilogale gracilis*), White-tailed Jackrabbit (*Lepus townsendii*), Bushy-tailed Woodrat (*Neotoma cinerea*), and Wyoming Ground Squirrel (*Urocitellus elegans*) were detected in reference sites, but were not detected in either disturbance type. Mammal community composition between all three site-types had Jaccard dissimilarity indices lower than 0.5 (Table 4.1)

Effects of Mechanical and Natural Disturbances on Mammal Habitat Use

We were able to appropriately fit our hierarchical binomial-mixture model testing the effects of mechanical disturbance and wildfire on mammal habitat use to nine mammal species (Fig. 4.3). Bobcat (*Lynx rufus*), Coyote (*Canis latrans*), Mountain Cottontail (*Sylvilagus nuttallii*), and Golden-mantled Ground Squirrel (*Callospermophilus lateralis*) all showed a strong positive response to fire sites, whereas American Badger (*Taxidea taxus*) and Mountain Lion showed a strong negative response to fire sites. American Black Bear (*Ursus americanus*), Bobcat, Coyote, Golden-mantled Ground Squirrel, and Chipmunk sp. all showed a strong negative response to mechanical disturbance. No species showed a strong positive response to

mechanical disturbance. Rock Squirrel (*Otospermophilus variegatus*) showed little to no response to either disturbance type.

Influence of vegetation and environmental parameters on mammal habitat use

We were able to appropriately fit our hierarchical binomial-mixture model testing the effects of vegetation and environmental parameters on mammal habitat use to all of the previously listed species with the exception of American Black Bear. The magnitude and direction of effect of vegetation and environmental parameters on mammal habitat use differed by species (Table 4.4). The greatest drivers of habitat use for American Badger were proportion of grass cover (+) and size of disturbed area (-). Bobcat habitat use was most influenced by forest stand age (+) and tree cover (-), and Mountain Lion habitat use was most influenced by elevation (+) and distance to road (-). Distance to road had the greatest positive influence on Coyote habitat use, and shrub cover had the greatest negative influence. The greatest drivers of Mountain Cottontail habitat use were the size of disturbed area (+) and distance to road (-), and the greatest driver of Golden-mantled Ground Squirrel habitat use was the size of disturbed area (+) and well pad density (-). Rock Squirrel habitat use was most influenced by the size of disturbed area (+) and well pad density (-). Size of disturbed area had the greatest positive influence on Chipmunk habitat use, and proportion of grass cover had the strongest negative influence.

Table 4.4. Median posterior distribution values and proportion of the posterior distribution that lies below and above 0 from the hierarchical binomial-mixture model used to quantify the effects of vegetation (A) and environmental and anthropogenic (B) covariates on mammal habitat use in the Piceance Basin in northwest Colorado, USA. *Italic bolding* indicates the most influential positive and negative covariates for each species.

A. Species	<i>Grass Cover</i>			<i>Shrub Cover</i>			<i>Tree Cover</i>			<i>Forest Stand Age</i>		
	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0
Am. Badger	0.70	0.00	1.00	0.11	0.30	0.70	0.39	0.09	0.91	-0.19	0.70	0.30
Bobcat	-0.29	0.96	0.04	-0.08	0.67	0.33	-0.70	1.00	0.00	0.77	0.00	1.00
Mnt. Lion	0.01	0.47	0.53	0.19	0.15	0.85	0.21	0.11	0.89	0.16	0.24	0.76
Coyote	0.07	0.23	0.77	-0.11	0.84	0.16	-0.04	0.60	0.40	-0.08	0.68	0.32
Ground Squirrel	-0.03	0.60	0.40	0.81	0.00	1.00	0.08	0.35	0.65	0.63	0.00	1.00
Rock Squirrel	-0.12	0.68	0.32	-0.50	0.98	0.02	-0.43	0.89	0.11	0.77	0.02	0.98
Chipmunk	-1.02	0.99	0.01	0.23	0.28	0.72	-0.76	0.84	0.16	0.21	0.39	0.61
Mnt. Cottontail	0.07	0.07	0.93	0.22	0.00	1.00	0.10	0.04	0.96	0.22	0.00	1.00
B. Species	<i>Elevation</i>			<i>Distance to Road</i>			<i>Size of disturbed area</i>			<i>Well density</i>		
	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0	Coeff.	Prop.<0	Prop.>0
Am. Badger	0.20	0.25	0.75	-0.20	0.70	0.30	-2.77	0.98	0.02	0.03	0.46	0.54
Bobcat	0.25	0.07	0.93	-0.39	0.95	0.05	0.32	0.05	0.95	-0.47	0.99	0.01
Mnt. Lion	0.34	0.09	0.91	-0.27	0.80	0.20	0.14	0.38	0.62	0.09	0.30	0.70
Coyote	0.20	0.05	0.95	0.24	0.03	0.97	0.12	0.17	0.83	-0.09	0.75	0.25
Ground Squirrel	0.11	0.23	0.77	0.14	0.21	0.79	4.89	0.00	1.00	-0.69	1.00	0.00
Rock Squirrel	-0.26	0.84	0.16	-0.13	0.65	0.35	0.85	0.08	0.92	-0.62	0.99	0.01
Chipmunk	0.44	0.20	0.80	-0.87	0.88	0.12	0.78	0.32	0.68	0.19	0.35	0.65
Mnt. Cottontail	-0.05	0.83	0.17	-0.36	1.00	0.00	0.56	0.00	1.00	-0.15	1.00	0.00

DISCUSSION

Land use change from energy development and other anthropogenic activities is having a global impact on ecosystems (Foley et al. 2005, Fuschs et al. 2015, Northrup et al. 2015). Increasingly, these ecosystems are further manipulated to offset impacts on species of economic or conservation concern. Where historical disturbance regimes have been disrupted, habitat mitigation may mimic natural disturbance, with unknown consequences for diverse bird and mammal communities (Gallo and Pejchar 2016). We compared the effects of mechanical disturbance and wildfires on birds and mammals in the Piceance Basin in northwestern Colorado. We found that bird and mammal community composition did not differ between mechanically disturbed and fire sites, but bird densities and mammal habitat use were markedly different for many taxa. Our results highlight the need to consider the unintended consequences of simulating natural disturbance on animal communities.

We found no significant difference in species richness or diversity between mechanically disturbed sites, fire sites, and reference sites. These results are somewhat surprising as others have demonstrated that species richness and diversity either decreases because of habitat loss due to disturbance and fragmentation (Gascon et al. 1999), or increase because new species colonize new habitat types created by disturbance (Malavasi et al. 2009). Bird and mammal community dissimilarity indices between both disturbance types and reference sites were low (<0.05); indicating that species composition in disturbed sites were similar to species composition in reference sites and species turnover was low. Keller et al. (2003) did not see a significant decrease in bird species richness until 15-25 years after mechanical tree removal in northwestern U.S., and Fisher and Wilkinson (2005) demonstrated that it can take years before seeing the effects of forest fires and timber harvests on mammals in boreal forests. Our study was

conducted within the first two years following disturbance; thus, species composition may still differentiate between mechanical disturbance sites, fire sites, and reference sites over time (Gallo et al. in prep – Chapter 3).

Although we did not detect differences in community composition or diversity during the timeframe of our study, mechanical disturbance and wildfires had strong and divergent effects on the density and habitat use of some bird and mammal species, respectively. Only Mountain Bluebird and Mourning Dove densities responded positively to mechanical disturbance; whereas the densities of 10 bird species responded positively to fire, and we further found marked differences in habitat use between mechanically disturbed sites and fire sites for most mammal species (Fig. 4.3).

Habitat structure and heterogeneity plays in an important role in shaping bird abundance (MacArthur and MacArthur 1961), and mechanically disturbed and fire sites differed in their vegetative structure and cover. Fire sites had little canopy cover (5%) and a large proportion of bare ground (54%). Whereas, some level of vegetative cover and structure was retained after mechanical tree reduction (e.g., tree (7%) and shrub cover (30%)). Blue-gray gnatcatcher, Brewer's Sparrow, *Empidonax* flycatcher, Green-tailed Towhee, Mourning Dove, Spotted Towhee, and White-breasted Nuthatch all showed a strong negative response to fire sites, and all of these species rely on vegetative cover and structure for foraging or nesting (De Graaf et al. 1985, Degraaf and Wentworth 1986). Shrub and tree cover had strong influence on the density of many of these species (Table 4.3); tree cover was the most important habitat characteristic influencing Blue-gray Gnatcatcher and White-breasted Nuthatch densities. Therefore, selectively leaving structurally diverse vegetation after mechanical disturbance may be important for bird species that are dependent on vegetative structure and cover for foraging and nesting.

Lark Sparrow showed a strong positive response to fire sites. However, we found that Lark sparrow density had a strong positive relationship with tree cover. Lark Sparrows prefer open habitats with scattered trees, bare ground, and brushy edges (Martin and Parrish 2000, Lusk et al. 2003). We further found that increased shrub cover had a strong negative relationship with Lark Sparrow density. In our study area, fire sites had a significantly higher proportion of bare ground and a significantly lower proportion of shrub cover compared to mechanically disturbed sites and reference sites (Table 4.2). Perhaps Lark Sparrow habitat use is driven by a combination of forest edge characteristics (Martin and Parrish 2000) and habitat characteristics within disturbed sites.

Density of snags had the greatest positive effect on Hairy Woodpecker, which is not surprising as these are cavity nesting birds that prefer to nest in snags (Raphael and White 1984). In our study area, fire sites had four times the density of snags compared with mechanically disturbed sites and six times more snags than reference sites. Thus, for some species, fire appears to create conditions more favorable than mechanical disturbance. In contrast, however, density of snags had the greatest negative effect on Spotted Towhee habitat use. Spotted Towhee also showed strong negative response to fire sites; and in our study a large number of snags was a prominent characteristic of fire sites (Table 4.2). Fire sites also had a significantly lower proportion of shrub cover (11%) compared to reference sites (36%; Table 4.2), and increased shrub cover was the strongest indicator of Spotted Towhee habitat use.

Vegetation parameters were also important predictors of habitat use for several mammal species in our study. Grass cover had the strongest negative relationship with Chipmunk habitat use, which is unsurprising because Chipmunks are known for utilizing areas of dense shrub cover and forest edges, and are typically absent from grassy areas (Verts and Carraway 2001,

Braun et al. 2011). Grass cover had the strongest positive relationship with American Badger habitat use. American Badgers play an important role in the ecosystems they inhabit by creating small-scale soil disturbance and burrowing systems that are used by burrowing owls, rodents, and herptiles (Eldridge 2004); yet most studies assessing American Badger habitat use focus on prey item availability and few studies have been conducted to assess the influence vegetation characteristics have on American Badger habitat use (Apps et al. 2002). To our knowledge we are the first to estimate important vegetation characteristics influencing habitat use of American Badgers in our study region.

We found that shrub cover had the strongest negative influence on Coyote habitat use. Gese et al. (1988) demonstrated that Coyotes in southeastern Colorado preferred pinyon-juniper woodlands and only utilized shrubland-grassland habitats if pinyon-juniper woodlands were unavailable. Our study was conducted in large matrix of pinyon-juniper woodlands; therefore Coyotes may utilize shrubland habitats less than pinyon-juniper woodlands because a sufficient amount of preferred habitat is available.

Tree cover had the strongest negative influence on Bobcat habitat use, but forest age had the strongest positive influence. Bobcats prefer a wide variety of habitat characteristics but have been shown to avoid large open areas in summer months (Koehler and Hornocker 1991, Dickson and Beier 2002). Bobcats also showed a strong negative relationship with proportion of grass cover. In Arizona, Lawhead (1984) found that Bobcats used open shrubland habitats but avoided large extensive grasslands, and showed no strong preference for any single habitat feature as long as prey items and protective cover was available. Bobcats have large home ranges that

encompass a wide variety of habitat characteristics (Schonewald-Cox et al. 1991). Therefore, it is likely that habitat characteristics at varying scales influence habitat use of bobcats more than local-scale features.

Some measure of anthropogenic disturbance (e.g. well pad density) influenced the density of all birds in the study except Brewer's Sparrow and Mourning Dove, and habitat use for all mammal species responded strongly to at least one anthropogenic covariate. The size of the area disturbed mechanically or by fire had the greatest negative relationship with Blue-gray Gnatcatcher, Green-tailed Towhee, and White-breasted Nuthatch densities. These findings are consistent with previous studies showing that Blue-gray Gnatcatchers rely on forest cover for nesting structure and protective cover (Kershner and Ellison 2012, Smith et al. 2015), and that White-breasted Nuthatches rely on living trees to glean insects from bark crevices (Willson 1970).

Contrary to previous findings (Bayne et al. 2008, Francis et al. 2009), we found that well pad density had the strongest positive effect on the density of *Empidonax* flycatchers. Bayne et al. (2008) found that Least Flycatcher (*Empidonax minimus*) occurrence and Francis et al. (2009) found that Gray Flycatcher (*Empidonax wrightii*) abundance were negatively related to the proximity of the nearest natural gas well pad. However, both studies inferred that the negative relationships were due to chronic noise levels associated with the natural gas compressors located on the well pads rather than the physical footprint of the well pad. Well pad density had the greatest negative influence on Lark Sparrow densities. Lanen et al. (2011) demonstrated similar results in the Atlantic Rim Natural Gas Development Project Area near Rawlins, Wyoming, USA, where Lark Sparrow occurrences were significantly less in high development areas compared to low development areas.

Distance to nearest road had the strongest negative influence on Mountain Lion habitat use, but Coyote habitat use had a positive relationship with distance to nearest road. Neither of these results are surprising as Mountain Lions are vulnerable to road development (Trombulak and Frissell 2000, Gloyne and Clevenger 2001), and Coyotes have been shown to utilize areas near or on roads as travelling corridors (Gese et al. 1996). However, the relationships between habitat use, well pad density, and distance to nearest road differed for Coyote and Mountain Lion. Although Coyotes utilize roads (Gese et al. 1996), they alter activity patterns or completely avoid areas of high human activity (Tigas et al. 2002), and in Arizona and Utah, Mountain Lions utilized timber harvest areas, but avoided active logging roads (Van Dyke et al. 1986). Mountain Lions and Coyotes compete for resources, and Mountain Lions have been known to prey on Coyotes (Koehler and Hornocker 1991). Therefore, another possible explanation is that Coyotes and Mountain Lions may be spatially segregated across an anthropogenic gradient. We were not able to directly test this hypothesis, but this would be an interesting focus of future research.

Mountain Cottontail habitat use was most negatively influenced by distance to nearest road. This negative relationship between distance to nearest road and Mountain Cottontail habitat use is difficult to interpret. Lagomorphs are often the most common species collected in road mortality studies (Haugen 1944, Clevenger et al. 2003). Anecdotally, Mountain Cottontails were often seen alongside or near roads in our study area. Mountain Cottontails were the most detected species in our study and were detected at >75% of our sites. Unmeasured variables associated with each camera site may better explain variation in Mountain Cottontail habitat use.

The size of the disturbed area was negatively associated with American Badger habitat use. In a simulated study, Jager et al. (2006) found that the size of habitat loss had a negative relationship with American Badger fitness (i.e. survival and reproductive fitness). We did not

directly measure habitat loss per se, however, we did find a strong positive relationship between American badger habitat use and proportion of tree cover. One might argue that large-scale tree reduction – either by mechanical or natural disturbance – may constitute short-term habitat loss for American Badger.

With respect to small mammals, we were only able to detect a small subset of the small mammal community using wildlife cameras. Of those species detected, we found that Golden-mantled Ground Squirrel, Rock Squirrel, Chipmunk and Mountain Cottontail habitat use had a positive relationship to size of disturbed area. O'Meara et al. (1981) and Sedgwick and Ryder (1986) both found that Golden-mantled Ground Squirrels and Chipmunks preferred mechanically disturbed woodlands over intact woodlands; and Howard et al. (1987) and Kundaeli and Reynolds (1972) found that lagomorph abundance increased in mechanically disturbed woodlands. Further, Fontaine and Kennedy (2012) demonstrated that both Chipmunk and Golden-mantled Ground Squirrels had a positive response to fire severity. Although these studies did not directly measure the effect of disturbance size, they demonstrate an affinity to disturbed woodlands for these small mammal species.

PRIORITIES FOR RESEARCH AND MANAGEMENT

Our response metrics for evaluating the effects of natural and human-induced disturbance on animals were limited to bird and mammal species composition, bird species richness and density, and mammal habitat use. Future research should include other important state variables such as the density and abundance of mammals and fitness metrics (i.e., survival and fecundity) for bird populations. We also encourage studies that evaluate the effects of mechanical disturbance on other taxa that are of conservation concern (e.g., reptiles and amphibians) or play important roles in food webs and ecological processes (e.g. arthropods) (Debinski and Holt

2000). Further, we measured only the short-term effects of mechanical disturbance and wildfire on bird and mammal populations and communities, yet these effects may change over time if trajectories of ecological succession differ between mechanically disturbed and fire sites. For example, post-disturbance soil chemistry and soil structure may differ (Certini 2005, Busse et al. 2014), with consequences for plant community structure and composition and the animal species that rely on particular habitat characteristics (Keller et al. 2003, Fisher and Wilkinson 2005). Long-term monitoring of the effects of mechanical and natural disturbances on animal populations could make important contributions to the field of ecology as well as informing management practices.

Landscape-level characteristics may have a greater influence on bird and mammal communities than fine scale vegetation parameters (Schonewald-Cox et al. 1991, Saab 1999). Therefore, it may be important to incorporate landscape-scale attributes hypothesized to influence animal communities, such as habitat characteristics at differing spatial scales or landscape connectivity metrics (see Kindlmann and Burel 2008). To be cautious of model over parameterization we chose to use relatively simple habitat use and density models to examine main effects alone, without considering additive effects or interactions between habitat characteristics. It is likely that anthropogenic factors (e.g., well pad density or size of disturbed area) combined with vegetation characteristic have an influence on bird and mammal populations. Future research should test for these potential interactions; and tree removal efforts in human dominated landscapes should explicitly account for these interactions in future management plans.

Finally, the relationships we identified between animal response variables and specific habitat characteristics could be factored into management decisions. Modern tree removal

techniques (e.g., mastication; Wästerlund and Hassan 1995) can be highly selective in the vegetation that is removed (Wästerlund and Hassan 1995; Crow and van Riper 2010). For example, we found mean DBH/plot (a proxy for forest age) was a significant vegetative parameter influencing the densities of cavity nesting and bark gleaning birds and habitat use of mammal species that rely on trees for protective cover. Therefore, preserving large stands of older trees near or within treated areas might maintain habitat for these species. At minimum, selective tree removal allows the flexibility to more closely emulate natural disturbances. Therefore, accounting for important habitat characteristics during mechanical disturbance intended to emulate natural processes should be a priority for future forest management.

Our findings may or may not be relevant to other forms of deliberate habitat disturbance in other regions or ecosystems. The broad geographical distribution of pinyon-juniper woodlands supports a wide variety of soil types, precipitation, and elevation, which could affect rates and patterns of regeneration after disturbance (Schott and Pieper 1987; Romme et al. 2009; Tausch et al. 2009). Therefore, we urge caution in generalizing our results to other woodland ecosystems.

CONCLUSION

Habitat mitigation in the form of tree reduction – intended to emulate natural disturbances – had short-term consequences for birds and mammals in our study region. Thus, we suggest that future habitat mitigation involving large-scale mechanical disturbance should explicitly measure the intended and unintended consequences on birds, mammals, and other taxonomic groups. The hierarchical binomial-mixture models that we used to calculate bird density and mammal habitat use can be applied more broadly to other taxonomic groups. This information could be used to adaptively guide management decisions, taking into account short-term and long-term costs and benefits to species of conservation and/or economic value. Given

the scale of land cover change and well-documented impacts on natural ecosystems (Foley et al. 2005, Fuschs et al. 2015), it is imperative that we understand the full effects of habitat mitigation on natural communities.

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APPENDIX 1

SUPPORTING INFORMATION FOR CHAPTER ONE

PROTOCOL FOR IDENTIFYING AND SELECTING ARTICLES, AND CATEGORIZING FINDINGS FROM THESE STUDIES, IN OUR REVIEW OF THE EFFECTS OF GAME MANAGEMENT ON NON-TARGET SPECIES.

Identifying Articles

Following recommendations from the Collaboration for Environmental Evidence's Guidelines for Systematic Review and Evidence Synthesis in Environmental Management Version 4.2 (referred to hereafter as the Guidelines) we reviewed the literature on the effects of game management on non-targeted wildlife through a search of Thomas Reuters Web of Knowledge, Google Scholar, Agricola, and Google search engines. Searches were conducted using a combination of the terms biodiversity, non-game species, non-target species, species interactions, trophic cascades, indirect effects, impacts and the following terms:

- wildlife management
- game management
- habitat management
- habitat manipulation
- brush management
- deer management
- grouse management
- elk management
- dove management
- quail management
- waterfowl management
- moose management

- bear management
- big horn sheep management
- antelope management
- ungulate management
- small game management
- big game management
- pheasant management
- game bird management

Based on recommendations from the Guidelines, we refined our Google web search by using Google operators, Boolean logic, and truncation options. Since we were primarily using Google to identify relevant grey literature, we restricted our search to PDF, Microsoft Excel, and Microsoft Word documents to increase the likelihood that the search results contained useful documents and/or data. We observed that the relevance of the articles to the objectives of our literature search declined substantially after 250 hits. Therefore, we examined the first 250 hits returned by the Google search engine. As recommended by the Guidelines, we created our searches from the same location on the same day to maintain consistency in the page-ranking algorithm employed by Google.

Selecting Articles

We reviewed all published studies, including empirical research, meta-analyses, modeling articles, thesis, dissertations, conference proceedings, and grey literature, up until September 2015. One author (TG) read the abstract and introduction sections of all articles with the

keywords listed above. Studies that met the following criteria were included in our analysis and discussion:

- included land management practices specifically intended to improve habitat for game species;
- directly measured and reported the quantitative effects of these practices on non-target species;
- Focused on terrestrial game and non-game animals (plants and aquatic animals were excluded).

Articles were excluded if they addressed the effects of hunting on non-game species but did not explicitly focus on habitat management (e.g. ammunition poisoning). We also did not include studies that assessed the effects of predator control on non-target species, since this topic has been thoroughly reviewed elsewhere (see Reynolds and Tapper 1996). Review articles that only reported findings from previous studies were also excluded from our analysis.

Reynolds, J.C. and S.C. Tapper. 1996. Control of mammalian predators in game management and conservation. *Mammalian Review* 26:127-155. DOI: 10.1111/j.1365-2907.1996.tb00150.x

Extracting and categorizing findings from articles

The content of each article that met the criteria for our review was extracted and reported according to the mechanism used to alter habitat to benefit game species (Fig. 1.1; Table A1), the non-target taxa, and the direction of effect of the mechanism on non-target taxa (positive, negative, or no effect). The content of a single article could thus be assigned to multiple

mechanism and taxa categories if the authors studied multiple mechanisms or taxa, and/or found that the direction of the effect varied among mechanisms or taxa.

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Table A1. Full list of studies that measured the effects of game management on non-target species, and met all other selection criteria (see Appendix). We report the type of game management assessed, the direction of the effect on non-game species, and the taxa affected. Articles appear multiple times if they evaluated the effect of more than one type of game management activity, more than one non-target taxa, and/or found that the direction of the effect varied among management activity or taxa.

Type of Game Management	Direction of Effect	Affected non-target taxa	Source
Artificial Water Catchment	+	Birds - raptor species	O'Brien et al. 2006
Artificial Water Catchment	+	Mammals - predators, bats, and rodents	O'Brien et al. 2006
Artificial Water Catchment	<i>no effect</i>	Birds - passerine and raptor species	Lynn 2006
Increased Abundance of Game Animal	+	Birds - warblers and woodpigeons	Draycott et al. 2008
Increased Abundance of Game Animal	+	Mammals - fox, rabbit, and hares	Beja et al. 2009
Increased Abundance of Game Animal	-	Birds - shorebirds and riparian species	Matson 2000 ^g
Increased Abundance of Game Animal	-	Mammals - ungulates and beavers	Matson 2000 ^g
Increased Abundance of Game Animal	-	Mammals - Iberian ibex (<i>Capra pyrenaica</i>)	Acevedo et al. 2007
Increased Abundance of Game Animal	-	Mammals - European wildcat (<i>Felis silvestris</i>)	Lozano et al. 2007
Increased Abundance of Game Animal	-	Birds - ground feeding birds	Draycott et al. 2008
Increased Abundance of Game Animal	-	Birds - Kestrel (<i>Falco tinnunculus</i>)	Beja et al. 2009
Increased Abundance of Game Animal	-	Birds - woodland species	Newson et al. 2012
Increased Abundance of Game Animal	<i>no effect</i>	Birds - passerines, raptors and Corvids	Beja et al. 2009

Increased Abundance of Game Animal	<i>no effect</i>	Mammals - mesopredator	Beja et al. 2009
Increased Abundance of Game Animal	<i>no effect</i>	Arthropods - Adonis blue butterfly (<i>Polyommatus bellargus</i>)	Callegari et al. 2014
Mechanical Reduction of Woody Vegetation	+	Mammals - small mammals	O'Meara et al. 1981
Mechanical Reduction of Woody Vegetation	+	Birds - woodland and generalist species	Yahner 1984
Mechanical Reduction of Woody Vegetation	+	Birds - grassland sparrows	Gruver and Guthery 1986
Mechanical Reduction of Woody Vegetation	+	Birds - shrubland and granivorous species	Kozicky and Fulbright 1991
Mechanical Reduction of Woody Vegetation	+	Birds - woodland and generalist species	Yahner 1993
Mechanical Reduction of Woody Vegetation	+	Birds - open habitat species	Hanowski et al. 1999
Mechanical Reduction of Woody Vegetation	+	Mammals - rodent species	Sperry and Weatherhead 2010
Mechanical Reduction of Woody Vegetation	-	Birds - woodland species	O'Meara et al. 1981
Mechanical Reduction of Woody Vegetation	-	Birds - red-eyed vireo (<i>Vireo olivaceus</i>)	Yahner 1984
Mechanical Reduction of Woody Vegetation	-	Birds - Northern Mockingbird (<i>Mimus polyglottos</i>)	Gruver and Guthery 1986
Mechanical Reduction of Woody Vegetation	-	Birds - woodland species	Kozicky and Fulbright 1991
Mechanical Reduction of Woody Vegetation	-	Birds - red-eyed vireo and ovenbird	Yahner 1993

Mechanical Reduction of Woody Vegetation	-	Birds - shrubland species	Hanowski et al. 1999
Mechanical Reduction of Woody Vegetation	<i>no effect</i>	Birds - shrubland and grassland birds	Vega and Rappole 1994
Mechanical Reduction of Woody Vegetation	<i>no effect</i>	Birds - passerine species	Sperry and Weatherhead 2010
Planting Wildlife Crops	+	Birds - passerine species	Parish and Sotherton 2004
Planting Wildlife Crops	+	Birds - passerine species	Sage 2005
Planting Wildlife Crops	+	Birds - raptors and ground nesting species	Caro et al. 2015
Planting Wildlife Crops	<i>no effect</i>	Birds - passerine species	Stoate 2002
Planting Wildlife Crops	<i>no effect</i>	Birds - passerine species	Caro et al. 2015
Prescribed Fire	+	Birds - wader species	Tharme et al. 2000
Prescribed Fire	+	Birds - wader species	Irvine 2011
Prescribed Fire	+	Arthropods - disturbance tolerant species	Brown et al. 2014 ^g
Prescribed Fire	-	Birds - passerine species and crow	Tharme et al. 2000
Prescribed Fire	-	Birds - passerine species	Irvine 2012
Prescribed Fire	-	Arthropods - disturbance sensitive species	Brown et al. 2014
Prescribed Fire	<i>no effect</i>	Birds - shrubland species	Petersen and Best 1987
Prescribed Fire	<i>no effect</i>	Herptiles - lizards and	Radke 2008

^g Indicates grey literature

APPENDIX 2

SUPPORTING INFORMATION FOR CHAPTER TWO

Table A2.1. A complete list of bird species used to analyze community composition, and their respective maximum detection distance and where they generally display in the vegetation strata in the Piceance Basin of Northwest Colorado.

Species	Max Detection Distance	Display Strata
American Robin	>100 m and ≤150 m	Mid-level
Ash-throated Flycatcher	>150 m	Top
Yellow-rumped Warbler	>100 m and ≤150 m	Mid-level
Black-billed Magpie	>150 m	Top
Black-chinned Hummingbird	≤50 m	Mid-level
Bewick's Wren	>50 m and ≤100 m	Mid-level
Black-throated Gray Warbler	>150 m	Mid-level
Blue-gray gnatcatcher	>150 m	Mid-level
Brown-headed Cowbird	>150 m	Top
Brewer's Blackbird	>150 m	Top
Brewer's Sparrow	>150 m	Top
Broad-tailed Hummingbird	>150 m	Mid-level
Bullock's Oriole	>100 m and ≤150 m	Top
Bushtit	>50 m and ≤100 m	Mid-level
Cassin's Finch	>150 m	Top
Canyon Wren	>100 m and ≤150 m	Mid-level
Cedar Waxwing	>100 m and ≤150 m	Top
Chipping Sparrow	>150 m	Mid-level
Clark's Nutcracker	>150 m	Top
Common Nighthawk	>150 m	Aerial
Common Raven	>150 m	Top
Dark-eyed Junco	>50 m and ≤100 m	Mid-level
<i>Empidonax</i> Flycatcher	>150 m	Top
Evening Grosbeak	>150 m	Top
Green-tailed Towhee	>150 m	Top
Hairy Woodpecker	>150 m	Mid-level
Hermit Thrush	>150 m	Ground
House Finch	>50 m and ≤100 m	Top
House Wren	>150 m	Mid-level
Juniper Titmouse	>150 m	Mid-level
Lark Sparrow	>50 m and ≤100 m	Ground
Lazuli Bunting	>100 m and ≤150 m	Mid-level
Lesser Goldfinch	>100 m and ≤150 m	Mid-level
Lincoln Sparrow	>50 m and ≤100 m	Mid-level
MacGillivray's Warbler	>150 m	Mid-level
Mountain Bluebird	>150 m	Top
Mountain Chickadee	>150 m	Mid-level

Mourning Dove	>150 m	Ground
Northern Flicker	>150 m	Mid-level
Northern Rough-winged Swallow	≤50 m	Aerial
Orange-crowned Warbler	>50 m and ≤100 m	Top
Pinyon Jay	>150 m	Top
Plumbeous Vireo	>150 m	Mid-level
Red-breasted Nuthatch	>150 m	Mid-level
Ruby-crowned Kinglet	≤50 m	Mid-level
Red Crossbill	≤50 m	Top
Rock Wren	>150 m	Ground
Red-winged Blackbird	≤50 m	Top
Say's Phoebe	>50 m and ≤100 m	Top
Savannah Sparrow	≤50 m	Ground
Sage Thrasher	>150 m	Top
Spotted Towhee	>150 m	Mid-level
Vesper Sparrow	≤50 m	Ground
Violet-green Swallow	>150 m	Aerial
Virginia's Warbler	>50 m and ≤100 m	Top
White-crowned Sparrow	≤50 m	Mid-level
White-breasted Nuthatch	>150 m	Mid-level
Western Bluebird	>50 m and ≤100 m	Top
Western Kingbird	≤50 m	Top
Western Meadowlark	>150 m	Ground
Western Scrubjay	>150 m	Top
Western Tanager	>100 m and ≤150 m	Top
Western Wood Pewee	≤50 m	Top
Wild Turkey	>50 m and ≤100 m	Ground
White-throated Swift	>150 m	Aerial

Table A2.2. The median value of the posterior distribution of site abundance (N_i) and associated 95% credible intervals for each bird species at chained (C), reference (R), sites in the Piceance Basin of Northwest Colorado.

Species	Site	2013			2014		
		N_i	LCI	UCI	N_i	LCI	UCI
American Robin	C1	0	0	6	0	0	6
American Robin	C2	0	0	6	0	0	6
American Robin	C3	0	0	6	0	0	6
American Robin	C4	0	0	6	0	0	6
American Robin	C5	0	0	6	0	0	6
American Robin	C6	0	0	6	0	0	6
American Robin	C7	0	0	6	0	0	6
American Robin	C8	1	0	13	1	0	13
American Robin	C9	1	0	13	1	0	13
American Robin	C10	1	0	13	1	0	13
American Robin	C11	1	0	13	2	1	14
American Robin	C12	1	0	13	1	0	13
American Robin	C13	1	0	13	1	0	13
American Robin	C14	0	0	10	0	0	10
American Robin	C15	0	0	10	0	0	10
American Robin	C16	1	0	29	1	0	29
American Robin	C17	1	0	29	2	1	30
American Robin	C18	0	0	7	0	0	7
American Robin	C19	0	0	7	0	0	7
American Robin	C20	0	0	7	0	0	7
American Robin	C21	0	0	7	0	0	7
American Robin	C22	1	0	29	1	0	29
American Robin	C23	1	0	29	2	1	30
American Robin	C24	0	0	7	0	0	7
American Robin	C25	0	0	7	0	0	7
American Robin	R1	2	0	23	2	0	23
American Robin	R2	2	0	23	2	0	23
American Robin	R3	2	0	23	3	1	24
American Robin	R4	2	0	23	2	0	23
American Robin	R5	3	0	46	3	0	46
American Robin	R6	3	0	46	3	0	46
American Robin	R7	3	0	46	3	0	46
American Robin	R8	3	0	46	5	2	48
American Robin	R9	3	0	46	3	0	46
American Robin	R10	3	0	46	3	0	46
American Robin	R11	3	0	46	3	0	46
American Robin	R12	4	1	47	3	0	46

American Robin	R13	3	0	46	3	0	46
American Robin	R14	3	0	46	3	0	46
American Robin	R15	3	0	46	3	0	46
American Robin	R16	3	0	46	6	2	49
American Robin	R17	3	0	46	5	1	48
American Robin	R18	3	0	46	3	0	46
American Robin	R19	4	1	47	3	0	46
American Robin	R20	2	0	23	3	1	24
American Robin	R21	2	0	23	2	0	23
American Robin	R22	2	0	23	2	0	23
American Robin	R23	2	0	23	2	0	23
American Robin	R24	2	0	23	3	1	24
American Robin	R25	2	0	23	2	0	23
Ash-throated Flycatcher	C1	1	0	21	1	0	21
Ash-throated Flycatcher	C2	1	0	21	1	0	21
Ash-throated Flycatcher	C3	1	0	21	1	0	21
Ash-throated Flycatcher	C4	1	0	21	1	0	21
Ash-throated Flycatcher	C5	1	0	21	2	1	22
Ash-throated Flycatcher	C6	1	0	21	1	0	21
Ash-throated Flycatcher	C7	1	0	21	1	0	22
Ash-throated Flycatcher	C8	1	0	15	1	0	15
Ash-throated Flycatcher	C9	1	0	15	1	0	15
Ash-throated Flycatcher	C10	1	0	15	1	0	15
Ash-throated Flycatcher	C11	1	0	15	1	0	15
Ash-throated Flycatcher	C12	1	0	15	1	0	15
Ash-throated Flycatcher	C13	1	0	15	1	0	15
Ash-throated Flycatcher	C14	5	0	62	5	0	62
Ash-throated Flycatcher	C15	5	0	62	7	2	64
Ash-throated Flycatcher	C16	8	0	95	9	1	95
Ash-throated Flycatcher	C17	8	0	94	9	1	96
Ash-throated Flycatcher	C18	7	0	92	7	0	92
Ash-throated Flycatcher	C19	7	0	92	8	1	93
Ash-throated Flycatcher	C20	7	0	92	9	1	93
Ash-throated Flycatcher	C21	9	2	94	8	1	93
Ash-throated Flycatcher	C22	8	0	94	9	1	95
Ash-throated Flycatcher	C23	10	2	97	8	0	94
Ash-throated Flycatcher	C24	10	2	95	7	0	92
Ash-throated Flycatcher	C25	7	0	92	7	0	92
Ash-throated Flycatcher	R1	3	0	35	3	0	35
Ash-throated Flycatcher	R2	3	0	35	3	0	35
Ash-throated Flycatcher	R3	3	0	35	3	0	35
Ash-throated Flycatcher	R4	5	1	37	3	0	35
Ash-throated Flycatcher	R5	5	0	52	6	1	53

Ash-throated Flycatcher	R6	5	0	52	5	0	52
Ash-throated Flycatcher	R7	6	1	53	5	0	52
Ash-throated Flycatcher	R8	5	0	52	5	0	52
Ash-throated Flycatcher	R9	5	0	51	5	0	52
Ash-throated Flycatcher	R10	5	0	52	5	0	52
Ash-throated Flycatcher	R11	5	0	52	7	1	54
Ash-throated Flycatcher	R12	5	0	52	5	0	52
Ash-throated Flycatcher	R13	7	2	54	6	1	53
Ash-throated Flycatcher	R14	5	0	52	5	0	52
Ash-throated Flycatcher	R15	7	2	54	5	0	52
Ash-throated Flycatcher	R16	5	0	52	5	0	52
Ash-throated Flycatcher	R17	5	0	52	5	0	52
Ash-throated Flycatcher	R18	5	0	52	5	0	51
Ash-throated Flycatcher	R19	6	1	53	5	0	52
Ash-throated Flycatcher	R20	3	0	35	3	0	35
Ash-throated Flycatcher	R21	3	0	35	4	1	36
Ash-throated Flycatcher	R22	4	1	36	3	0	35
Ash-throated Flycatcher	R23	3	0	35	3	0	35
Ash-throated Flycatcher	R24	3	0	35	3	0	35
Ash-throated Flycatcher	R25	3	0	35	4	1	36
Black-throated Gray Warbler	C1	1	0	7	1	0	8
Black-throated Gray Warbler	C2	2	1	8	1	0	7
Black-throated Gray Warbler	C3	2	1	8	1	0	8
Black-throated Gray Warbler	C4	1	0	7	1	0	8
Black-throated Gray Warbler	C5	2	1	9	1	0	7
Black-throated Gray Warbler	C6	2	1	9	3	1	9
Black-throated Gray Warbler	C7	1	0	7	1	0	7
Black-throated Gray Warbler	C8	2	1	8	2	1	8
Black-throated Gray Warbler	C9	1	0	7	1	0	7
Black-throated Gray Warbler	C10	1	0	7	1	0	7
Black-throated Gray Warbler	C11	1	0	7	2	1	8
Black-throated Gray Warbler	C12	1	0	7	1	0	7
Black-throated Gray Warbler	C13	2	1	8	1	0	7
Black-throated Gray Warbler	C14	1	0	6	1	0	6
Black-throated Gray Warbler	C15	1	0	6	1	0	6
Black-throated Gray Warbler	C16	2	0	14	2	0	14
Black-throated Gray Warbler	C17	7	4	20	2	0	14
Black-throated Gray Warbler	C18	4	2	14	3	1	12
Black-throated Gray Warbler	C19	3	1	12	2	0	11
Black-throated Gray Warbler	C20	2	0	11	2	0	11
Black-throated Gray Warbler	C21	3	1	12	3	1	12
Black-throated Gray Warbler	C22	4	2	16	2	0	14
Black-throated Gray Warbler	C23	2	0	14	4	2	16

Black-throated Gray Warbler	C24	2	0	11	4	1	13
Black-throated Gray Warbler	C25	2	0	11	2	0	11
Black-throated Gray Warbler	R1	5	1	24	5	1	24
Black-throated Gray Warbler	R2	5	1	24	8	3	27
Black-throated Gray Warbler	R3	7	3	26	6	2	25
Black-throated Gray Warbler	R4	7	3	26	4	0	23
Black-throated Gray Warbler	R5	3	0	17	3	0	17
Black-throated Gray Warbler	R6	4	1	18	4	1	18
Black-throated Gray Warbler	R7	4	1	18	3	0	17
Black-throated Gray Warbler	R8	6	2	20	5	2	19
Black-throated Gray Warbler	R9	4	1	18	3	0	17
Black-throated Gray Warbler	R10	3	0	17	3	0	17
Black-throated Gray Warbler	R11	5	1	19	4	1	18
Black-throated Gray Warbler	R12	7	3	20	3	0	17
Black-throated Gray Warbler	R13	4	1	18	3	0	17
Black-throated Gray Warbler	R14	3	0	17	3	0	17
Black-throated Gray Warbler	R15	3	0	17	3	0	17
Black-throated Gray Warbler	R16	6	2	20	3	0	17
Black-throated Gray Warbler	R17	6	2	20	3	0	17
Black-throated Gray Warbler	R18	6	2	20	4	1	18
Black-throated Gray Warbler	R19	6	3	20	7	3	21
Black-throated Gray Warbler	R20	5	1	24	8	3	27
Black-throated Gray Warbler	R21	9	4	28	4	0	23
Black-throated Gray Warbler	R22	4	0	23	4	0	23
Black-throated Gray Warbler	R23	8	3	26	4	0	23
Black-throated Gray Warbler	R24	5	1	24	4	0	23
Black-throated Gray Warbler	R25	6	2	25	6	2	25
Brewer's Sparrow	C1	10	3	51	9	2	50
Brewer's Sparrow	C2	7	1	48	7	1	48
Brewer's Sparrow	C3	10	3	51	12	5	53
Brewer's Sparrow	C4	7	1	48	7	1	48
Brewer's Sparrow	C5	11	4	52	9	2	50
Brewer's Sparrow	C6	11	5	52	8	2	49
Brewer's Sparrow	C7	8	2	49	7	1	48
Brewer's Sparrow	C8	14	3	87	16	5	89
Brewer's Sparrow	C9	17	5	89	17	6	90
Brewer's Sparrow	C10	16	5	89	15	4	88
Brewer's Sparrow	C11	17	6	90	15	4	88
Brewer's Sparrow	C12	20	8	93	16	5	89
Brewer's Sparrow	C13	19	7	92	14	3	87
Brewer's Sparrow	C14	37	14	187	38	16	188
Brewer's Sparrow	C15	31	9	182	32	10	182
Brewer's Sparrow	C16	26	9	137	27	10	137

Brewer's Sparrow	C17	20	5	130	24	8	135
Brewer's Sparrow	C18	13	3	79	15	5	82
Brewer's Sparrow	C19	13	3	79	13	3	79
Brewer's Sparrow	C20	15	5	81	22	11	90
Brewer's Sparrow	C21	13	3	79	15	5	81
Brewer's Sparrow	C22	25	9	135	29	12	140
Brewer's Sparrow	C23	20	5	131	22	6	132
Brewer's Sparrow	C24	14	4	81	13	3	79
Brewer's Sparrow	C25	15	5	81	15	5	82
Brewer's Sparrow	R1	0	0	4	0	0	4
Brewer's Sparrow	R2	1	0	4	1	0	4
Brewer's Sparrow	R3	1	0	4	0	0	4
Brewer's Sparrow	R4	0	0	4	0	0	4
Brewer's Sparrow	R5	1	0	8	3	1	9
Brewer's Sparrow	R6	1	0	8	1	0	8
Brewer's Sparrow	R7	1	0	8	1	0	8
Brewer's Sparrow	R8	2	0	8	1	0	8
Brewer's Sparrow	R9	1	0	8	1	0	8
Brewer's Sparrow	R10	1	0	8	1	0	8
Brewer's Sparrow	R11	1	0	8	1	0	8
Brewer's Sparrow	R12	1	0	8	1	0	8
Brewer's Sparrow	R13	1	0	8	2	1	9
Brewer's Sparrow	R14	2	1	9	5	3	12
Brewer's Sparrow	R15	3	2	10	2	1	9
Brewer's Sparrow	R16	2	1	9	1	0	8
Brewer's Sparrow	R17	1	0	8	1	0	8
Brewer's Sparrow	R18	1	0	8	1	0	8
Brewer's Sparrow	R19	3	1	9	1	0	8
Brewer's Sparrow	R20	1	0	4	0	0	4
Brewer's Sparrow	R21	1	0	4	0	0	4
Brewer's Sparrow	R22	0	0	4	0	0	4
Brewer's Sparrow	R23	0	0	4	2	2	6
Brewer's Sparrow	R24	0	0	4	0	0	4
Brewer's Sparrow	R25	0	0	4	0	0	4
Broad-tailed Hummingbird	C1	3	1	23	3	1	23
Broad-tailed Hummingbird	C2	2	0	22	2	0	22
Broad-tailed Hummingbird	C3	2	0	22	2	0	22
Broad-tailed Hummingbird	C4	2	0	22	2	0	22
Broad-tailed Hummingbird	C5	3	1	23	2	0	22
Broad-tailed Hummingbird	C6	2	0	22	2	0	22
Broad-tailed Hummingbird	C7	2	0	22	2	0	22
Broad-tailed Hummingbird	C8	2	0	26	2	0	26
Broad-tailed Hummingbird	C9	2	0	26	3	1	27

Broad-tailed Hummingbird	C10	2	0	26	2	0	26
Broad-tailed Hummingbird	C11	2	0	26	3	1	28
Broad-tailed Hummingbird	C12	2	0	26	3	1	28
Broad-tailed Hummingbird	C13	2	0	27	2	0	27
Broad-tailed Hummingbird	C14	1	0	17	1	0	17
Broad-tailed Hummingbird	C15	1	0	17	1	0	17
Broad-tailed Hummingbird	C16	1	0	17	1	0	17
Broad-tailed Hummingbird	C17	1	0	17	1	0	17
Broad-tailed Hummingbird	C18	1	0	16	1	0	16
Broad-tailed Hummingbird	C19	1	0	16	1	0	16
Broad-tailed Hummingbird	C20	1	0	16	1	0	16
Broad-tailed Hummingbird	C21	1	0	16	1	0	16
Broad-tailed Hummingbird	C22	1	0	17	1	0	17
Broad-tailed Hummingbird	C23	1	0	17	1	0	17
Broad-tailed Hummingbird	C24	1	0	16	2	1	17
Broad-tailed Hummingbird	C25	1	0	16	1	0	16
Broad-tailed Hummingbird	R1	2	0	19	2	0	19
Broad-tailed Hummingbird	R2	2	0	19	2	0	19
Broad-tailed Hummingbird	R3	3	1	20	2	0	19
Broad-tailed Hummingbird	R4	2	0	19	2	0	19
Broad-tailed Hummingbird	R5	3	1	21	2	0	20
Broad-tailed Hummingbird	R6	2	0	20	2	0	20
Broad-tailed Hummingbird	R7	2	0	20	2	0	20
Broad-tailed Hummingbird	R8	2	0	20	2	0	20
Broad-tailed Hummingbird	R9	2	0	20	2	0	20
Broad-tailed Hummingbird	R10	2	0	20	2	0	20
Broad-tailed Hummingbird	R11	4	1	22	2	0	20
Broad-tailed Hummingbird	R12	2	0	20	2	0	20
Broad-tailed Hummingbird	R13	3	1	21	2	0	20
Broad-tailed Hummingbird	R14	3	1	21	2	0	20
Broad-tailed Hummingbird	R15	2	0	20	2	0	20
Broad-tailed Hummingbird	R16	2	0	20	2	0	20
Broad-tailed Hummingbird	R17	2	0	20	2	0	20
Broad-tailed Hummingbird	R18	2	0	20	2	0	20
Broad-tailed Hummingbird	R19	2	0	20	2	0	20
Broad-tailed Hummingbird	R20	2	0	19	2	0	19
Broad-tailed Hummingbird	R21	2	0	19	3	1	20
Broad-tailed Hummingbird	R22	2	0	19	2	0	19
Broad-tailed Hummingbird	R23	2	0	19	2	0	19
Broad-tailed Hummingbird	R24	2	0	19	2	0	19
Broad-tailed Hummingbird	R25	2	0	19	2	0	19
Cassin's Finch	C1	2	0	24	2	0	24
Cassin's Finch	C2	2	0	24	3	1	25

Cassin's Finch	C3	2	0	24	2	0	24
Cassin's Finch	C4	2	0	24	2	0	24
Cassin's Finch	C5	2	0	24	3	1	25
Cassin's Finch	C6	2	0	24	2	0	24
Cassin's Finch	C7	2	0	24	2	0	24
Cassin's Finch	C8	2	0	27	2	0	27
Cassin's Finch	C9	2	0	27	2	0	27
Cassin's Finch	C10	2	0	27	2	0	27
Cassin's Finch	C11	2	0	27	4	2	29
Cassin's Finch	C12	2	0	27	2	0	27
Cassin's Finch	C13	2	0	27	2	0	27
Cassin's Finch	C14	1	0	20	1	0	20
Cassin's Finch	C15	1	0	20	1	0	20
Cassin's Finch	C16	1	0	18	1	0	18
Cassin's Finch	C17	1	0	18	1	0	18
Cassin's Finch	C18	4	2	35	2	0	33
Cassin's Finch	C19	2	0	33	2	0	33
Cassin's Finch	C20	2	0	33	2	0	33
Cassin's Finch	C21	2	0	33	2	0	33
Cassin's Finch	C22	1	0	18	1	0	18
Cassin's Finch	C23	1	0	18	1	0	18
Cassin's Finch	C24	2	0	33	3	1	34
Cassin's Finch	C25	2	0	33	2	0	33
Cassin's Finch	R1	9	1	101	10	2	102
Cassin's Finch	R2	9	1	102	10	2	103
Cassin's Finch	R3	9	1	102	9	1	102
Cassin's Finch	R4	9	1	102	11	3	103
Cassin's Finch	R5	7	1	65	6	0	64
Cassin's Finch	R6	6	0	64	7	1	65
Cassin's Finch	R7	6	0	64	6	0	64
Cassin's Finch	R8	7	1	65	6	0	64
Cassin's Finch	R9	7	1	65	7	1	65
Cassin's Finch	R10	6	0	64	6	0	64
Cassin's Finch	R11	6	0	64	6	0	64
Cassin's Finch	R12	8	2	66	6	0	64
Cassin's Finch	R13	8	2	67	6	0	64
Cassin's Finch	R14	6	0	64	6	0	64
Cassin's Finch	R15	6	0	64	6	0	64
Cassin's Finch	R16	6	0	64	7	1	65
Cassin's Finch	R17	6	0	64	7	1	65
Cassin's Finch	R18	6	0	64	6	0	64
Cassin's Finch	R19	6	0	64	6	0	64
Cassin's Finch	R20	9	1	101	10	2	103

Cassin's Finch	R21	9	1	101	10	2	103
Cassin's Finch	R22	9	1	102	9	1	101
Cassin's Finch	R23	9	1	101	9	1	101
Cassin's Finch	R24	10	2	102	11	2	103
Cassin's Finch	R25	12	4	104	11	3	103
Chipping Sparrow	C1	15	2	109	14	1	108
Chipping Sparrow	C2	16	3	110	14	1	108
Chipping Sparrow	C3	15	2	109	14	1	108
Chipping Sparrow	C4	17	4	111	16	3	110
Chipping Sparrow	C5	15	2	109	14	1	108
Chipping Sparrow	C6	14	1	108	15	2	109
Chipping Sparrow	C7	14	1	107	14	1	108
Chipping Sparrow	C8	10	0	84	10	0	84
Chipping Sparrow	C9	11	2	85	12	3	86
Chipping Sparrow	C10	10	0	84	10	0	84
Chipping Sparrow	C11	10	1	84	10	1	84
Chipping Sparrow	C12	10	0	84	11	1	85
Chipping Sparrow	C13	10	1	84	10	1	84
Chipping Sparrow	C14	13	2	102	12	1	101
Chipping Sparrow	C15	12	1	101	13	2	102
Chipping Sparrow	C16	30	5	202	29	4	201
Chipping Sparrow	C17	30	6	203	29	4	201
Chipping Sparrow	C18	21	2	148	23	4	151
Chipping Sparrow	C19	22	3	150	22	3	150
Chipping Sparrow	C20	25	6	153	21	2	149
Chipping Sparrow	C21	21	2	148	24	5	152
Chipping Sparrow	C22	30	6	203	29	4	201
Chipping Sparrow	C23	30	6	203	29	4	202
Chipping Sparrow	C24	23	4	151	21	2	149
Chipping Sparrow	C25	23	4	151	21	2	149
Chipping Sparrow	R1	19	2	138	19	2	138
Chipping Sparrow	R2	19	2	138	21	4	140
Chipping Sparrow	R3	19	2	138	19	2	138
Chipping Sparrow	R4	19	2	138	21	4	140
Chipping Sparrow	R5	13	1	93	13	1	93
Chipping Sparrow	R6	14	2	93	14	2	94
Chipping Sparrow	R7	15	3	95	15	3	95
Chipping Sparrow	R8	13	1	93	15	3	95
Chipping Sparrow	R9	13	1	92	13	1	93
Chipping Sparrow	R10	13	1	93	13	1	93
Chipping Sparrow	R11	14	2	94	13	1	92
Chipping Sparrow	R12	13	1	93	14	2	94
Chipping Sparrow	R13	16	4	96	13	1	93

Chipping Sparrow	R14	13	1	93	13	1	93
Chipping Sparrow	R15	15	3	95	14	2	94
Chipping Sparrow	R16	13	1	93	13	1	93
Chipping Sparrow	R17	14	2	94	14	2	94
Chipping Sparrow	R18	13	1	93	13	1	93
Chipping Sparrow	R19	14	2	94	13	1	93
Chipping Sparrow	R20	19	2	138	19	2	138
Chipping Sparrow	R21	22	5	141	20	3	139
Chipping Sparrow	R22	20	3	139	19	2	138
Chipping Sparrow	R23	22	4	140	21	4	140
Chipping Sparrow	R24	20	3	139	21	4	140
Chipping Sparrow	R25	21	4	139	22	5	141
Dark-eyed Junco	C1	8	1	147	7	0	146
Dark-eyed Junco	C2	7	0	146	7	0	146
Dark-eyed Junco	C3	11	4	149	7	0	145
Dark-eyed Junco	C4	7	0	146.025	7	0	146
Dark-eyed Junco	C5	7	0	145	7	0	145
Dark-eyed Junco	C6	7	0	146	7	0	145
Dark-eyed Junco	C7	7	0	146	7	0	146
Dark-eyed Junco	C8	0	0	3	0	0	3
Dark-eyed Junco	C9	0	0	3	0	0	3
Dark-eyed Junco	C10	0	0	3	0	0	3
Dark-eyed Junco	C11	0	0	3	0	0	3
Dark-eyed Junco	C12	0	0	3	0	0	3
Dark-eyed Junco	C13	0	0	3	0	0	3
Dark-eyed Junco	C14	0	0	11	0	0	11
Dark-eyed Junco	C15	0	0	11	0	0	11
Dark-eyed Junco	C16	0	0	20	0	0	20
Dark-eyed Junco	C17	0	0	20	0	0	20
Dark-eyed Junco	C18	0	0	5	0	0	5
Dark-eyed Junco	C19	0	0	5	0	0	5
Dark-eyed Junco	C20	0	0	5	0	0	5
Dark-eyed Junco	C21	0	0	5	0	0	5
Dark-eyed Junco	C22	0	0	20	0	0	20
Dark-eyed Junco	C23	0	0	20	0	0	20
Dark-eyed Junco	C24	0	0	5	0	0	5
Dark-eyed Junco	C25	0	0	5	0	0	5
Dark-eyed Junco	R1	0	0	3	0	0	3
Dark-eyed Junco	R2	0	0	3	0	0	3
Dark-eyed Junco	R3	0	0	3	0	0	3
Dark-eyed Junco	R4	0	0	3	0	0	3
Dark-eyed Junco	R5	1	0	35	1	0	36
Dark-eyed Junco	R6	1	0	35	1	0	36

Dark-eyed Junco	R7	1	0	35	1	0	36
Dark-eyed Junco	R8	1	0	35	1	0	35
Dark-eyed Junco	R9	1	0	35	1	0	35
Dark-eyed Junco	R10	1	0	35	1	0	35
Dark-eyed Junco	R11	2	1	37	1	0	35
Dark-eyed Junco	R12	1	0	35	1	0	35
Dark-eyed Junco	R13	1	0	36	1	0	36
Dark-eyed Junco	R14	1	0	35	1	0	35
Dark-eyed Junco	R15	1	0	35	1	0	36
Dark-eyed Junco	R16	1	0	36	1	0	35
Dark-eyed Junco	R17	1	0	35	1	0	35
Dark-eyed Junco	R18	1	0	35	2	1	37
Dark-eyed Junco	R19	1	0	36	1	0	35.025
Dark-eyed Junco	R20	0	0	3	0	0	3
Dark-eyed Junco	R21	0	0	3	0	0	3
Dark-eyed Junco	R22	0	0	3	0	0	3
Dark-eyed Junco	R23	0	0	3	0	0	3
Dark-eyed Junco	R24	0	0	3	0	0	3
Dark-eyed Junco	R25	0	0	3	0	0	3
<i>Empidonax</i> flycatcher	C1	0	0	4	2	1	6
<i>Empidonax</i> flycatcher	C2	1	1	5	1	1	5
<i>Empidonax</i> flycatcher	C3	0	0	4	0	0	4
<i>Empidonax</i> flycatcher	C4	0	0	4	1	1	5
<i>Empidonax</i> flycatcher	C5	0	0	4	0	0	4
<i>Empidonax</i> flycatcher	C6	0	0	4	1	1	5
<i>Empidonax</i> flycatcher	C7	0	0	4	0	0	4
<i>Empidonax</i> flycatcher	C8	0	0	2	0	0	2
<i>Empidonax</i> flycatcher	C9	0	0	3	0	0	2
<i>Empidonax</i> flycatcher	C10	1	1	3	0	0	2
<i>Empidonax</i> flycatcher	C11	0	0	3	0	0	3
<i>Empidonax</i> flycatcher	C12	0	0	2	0	0	2
<i>Empidonax</i> flycatcher	C13	0	0	3	0	0	2
<i>Empidonax</i> flycatcher	C14	1	0	7	3	2	9
<i>Empidonax</i> flycatcher	C15	1	0	7	3	1	9
<i>Empidonax</i> flycatcher	C16	4	1	15	5	2	16
<i>Empidonax</i> flycatcher	C17	2	0	13	5	2	15
<i>Empidonax</i> flycatcher	C18	5	2	15	4	1	14
<i>Empidonax</i> flycatcher	C19	5	2	16	4	1	15
<i>Empidonax</i> flycatcher	C20	5	2	16	3	0	13
<i>Empidonax</i> flycatcher	C21	5	2	16	5	2	15
<i>Empidonax</i> flycatcher	C22	4	1	15	5	2	17
<i>Empidonax</i> flycatcher	C23	3	1	14	6	2	17
<i>Empidonax</i> flycatcher	C24	7	3	19	6	2	17

<i>Empidonax flycatcher</i>	C25	3	0	13	4	1	14
<i>Empidonax flycatcher</i>	R1	2	0	9	2	0	9
<i>Empidonax flycatcher</i>	R2	3	1	10	3	1	10
<i>Empidonax flycatcher</i>	R3	2	0	9	2	0	9
<i>Empidonax flycatcher</i>	R4	4	2	12	5	2	13
<i>Empidonax flycatcher</i>	R5	3	1	9	4	1	11
<i>Empidonax flycatcher</i>	R6	2	0	8	5	2	12
<i>Empidonax flycatcher</i>	R7	3	1	9	3	1	9
<i>Empidonax flycatcher</i>	R8	2	0	8	4	2	11
<i>Empidonax flycatcher</i>	R9	4	2	11	2	0	8
<i>Empidonax flycatcher</i>	R10	2	0	9	3	1	10
<i>Empidonax flycatcher</i>	R11	2	0	8	4	2	12
<i>Empidonax flycatcher</i>	R12	3	1	9	2	0	8
<i>Empidonax flycatcher</i>	R13	2	0	8	3	1	10
<i>Empidonax flycatcher</i>	R14	2	0	8	4	2	11
<i>Empidonax flycatcher</i>	R15	2	0	8	3	1	9
<i>Empidonax flycatcher</i>	R16	4	1	11	3	1	9
<i>Empidonax flycatcher</i>	R17	2	0	8	3	1	9
<i>Empidonax flycatcher</i>	R18	2	0	8	2	0	8
<i>Empidonax flycatcher</i>	R19	4	2	11	4	1	11
<i>Empidonax flycatcher</i>	R20	2	0	9	2	0	9
<i>Empidonax flycatcher</i>	R21	3	1	10	3	1	10
<i>Empidonax flycatcher</i>	R22	3	1	10	5	2	13
<i>Empidonax flycatcher</i>	R23	3	1	11	4	1	12
<i>Empidonax flycatcher</i>	R24	3	1	10	3	1	10
<i>Empidonax flycatcher</i>	R25	5	2	13	3	1	11
Green-tailed Towhee	C1	7	4	15	6	3	13
Green-tailed Towhee	C2	4	2	11	3	1	9
Green-tailed Towhee	C3	2	0	8	4	2	11
Green-tailed Towhee	C4	4	2	11	2	0	8
Green-tailed Towhee	C5	3	1	9	2	0	8
Green-tailed Towhee	C6	2	0	8	2	0	8
Green-tailed Towhee	C7	4	2	10	2	0	8
Green-tailed Towhee	C8	4	1	12	4	1	12
Green-tailed Towhee	C9	4	1	12	5	2	13
Green-tailed Towhee	C10	5	2	13	3	1	11
Green-tailed Towhee	C11	6	3	14	4	1	12
Green-tailed Towhee	C12	6	3	14	3	1	11
Green-tailed Towhee	C13	2	0	10	4	1	12
Green-tailed Towhee	C14	4	1	13	7	3	16
Green-tailed Towhee	C15	4	1	13	6	3	15
Green-tailed Towhee	C16	2	0	8	4	2	10
Green-tailed Towhee	C17	2	0	8	4	1	10

Green-tailed Towhee	C18	4	1	10	3	1	9
Green-tailed Towhee	C19	2	0	8	4	2	10
Green-tailed Towhee	C20	4	1	10	3	1	9
Green-tailed Towhee	C21	5	2	12	4	2	11
Green-tailed Towhee	C22	2	0	8	6	4	13
Green-tailed Towhee	C23	2	0	8	3	1	9
Green-tailed Towhee	C24	2	0	8	5	2	12
Green-tailed Towhee	C25	3	1	9	3	1	9
Green-tailed Towhee	R1	1	0	6	3	1	9
Green-tailed Towhee	R2	1	0	6	3	1	9
Green-tailed Towhee	R3	2	1	7	4	2	9
Green-tailed Towhee	R4	1	0	6	1	0	6
Green-tailed Towhee	R5	1	0	4	3	2	7
Green-tailed Towhee	R6	1	0	4	1	0	4
Green-tailed Towhee	R7	1	0	4	1	0	4
Green-tailed Towhee	R8	1	0	4	2	1	5
Green-tailed Towhee	R9	1	0	4	1	0	4
Green-tailed Towhee	R10	2	1	6	2	1	5
Green-tailed Towhee	R11	2	1	5	2	1	5
Green-tailed Towhee	R12	1	0	4	1	0	4
Green-tailed Towhee	R13	1	0	4	2	1	5
Green-tailed Towhee	R14	2	1	5	4	2	9
Green-tailed Towhee	R15	2	1	5	1	0	4
Green-tailed Towhee	R16	2	1	5	2	1	6
Green-tailed Towhee	R17	1	0	4	3	1	7
Green-tailed Towhee	R18	1	0	4	1	0	4
Green-tailed Towhee	R19	1	0	4	1	0	4
Green-tailed Towhee	R20	4	2	9	1	0	6
Green-tailed Towhee	R21	1	0	6	3	1	8
Green-tailed Towhee	R22	3	2	8	1	0	6
Green-tailed Towhee	R23	2	1	7	3	1	8
Green-tailed Towhee	R24	3	1	8	4	3	9
Green-tailed Towhee	R25	2	1	7	4	3	9
Juniper Titmouse	C1	0	0	3	0	0	3
Juniper Titmouse	C2	0	0	3	0	0	3
Juniper Titmouse	C3	0	0	3	0	0	3
Juniper Titmouse	C4	0	0	3	0	0	3
Juniper Titmouse	C5	0	0	3	0	0	3
Juniper Titmouse	C6	0	0	3	0	0	3
Juniper Titmouse	C7	0	0	3	0	0	3
Juniper Titmouse	C8	0	0	3	0	0	3
Juniper Titmouse	C9	0	0	3	0	0	3
Juniper Titmouse	C10	0	0	3	0	0	3

Juniper Titmouse	C11	0	0	3	0	0	3
Juniper Titmouse	C12	0	0	3	0	0	3
Juniper Titmouse	C13	0	0	3	0	0	3
Juniper Titmouse	C14	0	0	4	0	0	4
Juniper Titmouse	C15	0	0	4	0	0	4
Juniper Titmouse	C16	0	0	5	0	0	5
Juniper Titmouse	C17	0	0	5	0	0	5
Juniper Titmouse	C18	1	0	10	2	1	11
Juniper Titmouse	C19	1	0	10	1	0	10
Juniper Titmouse	C20	2	1	11	1	0	10
Juniper Titmouse	C21	1	0	10	1	0	10
Juniper Titmouse	C22	0	0	5	0	0	5
Juniper Titmouse	C23	0	0	5	1	1	6
Juniper Titmouse	C24	2	1	11	1	0	10
Juniper Titmouse	C25	1	0	10	1	0	10
Juniper Titmouse	R1	1	0	9	1	0	9
Juniper Titmouse	R2	1	0	9	1	0	9
Juniper Titmouse	R3	1	0	9	1	0	9
Juniper Titmouse	R4	1	0	9	1	0	9
Juniper Titmouse	R5	1	0	9	1	0	9
Juniper Titmouse	R6	1	0	9	1	0	9
Juniper Titmouse	R7	1	0	9	1	0	9
Juniper Titmouse	R8	1	0	9	1	0	9
Juniper Titmouse	R9	1	0	9	1	0	9
Juniper Titmouse	R10	4	3	14	1	0	9
Juniper Titmouse	R11	1	0	9	1	0	9
Juniper Titmouse	R12	1	0	9	1	0	9
Juniper Titmouse	R13	1	0	9	1	0	9
Juniper Titmouse	R14	1	0	9	1	0	9
Juniper Titmouse	R15	1	0	9	1	0	9
Juniper Titmouse	R16	1	0	9	1	0	9
Juniper Titmouse	R17	1	0	9	1	0	9
Juniper Titmouse	R18	1	0	9	1	0	9
Juniper Titmouse	R19	2	1	10	1	0	9
Juniper Titmouse	R20	1	0	9	1	0	9
Juniper Titmouse	R21	2	1	10	1	0	9
Juniper Titmouse	R22	2	1	10	1	0	9
Juniper Titmouse	R23	2	1	10	1	0	9
Juniper Titmouse	R24	1	0	9	1	0	9
Juniper Titmouse	R25	1	0	9	2	1	10
Mountain Bluebird	C1	15	1	98	17	2	100
Mountain Bluebird	C2	15	1	98	15	1	98
Mountain Bluebird	C3	16	2	99	17	3	99

Mountain Bluebird	C4	15	1	98	15	1	97
Mountain Bluebird	C5	15	1	98	15	1	98
Mountain Bluebird	C6	15	1	98	15	1	98
Mountain Bluebird	C7	15	1	98	16	2	99
Mountain Bluebird	C8	28	4	130	28	4	131
Mountain Bluebird	C9	26	2	129	26	2	129
Mountain Bluebird	C10	26	2	128	28	4	131
Mountain Bluebird	C11	26	2	129	26	2	129
Mountain Bluebird	C12	27	3	129	26	2	129
Mountain Bluebird	C13	27	3	129	27	3	130
Mountain Bluebird	C14	11	0	104	11	0	103
Mountain Bluebird	C15	12	1	104	11	0	103
Mountain Bluebird	C16	127	17	601	127	17	601
Mountain Bluebird	C17	126	16	600	125	15	599
Mountain Bluebird	C18	34	5	165	33	4	164
Mountain Bluebird	C19	32	3	163	35	6	165
Mountain Bluebird	C20	32	3	163	33	4	164
Mountain Bluebird	C21	32	3	163	33	4	164
Mountain Bluebird	C22	133	23	607.025	123	13	597
Mountain Bluebird	C23	126	16	599	129	19	602.025
Mountain Bluebird	C24	33	4	164	33	4	163
Mountain Bluebird	C25	33	4	163	33	4	164
Mountain Bluebird	R1	21	2	117	21	2	117
Mountain Bluebird	R2	22	3	118	21	2	117
Mountain Bluebird	R3	22	3	118	23	4	119
Mountain Bluebird	R4	21	2	117	21	2	117
Mountain Bluebird	R5	31	7	146	26	2	140
Mountain Bluebird	R6	26	2	141	27	3	142
Mountain Bluebird	R7	28	4	142	26	2	141
Mountain Bluebird	R8	28	4	143	26	2	141
Mountain Bluebird	R9	29	5	144	28	4	142
Mountain Bluebird	R10	26	2	140	26	2	141
Mountain Bluebird	R11	26	2	141	27	3	141
Mountain Bluebird	R12	26	2	140.025	26	2	141
Mountain Bluebird	R13	26	2	141	29	5	144
Mountain Bluebird	R14	26	2	140	26	2	140
Mountain Bluebird	R15	26	2	140	26	2	141
Mountain Bluebird	R16	26	2	140.025	27	3	142
Mountain Bluebird	R17	26	2	141	27	3	142
Mountain Bluebird	R18	26	2	140	26	2	141
Mountain Bluebird	R19	26	2	140	28	4	143
Mountain Bluebird	R20	21	2	117	22	3	117
Mountain Bluebird	R21	24	5	120	21	2	117

Mountain Bluebird	R22	22	3	118	21	2	117
Mountain Bluebird	R23	21	2	117	22	3	118
Mountain Bluebird	R24	22	3	118	21	2	117
Mountain Bluebird	R25	21	2	117	24	4	120
Mountain Chickadee	C1	2	0	30	2	0	30
Mountain Chickadee	C2	2	0	30	4	2	32
Mountain Chickadee	C3	3	1	31	2	0	30
Mountain Chickadee	C4	2	0	30	3	1	31
Mountain Chickadee	C5	2	0	30	2	0	30
Mountain Chickadee	C6	2	0	30	2	0	30
Mountain Chickadee	C7	3	1	31	2	0	30
Mountain Chickadee	C8	0	0	7	0	0	7
Mountain Chickadee	C9	0	0	7	0	0	7
Mountain Chickadee	C10	0	0	7	0	0	7
Mountain Chickadee	C11	0	0	7	0	0	7
Mountain Chickadee	C12	0	0	7	0	0	7
Mountain Chickadee	C13	0	0	7	0	0	7
Mountain Chickadee	C14	0	0	12	0	0	12
Mountain Chickadee	C15	0	0	12	0	0	12
Mountain Chickadee	C16	3	0	49	3	0	49
Mountain Chickadee	C17	3	0	49.025	3	0	50
Mountain Chickadee	C18	5	0	75	5	0	76
Mountain Chickadee	C19	8	2	78	6	1	76
Mountain Chickadee	C20	6	1	76	5	0	75
Mountain Chickadee	C21	7	1	77	5	0	75
Mountain Chickadee	C22	5	1	52	3	0	49
Mountain Chickadee	C23	4	1	50	6	2	52
Mountain Chickadee	C24	6	1	76	8	3	78
Mountain Chickadee	C25	6	1	76	5	0	75
Mountain Chickadee	R1	10	3	96	7	0	94
Mountain Chickadee	R2	7	0	94	7	0	94
Mountain Chickadee	R3	8	1	95	11	4	98
Mountain Chickadee	R4	7	0	93	9	2	96
Mountain Chickadee	R5	7	0	103	7	0	104
Mountain Chickadee	R6	10	4	107	9	2	106
Mountain Chickadee	R7	7	0	104	8	1	105
Mountain Chickadee	R8	11	3	107	7	0	104
Mountain Chickadee	R9	9	2	106	8	1	104
Mountain Chickadee	R10	10	3	107	7	0	104
Mountain Chickadee	R11	9	2	106	7	0	104
Mountain Chickadee	R12	9	2	105	8	1	105
Mountain Chickadee	R13	7	0	104	7	0	104
Mountain Chickadee	R14	7	0	103	8	1	104

Mountain Chickadee	R15	7	0	104	7	0	104
Mountain Chickadee	R16	10	3	108	7	0	104
Mountain Chickadee	R17	10	3	107	7	0	104
Mountain Chickadee	R18	8	1	105	7	0	103
Mountain Chickadee	R19	8	1	105	8	1	104
Mountain Chickadee	R20	9	2	95	8	1	94
Mountain Chickadee	R21	10	3	96	8	1	95
Mountain Chickadee	R22	10	3	97	7	0	94
Mountain Chickadee	R23	7	1	94	7	0	94
Mountain Chickadee	R24	9	2	96	7	0	93
Mountain Chickadee	R25	7	0	93.025	7	0	93
Mourning Dove	C1	0	0	4	0	0	4
Mourning Dove	C2	0	0	5	0	0	4
Mourning Dove	C3	1	1	5	0	0	4
Mourning Dove	C4	1	1	5	0	0	4
Mourning Dove	C5	0	0	4	0	0	4
Mourning Dove	C6	0	0	4	0	0	4
Mourning Dove	C7	1	1	5	0	0	4
Mourning Dove	C8	0	0	4	0	0	4
Mourning Dove	C9	0	0	5	0	0	4.025
Mourning Dove	C10	0	0	4	0	0	4
Mourning Dove	C11	0	0	5	0	0	4
Mourning Dove	C12	0	0	5	0	0	4
Mourning Dove	C13	0	0	5	2	2	7
Mourning Dove	C14	0	0	5	1	1	6
Mourning Dove	C15	0	0	5	0	0	5
Mourning Dove	C16	0	0	4	0	0	4
Mourning Dove	C17	0	0	4	1	1	5
Mourning Dove	C18	1	1	6	0	0	5
Mourning Dove	C19	0	0	5	0	0	5
Mourning Dove	C20	0	0	5	0	0	5
Mourning Dove	C21	1	1	6	1	1	6
Mourning Dove	C22	0	0	4	0	0	4
Mourning Dove	C23	0	0	4	0	0	5
Mourning Dove	C24	0	0	5	0	0	5
Mourning Dove	C25	0	0	5	0	0	5
Mourning Dove	R1	3	0	20	3	0	20
Mourning Dove	R2	4	1	21	4	1	21
Mourning Dove	R3	4	1	21	3	0	20
Mourning Dove	R4	5	2	23	3	0	20
Mourning Dove	R5	3	0	20	3	0	20
Mourning Dove	R6	3	0	20	5	1	22
Mourning Dove	R7	6	3	23	3	0	20

Mourning Dove	R8	3	0	20	4	1	21
Mourning Dove	R9	6	2	24	5	2	22
Mourning Dove	R10	5	1	22	5	2	22
Mourning Dove	R11	3	0	20	3	0	20
Mourning Dove	R12	3	0	20	3	0	20
Mourning Dove	R13	4	1	21	3	0	20
Mourning Dove	R14	7	3	25	3	0	20
Mourning Dove	R15	7	3	25	3	0	20
Mourning Dove	R16	5	2	22	4	1	21
Mourning Dove	R17	3	0	20	3	0	20
Mourning Dove	R18	3	0	20	3	0	20
Mourning Dove	R19	4	1	21	5	1	22
Mourning Dove	R20	4	1	21	3	0	20
Mourning Dove	R21	3	0	20	4	1	21
Mourning Dove	R22	4	1	22	3	0	20
Mourning Dove	R23	5	1	23	4	1	21
Mourning Dove	R24	6	2	23	3	0	21
Mourning Dove	R25	6	2	24	5	1	22
Northern Flicker	C1	0	0	9	0	0	9
Northern Flicker	C2	0	0	9	0	0	9
Northern Flicker	C3	0	0	9	0	0	9
Northern Flicker	C4	0	0	9	0	0	9
Northern Flicker	C5	0	0	9	0	0	9
Northern Flicker	C6	0	0	9	0	0	9
Northern Flicker	C7	0	0	9	0	0	9
Northern Flicker	C8	1	0	39	1	0	39
Northern Flicker	C9	1	0	39	1	0	39
Northern Flicker	C10	1	0	39	1	0	39
Northern Flicker	C11	1	0	39	1	0	39
Northern Flicker	C12	2	1	40	1	0	39
Northern Flicker	C13	1	0	39	1	0	39
Northern Flicker	C14	0	0	19	0	0	19
Northern Flicker	C15	0	0	19	0	0	19
Northern Flicker	C16	2	0	90	3	1	91
Northern Flicker	C17	2	0	91	2	0	90
Northern Flicker	C18	0	0	12	0	0	12
Northern Flicker	C19	0	0	12	0	0	12
Northern Flicker	C20	0	0	12	0	0	12
Northern Flicker	C21	0	0	12	0	0	12
Northern Flicker	C22	2	0	91	3	1	91
Northern Flicker	C23	2	0	90	3	1	91
Northern Flicker	C24	0	0	12	0	0	12
Northern Flicker	C25	0	0	12	0	0	12

Northern Flicker	R1	7	0	160	7	0	159
Northern Flicker	R2	9	2	161	7	0	159
Northern Flicker	R3	8	1	160	9	1	162
Northern Flicker	R4	8	1	160	7	0	159
Northern Flicker	R5	7	1	114	5	0	113
Northern Flicker	R6	6	1	114	5	0	113
Northern Flicker	R7	5	0	113	5	0	112
Northern Flicker	R8	5	0	113	6	1	113
Northern Flicker	R9	5	0	113	5	0	113
Northern Flicker	R10	5	0	113	5	0	113
Northern Flicker	R11	5	0	113	5	0	113
Northern Flicker	R12	5	0	113	5	0	113
Northern Flicker	R13	6	1	114	5	0	112
Northern Flicker	R14	7	1	114	6	1	114
Northern Flicker	R15	5	0	113	5	0	113
Northern Flicker	R16	5	0	113	5	0	113
Northern Flicker	R17	6	1	113	5	0	113
Northern Flicker	R18	5	0	112	5	0	113
Northern Flicker	R19	5	0	112	6	1	113
Northern Flicker	R20	7	0	160	7	0	159
Northern Flicker	R21	8	1	160	9	1	162
Northern Flicker	R22	7	0	160	7	0	159
Northern Flicker	R23	8	1	161	7	0	160
Northern Flicker	R24	7	0	160	7	0	160
Northern Flicker	R25	7	0	160	7	0	159
Plumbeous Vireo	C1	0	0	16	0	0	16
Plumbeous Vireo	C2	0	0	16	0	0	16
Plumbeous Vireo	C3	0	0	16	0	0	16
Plumbeous Vireo	C4	0	0	16	0	0	16
Plumbeous Vireo	C5	0	0	16	0	0	16
Plumbeous Vireo	C6	0	0	16	0	0	16
Plumbeous Vireo	C7	0	0	16	0	0	16
Plumbeous Vireo	C8	0	0	17	0	0	17
Plumbeous Vireo	C9	0	0	17	0	0	17
Plumbeous Vireo	C10	0	0	17	0	0	17
Plumbeous Vireo	C11	1	1	18	0	0	17
Plumbeous Vireo	C12	0	0	17	0	0	17
Plumbeous Vireo	C13	0	0	17	0	0	17
Plumbeous Vireo	C14	0	0	17	0	0	17
Plumbeous Vireo	C15	0	0	17	0	0	17
Plumbeous Vireo	C16	0	0	18	0	0	18
Plumbeous Vireo	C17	0	0	18	0	0	18
Plumbeous Vireo	C18	0	0	21	0	0	21

Plumbeous Vireo	C19	0	0	21	0	0	21
Plumbeous Vireo	C20	0	0	21	0	0	21
Plumbeous Vireo	C21	0	0	21	1	1	22
Plumbeous Vireo	C22	0	0	18	0	0	18
Plumbeous Vireo	C23	0	0	18	0	0	18
Plumbeous Vireo	C24	0	0	22	0	0	21
Plumbeous Vireo	C25	1	1	22	0	0	21
Plumbeous Vireo	R1	5	1	274	3	0	273
Plumbeous Vireo	R2	6	2	275	5	1	274
Plumbeous Vireo	R3	4	1	273	4	1	273
Plumbeous Vireo	R4	7	3	276	3	0	272
Plumbeous Vireo	R5	4	1	281	3	0	280
Plumbeous Vireo	R6	7	3	285	5	1	282
Plumbeous Vireo	R7	4	1	281	5	1	282
Plumbeous Vireo	R8	4	1	281	5	1	282
Plumbeous Vireo	R9	5	1	282.025	3	0	280
Plumbeous Vireo	R10	6	2	284	5	1	283
Plumbeous Vireo	R11	4	1	282	4	1	281
Plumbeous Vireo	R12	7	3	285	4	1	282
Plumbeous Vireo	R13	6	3	283	3	0	280
Plumbeous Vireo	R14	3	0	280	3	0	280
Plumbeous Vireo	R15	3	0	280	3	0	281
Plumbeous Vireo	R16	5	1	282	4	1	282
Plumbeous Vireo	R17	4	1	281	3	0	281
Plumbeous Vireo	R18	4	1	281	3	0	280
Plumbeous Vireo	R19	5	1	283	3	0	281
Plumbeous Vireo	R20	3	0	272	3	0	272
Plumbeous Vireo	R21	4	1	274	4	1	274
Plumbeous Vireo	R22	4	1	274	3	0	273
Plumbeous Vireo	R23	3	0	272	3	0	272
Plumbeous Vireo	R24	4	1	274	6	2	276
Plumbeous Vireo	R25	3	0	272	7	2	277
Rock Wren	C1	0	0	1	0	0	1
Rock Wren	C2	0	0	1	0	0	1
Rock Wren	C3	0	0	1	0	0	1
Rock Wren	C4	0	0	1	0	0	1
Rock Wren	C5	0	0	1	0	0	1
Rock Wren	C6	0	0	1	0	0	1
Rock Wren	C7	0	0	1	0	0	1
Rock Wren	C8	2	2	5	0	0	2
Rock Wren	C9	0	0	2	0	0	2
Rock Wren	C10	0	0	2	0	0	2
Rock Wren	C11	0	0	2	0	0	2

Rock Wren	C12	0	0	2	0	0	2
Rock Wren	C13	0	0	2	0	0	2
Rock Wren	C14	0	0	2	0	0	2
Rock Wren	C15	0	0	2	1	1	3
Rock Wren	C16	2	1	5	2	1	6
Rock Wren	C17	1	0	4	3	2	7
Rock Wren	C18	2	0	8	3	1	9
Rock Wren	C19	8	6	14	3	1	9
Rock Wren	C20	3	1	9	3	1	10
Rock Wren	C21	2	0	8	4	2	10
Rock Wren	C22	1	0	4	1	0	4
Rock Wren	C23	2	1	5	2	1	6
Rock Wren	C24	2	0	8	5	2	11
Rock Wren	C25	5	3	11	5	2	11
Rock Wren	R1	0	0	4	0	0	3
Rock Wren	R2	0	0	4	0	0	4
Rock Wren	R3	0	0	4	0	0	3
Rock Wren	R4	0	0	4	0	0	4
Rock Wren	R5	0	0	2	2	2	4
Rock Wren	R6	0	0	2	0	0	2
Rock Wren	R7	0	0	2	1	1	4
Rock Wren	R8	0	0	2	0	0	2
Rock Wren	R9	0	0	2	0	0	2
Rock Wren	R10	0	0	2	0	0	2
Rock Wren	R11	0	0	2	0	0	2
Rock Wren	R12	1	1	3	0	0	2
Rock Wren	R13	0	0	2	0	0	2
Rock Wren	R14	0	0	2	0	0	2
Rock Wren	R15	0	0	2	0	0	2
Rock Wren	R16	0	0	2	0	0	2
Rock Wren	R17	0	0	2	0	0	2
Rock Wren	R18	0	0	2	0	0	2
Rock Wren	R19	0	0	2	0	0	2
Rock Wren	R20	0	0	4	0	0	4
Rock Wren	R21	0	0	4	0	0	4
Rock Wren	R22	6	5	10	3	2	7
Rock Wren	R23	1	1	4	0	0	4
Rock Wren	R24	0	0	3	0	0	3
Rock Wren	R25	3	1	6	1	1	5
Spotted Towhee	C1	2	2	5	4	3	7
Spotted Towhee	C2	3	2	6	3	2	6
Spotted Towhee	C3	2	1	5	1	1	4
Spotted Towhee	C4	1	1	4	0	0	3

Spotted Towhee	C5	2	1	5	2	1	5
Spotted Towhee	C6	1	1	4	0	0	3
Spotted Towhee	C7	3	2	5	4	3	6
Spotted Towhee	C8	2	2	4	1	1	3
Spotted Towhee	C9	0	0	2	0	0	2
Spotted Towhee	C10	0	0	2	2	2	4
Spotted Towhee	C11	0	0	2	0	0	2
Spotted Towhee	C12	0	0	2	0	0	2
Spotted Towhee	C13	0	0	2	0	0	2
Spotted Towhee	C14	0	0	2	1	1	3
Spotted Towhee	C15	0	0	2	0	0	2
Spotted Towhee	C16	0	0	2	1	1	3
Spotted Towhee	C17	1	1	3	1	1	3
Spotted Towhee	C18	2	1	4	4	3	7
Spotted Towhee	C19	0	0	2	2	1	4
Spotted Towhee	C20	0	0	2	0	0	2
Spotted Towhee	C21	0	0	2	5	4	8
Spotted Towhee	C22	0	0	2	0	0	2
Spotted Towhee	C23	0	0	2	1	1	3
Spotted Towhee	C24	0	0	2	1	1	3
Spotted Towhee	C25	0	0	2	0	0	2
Spotted Towhee	R1	2	1	5	3	2	6
Spotted Towhee	R2	4	2	7	4	2	7
Spotted Towhee	R3	4	2	8	5	3	8
Spotted Towhee	R4	3	2	6	2	1	5
Spotted Towhee	R5	0	0	3	3	2	5
Spotted Towhee	R6	0	0	2	0	0	3
Spotted Towhee	R7	0	0	3	1	1	4
Spotted Towhee	R8	2	1	4	3	2	6
Spotted Towhee	R9	0	0	2	0	0	3
Spotted Towhee	R10	3	2	5	2	1	5
Spotted Towhee	R11	2	1	5	4	3	7
Spotted Towhee	R12	0	0	2	0	0	3
Spotted Towhee	R13	2	1	4	2	2	5
Spotted Towhee	R14	2	1	5	1	1	4
Spotted Towhee	R15	0	0	3	0	0	3
Spotted Towhee	R16	1	1	4	1	1	3
Spotted Towhee	R17	3	2	5	6	5	9
Spotted Towhee	R18	0	0	2	0	0	3
Spotted Towhee	R19	0	0	3	1	1	4
Spotted Towhee	R20	3	2	6	6	4	9
Spotted Towhee	R21	2	1	5	2	1	6
Spotted Towhee	R22	1	0	4	3	2	7

Spotted Towhee	R23	1	0	4	1	0	4
Spotted Towhee	R24	2	1	5	2	1	6
Spotted Towhee	R25	2	1	5	1	0	4
Vesper Sparrow	C1	0	0	3	0	0	3
Vesper Sparrow	C2	0	0	3	0	0	3
Vesper Sparrow	C3	0	0	3	0	0	3
Vesper Sparrow	C4	0	0	3	0	0	3
Vesper Sparrow	C5	0	0	3	0	0	3
Vesper Sparrow	C6	0	0	3	0	0	3
Vesper Sparrow	C7	0	0	3	0	0	3
Vesper Sparrow	C8	0	0	3	0	0	3
Vesper Sparrow	C9	0	0	3	0	0	3
Vesper Sparrow	C10	0	0	3	0	0	3
Vesper Sparrow	C11	0	0	3	0	0	3
Vesper Sparrow	C12	0	0	3	0	0	3
Vesper Sparrow	C13	0	0	3	0	0	3
Vesper Sparrow	C14	0	0	5	0	0	5
Vesper Sparrow	C15	0	0	5	0	0	5
Vesper Sparrow	C16	0	0	13	0	0	13
Vesper Sparrow	C17	0	0	13	0	0	13
Vesper Sparrow	C18	0	0	3	0	0	3
Vesper Sparrow	C19	0	0	3	0	0	3
Vesper Sparrow	C20	0	0	3	0	0	3
Vesper Sparrow	C21	0	0	3	0	0	3
Vesper Sparrow	C22	0	0	13	0	0	13
Vesper Sparrow	C23	0	0	13	0	0	13
Vesper Sparrow	C24	0	0	3	0	0	3
Vesper Sparrow	C25	0	0	3	0	0	3
Vesper Sparrow	R1	0	0	10	0	0	10
Vesper Sparrow	R2	0	0	10	0	0	10
Vesper Sparrow	R3	0	0	10	0	0	10
Vesper Sparrow	R4	0	0	10	0	0	10
Vesper Sparrow	R5	2	0	73	2	0	73
Vesper Sparrow	R6	2	0	73	2	0	73
Vesper Sparrow	R7	2	0	74	2	0	73
Vesper Sparrow	R8	2	0	73	2	0	73
Vesper Sparrow	R9	2	0	73	2	0	73
Vesper Sparrow	R10	2	0	73	2	0	72
Vesper Sparrow	R11	2	0	73	2	0	73
Vesper Sparrow	R12	2	0	73	2	0	73
Vesper Sparrow	R13	2	0	73	2	0	73
Vesper Sparrow	R14	2	0	73	2	0	73
Vesper Sparrow	R15	3	1	74	3	1	74

Vesper Sparrow	R16	2	0	73	2	0	73
Vesper Sparrow	R17	2	0	73	2	0	73
Vesper Sparrow	R18	2	0	74	2	0	73
Vesper Sparrow	R19	2	0	74	2	0	73
Vesper Sparrow	R20	0	0	10	0	0	10
Vesper Sparrow	R21	0	0	10	0	0	10
Vesper Sparrow	R22	0	0	10	0	0	10
Vesper Sparrow	R23	0	0	10	0	0	10
Vesper Sparrow	R24	0	0	10	0	0	10
Vesper Sparrow	R25	0	0	10	0	0	10
Violet-green Swallow	C1	4	0	49	4	0	50
Violet-green Swallow	C2	4	0	50	4	0	49
Violet-green Swallow	C3	5	1	51	5	1	50
Violet-green Swallow	C4	4	0	50	4	0	49
Violet-green Swallow	C5	4	0	50	4	0	50
Violet-green Swallow	C6	7	3	52	5	1	51
Violet-green Swallow	C7	6	2	51	5	1	52
Violet-green Swallow	C8	0	0	9	0	0	9
Violet-green Swallow	C9	0	0	9	0	0	9
Violet-green Swallow	C10	0	0	9	0	0	9
Violet-green Swallow	C11	0	0	9	0	0	9
Violet-green Swallow	C12	0	0	9	1	1	10
Violet-green Swallow	C13	0	0	9	0	0	9
Violet-green Swallow	C14	0	0	8	0	0	8
Violet-green Swallow	C15	0	0	8	0	0	8
Violet-green Swallow	C16	4	2	25	2	0	23
Violet-green Swallow	C17	2	0	23	2	0	23
Violet-green Swallow	C18	3	1	24	2	0	23
Violet-green Swallow	C19	2	0	23	2	0	23
Violet-green Swallow	C20	3	1	24	2	0	23
Violet-green Swallow	C21	2	0	23	2	0	23
Violet-green Swallow	C22	2	0	23	2	0	23
Violet-green Swallow	C23	2	0	23	2	0	23
Violet-green Swallow	C24	2	0	23	2	0	23
Violet-green Swallow	C25	4	2	25	2	0	23
Violet-green Swallow	R1	0	0	8	0	0	8
Violet-green Swallow	R2	0	0	8	0	0	8
Violet-green Swallow	R3	0	0	8	0	0	8
Violet-green Swallow	R4	0	0	8	1	1	9
Violet-green Swallow	R5	10	3	89	7	1	87
Violet-green Swallow	R6	6	0	84	6	0	85
Violet-green Swallow	R7	6	0	85	6	0	85
Violet-green Swallow	R8	6	0	85	6	0	85

Violet-green Swallow	R9	9	3	89	6	0	84
Violet-green Swallow	R10	6	0	85	6	0	84
Violet-green Swallow	R11	8	2	87	6	0	85
Violet-green Swallow	R12	6	0	84	6	0	85
Violet-green Swallow	R13	7	1	85	7	1	86
Violet-green Swallow	R14	7	1	85	8	2	87
Violet-green Swallow	R15	10	3	89	10	3	90
Violet-green Swallow	R16	8	2	87	8	2	86
Violet-green Swallow	R17	6	0	85	6	0	85
Violet-green Swallow	R18	7	1	86	6	0	85
Violet-green Swallow	R19	6	0	85	9	2	88
Violet-green Swallow	R20	0	0	8	0	0	8
Violet-green Swallow	R21	0	0	8	0	0	8
Violet-green Swallow	R22	0	0	8	0	0	8
Violet-green Swallow	R23	0	0	8	0	0	8
Violet-green Swallow	R24	0	0	8	0	0	8
Violet-green Swallow	R25	0	0	8	0	0	8
White-breasted Nuthatch	C1	0	0	2	1	1	4
White-breasted Nuthatch	C2	0	0	2	2	2	4
White-breasted Nuthatch	C3	0	0	2	0	0	2
White-breasted Nuthatch	C4	0	0	2	0	0	2
White-breasted Nuthatch	C5	0	0	2	0	0	2
White-breasted Nuthatch	C6	0	0	2	0	0	2
White-breasted Nuthatch	C7	0	0	2	0	0	2
White-breasted Nuthatch	C8	0	0	2	1	1	3
White-breasted Nuthatch	C9	0	0	2	0	0	2
White-breasted Nuthatch	C10	0	0	2	0	0	2
White-breasted Nuthatch	C11	0	0	2	0	0	2
White-breasted Nuthatch	C12	0	0	2	0	0	2
White-breasted Nuthatch	C13	1	1	3	0	0	2
White-breasted Nuthatch	C14	0	0	2	0	0	2
White-breasted Nuthatch	C15	1	1	3	1	1	3
White-breasted Nuthatch	C16	0	0	2	0	0	2
White-breasted Nuthatch	C17	0	0	2	0	0	2
White-breasted Nuthatch	C18	0	0	2	0	0	2
White-breasted Nuthatch	C19	2	2	4	0	0	2
White-breasted Nuthatch	C20	0	0	2	0	0	2
White-breasted Nuthatch	C21	0	0	2	1	1	4
White-breasted Nuthatch	C22	1	1	4	0	0	2
White-breasted Nuthatch	C23	0	0	2	1	1	4
White-breasted Nuthatch	C24	1	1	3	1	1	3
White-breasted Nuthatch	C25	0	0	2	0	0	2
White-breasted Nuthatch	R1	2	1	5	1	0	4

White-breasted Nuthatch	R2	2	1	6	2	1	6
White-breasted Nuthatch	R3	3	1	7	1	0	4
White-breasted Nuthatch	R4	3	2	6	2	1	5
White-breasted Nuthatch	R5	1	0	5	2	1	6
White-breasted Nuthatch	R6	2	1	6	1	0	5
White-breasted Nuthatch	R7	4	3	8	2	1	6
White-breasted Nuthatch	R8	2	1	7	2	1	6
White-breasted Nuthatch	R9	2	1	6	2	1	6
White-breasted Nuthatch	R10	2	1	7	2	1	7
White-breasted Nuthatch	R11	1	0	5	3	2	7
White-breasted Nuthatch	R12	4	2	9	4	2	9
White-breasted Nuthatch	R13	2	1	6	1	0	5
White-breasted Nuthatch	R14	1	0	5	1	0	5
White-breasted Nuthatch	R15	1	0	5	1	0	5
White-breasted Nuthatch	R16	1	0	5	1	0	5
White-breasted Nuthatch	R17	2	1	6	1	0	5
White-breasted Nuthatch	R18	2	1	6	1	0	5
White-breasted Nuthatch	R19	2	1	6	2	1	6
White-breasted Nuthatch	R20	1	0	4	1	0	4
White-breasted Nuthatch	R21	3	2	7	2	1	6
White-breasted Nuthatch	R22	1	0	5	1	0	4
White-breasted Nuthatch	R23	1	0	5	1	0	4
White-breasted Nuthatch	R24	1	0	4	1	0	4
White-breasted Nuthatch	R25	1	0	4	2	1	6

Table A2.3. Comprehensive list of birds detected in the Piceance Basin study site, Colorado, USA, and their classification by foraging guild (De Graaf et al. 1985), nesting guild (Degraaf and Wentworth 1986), and habitat preference (Rodewald 2015).

Common Name	Latin Name	# of detections		Foraging Guild	Nesting Guild	Habitat group
		Chai ned	Refere nce			
Cooper's hawk	<i>Accipiter cooperii</i>	1	6	Air hawk	Tree	Woodland
Red-tailed hawk	<i>Buteo jamaicensis</i>	0	3	Ground	Tree	Generalist
Merlin	<i>Falco columbarius</i>	1	0	Air hawk	Tree	Shrubland/grassland
American kestrel	<i>Falco sparverius</i>	1	1	Air hawk	Cavity	Shrubland/grassland
Wild turkey	<i>Meleagris gallopavo</i>	0	1	Ground	Ground	Generalist
Mourning dove	<i>Zenaidura macroura</i>	10	54	Ground	Tree	Generalist
Common nighthawk	<i>Chordeiles minor</i>	10	5	Air screener	Ground	Generalist
Broad-tailed hummingbird	<i>Selasphorus platycercus</i>	7	7	Floral-hover gleaner	Tree	Woodland
Black-chinned hummingbird	<i>Archilochus alexandri</i>	1	0	Floral-hover gleaner	Tree	Woodland
Hairy woodpecker	<i>Picoides villosus</i>	2	13	Bark gleaner	Cavity	Woodland
Northern flicker	<i>Colaptes auratus</i>	4	20	Ground	Cavity	Woodland
Western wood pewee	<i>Contopus sordidulus</i>	0	1	Air sallier	Tree	Woodland
Empidonax flycatcher	<i>Empidonax sp.</i>	56	66	Air sallier	Shrub Permanent Structure	Generalist Shrubland/grassland
Say's phoebe	<i>Sayornis saya</i>	0	1	Air sallier	Tree	Woodland
Ash-throated flycatcher	<i>Myiarchus cinerascens</i>	17	15	Shrub gleaner, upper canopy gleaner, Air sallier	Cavity	Woodland
Western kingbird	<i>Tyrannus verticalis</i>	0	3	Air sallier	Tree	Generalist
Plumbeous vireo	<i>Vireo plumbeus</i>	3	64	Shrub forager, shrub gleaner, upper canopy gleaner	Tree	Woodland
Western scrub-jay	<i>Aphelocoma</i>	7	15	Ground	Tree	Generalist

Pinyon jay	<i>Gymnorhinus californica cyanocephalus</i>	3	2	Ground	Tree	Generalist
Clark's nutcracker	<i>Nucifraga columbiana</i>	0	29	Ground	Tree	Woodland
Black-billed magpie	<i>Pica hudsonia</i>	0	1	Ground	Tree	Generalist
Common raven	<i>Corvus corax</i>	4	20	Ground	Permanent Structure	Generalist
Northern rough-winged swallow	<i>Stelgidopteryx serripennis</i>	1	0	Air screener	Burrow	Generalist
Violet-green swallow	<i>Tachycineta thalassina</i>	17	34	Air screener	Cavity	Generalist
Juniper titmouse	<i>Baeolophus ridgwayi</i>	4	10	Shrub forager	Cavity	Woodland
Mountain chickadee	<i>Poecile gambeli</i>	23	56	Shrub forager, shrub gleaner, upper canopy gleaner	Cavity	Woodland
Bushtit	<i>Psaltriparus minimu</i>	1	6	Shrub forager, shrub gleaner, upper canopy gleaner	Tree	Generalist
Red-breasted nuthatch	<i>Sitta canadensis</i>	0	9	Bark gleaner	Cavity	Woodland
White-breasted nuthatch	<i>Sitta carolinensis</i>	19	56	Bark gleaner	Cavity	Woodland
Bewick's wren	<i>Thryomanes bewickii</i>	0	11	Ground	Cavity	Generalist
House wren	<i>Troglodytes aedon</i>	7	8	Shrub forager	Cavity	Woodland
Rock wren	<i>Salpinctes obsoletus</i>	39	25	Ground	Ground	Generalist
Canyon wren	<i>Catherpes mexicanus</i>	0	1	Ground	Permenant Structure	N/A
Ruby-crowned kinglet	<i>Regulus calendula</i>	0	1	Shrub gleaner, upper canopy gleaner	Tree	Woodland
Blue-gray gnatcatcher	<i>Polioptila caerulea</i>	100	58	Upper canopy forager	Tree	Generalist
Mountain bluebird	<i>Sialia currucoides</i>	60	37	Shrub forager, Ground	Cavity	Shrubland/grassland

American robin	<i>Turdus migratorius</i>	3	12	Ground	Tree	Generalist
Hermit thrush	<i>Catharus guttatus</i>	0	7	Ground	Ground	Woodland
Sage thrasher	<i>Oreoscoptes montanus</i>	0	3	Ground	Shrub	Shrubland/grassland
Cedar waxwing	<i>Bombycilla cedrorum</i>	0	1	Air sallier	Tree	Generalist
Orange-crowned warbler	<i>Oreothlypis celata</i>	0	1	Shrub forager, shrub gleaner, upper canopy gleaner	Ground	Generalist
Virginia warbler	<i>Oreothlypis virginiae</i>	0	5	Shrub gleaner, upper canopy gleaner	Ground	Woodland
Black-throated gray warbler	<i>Setophaga nigrescens</i>	31	69	Shrub gleaner, upper canopy gleaner	Tree	Woodland
MacGillivray's warbler	<i>Geothlypis tolmiei</i>	0	2	Shrub forager	Shrub	Generalist
Spotted towhee	<i>Pipilo maculatus</i>	66	103	Ground	Ground	Generalist
Green-tailed towhee	<i>Pipilo chlorurus</i>	102	53	Ground	Shrub	Shrubland/grassland
Brewer's sparrow	<i>Spizella breweri</i>	176	14	Ground	Shrub	Shrubland/grassland
Chipping sparrow	<i>Spizella passerina</i>	47	40	Ground	Shrub	Generalist
Savannah sparrow	<i>Passerculus sandwichensis</i>	2	0	Ground	Ground	Shrubland/grassland
Vesper sparrow	<i>Pooecetes gramineus</i>	0	2	Ground	Ground	Shrubland/grassland
Lark sparrow	<i>Chondestes grammacus</i>	0	4	Ground	Ground	Shrubland/grassland
White-crowned sparrow	<i>Zonotrichia leucophrys</i>	1	0	Ground	Ground	Generalist
Dark-eyed junco	<i>Junco hyemalis</i>	5	2	Ground	Ground	Generalist
Western meadowlark	<i>Sturnella neglecta</i>	0	4	Ground	Ground	Shrubland/grassland
Brown-headed cowbird	<i>Molothrus ater</i>	1	5	Ground	Tree	Shrubland/grassland

Red-winged blackbird	<i>Agelaius phoeniceus</i>	0	1	Ground	Shrub	Shrubland/grassland
Evening grosbeak	<i>Coccothraustes vespertinus</i>	0	4	Upper canopy forager, Ground	Tree	Woodland
Cassin's finch	<i>Haemorhous cassinii</i>	7	26	Ground	Tree	Woodland
House finch	<i>Haemorhous mexicanus</i>	4	8	Ground	Tree	Generalist
Red crossbill	<i>Loxia curvirostra</i>	0	7	Upper canopy forager	Tree	Woodland
Lesser goldfinch	<i>Spinus psaltria</i>	0	3	Shrub gleaner, upper canopy gleaner, Ground	Tree	Generalist

APPENDIX 3

SUPPORTING INFORMATION FOR CHAPTER 3

Table A3. Full list of mammal species and the number and proportion of sites each species was detected on remotely-triggered wildlife cameras at historically chained and reference sites in Pinyon-juniper woodlands from April-September 2014 in the Piceance Basin of Northwest Colorado.

Species	Chaining	Prop. of Sites	Reference	Prop. of Sites
American Badger	4	0.18	9	0.41
Black Bear	0	0.00	4	0.18
Black-tailed Jackrabbit	1	0.05	1	0.05
Bobcat	6	0.27	9	0.41
Bushy-tailed Woodrat	1	0.05	1	0.05
Chipmunk	7	0.32	5	0.23
Cow	2	0.09	9	0.41
Coyote	15	0.68	14	0.64
Domestic Dog	3	0.14	4	0.18
Elk	17	0.77	16	0.73
Golden-mantled Ground Squirrel	13	0.59	13	0.59
Human	5	0.23	4	0.18
Long-tailed Weasel	1	0.05	1	0.05
Mountain Cottontail	19	0.86	21	0.95
Mountain Lion	1	0.05	7	0.32
Mule Deer	20	0.91	19	0.86
Red Fox	0	0.00	1	0.05
Rock Squirrel	2	0.09	4	0.18
Striped Skunk	1	0.05	2	0.09
Western Spotted Skunk	1	0.05	3	0.14
Wild Horse	6	0.27	4	0.18
Wyoming Ground Squirrel	0	0.00	1	0.05

APPENDIX 4

SUPPORTING INFORMATION FOR CHAPTER 4

Table A4.1. A complete list of bird species used to analyze community composition, and their respective maximum detection distance and where they generally display in the vegetation strata in the Piceance Basin of Northwest Colorado.

Species	Max Detection Distance	Display Strata
American Robin	>150 m	Mid-level
Ash-throated Flycatcher	>150 m	Top
Yellow-rumped Warbler	>50 m and ≤100 m	Mid-level
Black-chinned Hummingbird	≤50 m	Mid-level
Bewick's Wren	>150 m	Mid-level
Blue-gray gnatcatcher	>150 m	Mid-level
Brown-headed Cowbird	>150 m	Top
Brewer's Sparrow	>150 m	Top
Broad-tailed Hummingbird	>50 m and ≤100 m	Mid-level
Black-throated Gray Warbler	>150 m	Top
Bullock's Warbler	>150 m	Top
Bushtit	>50 m and ≤100 m	Mid-level
Cassin's Finch	>150 m	Top
Canyon Wren	>150 m	Top
Chipping Sparrow	>150 m	Mid-level
Clark's Nutcracker	>150 m	Top
Common Nighthawk	>150 m	Aerial
Common Raven	>150 m	Top
Dark-eyed Junco	>100 m and ≤150 m	Mid-level
<i>Empidonax</i> Flycatcher	>150 m	Top
Green-tailed Towhee	>150 m	Top
Hairy Woodpecker	>150 m	Mid-level
Hermit Thrush	>150 m	Ground
House Finch	>150 m	Mid-level
House Wren	>150 m	Mid-level
Juniper Titmouse	>150 m	Mid-level
Lark Sparrow	>150 m	Ground
Lesser Goldfinch	>100 m and ≤150 m	Mid-level
Lincoln Sparrow	>50 m and ≤100 m	Ground
MacGillivray's Warbler	>150 m	Mid-level
Mountain Bluebird	>150 m	Top
Mountain Chickadee	>150 m	Mid-level
Mourning Dove	>150 m	Ground
Northern Flicker	>150 m	Mid-level
Orange-crowned Warbler	>50 m and ≤100 m	Mid-level
Pinyon Jay	>150 m	Top
Plumbeous Vireo	>150 m	Mid-level

Red-breasted Nuthatch	>150 m	Mid-level
Ruby-crowned Kinglet	≤50 m	Mid-level
Rock Wren	>150 m	Ground
Say's Phoebe	>150 m	Top
Spotted Towhee	>150 m	Mid-level
Vesper Sparrow	>150 m	Ground
Violet-green Swallow	>150 m	Aerial
Virginia Warbler	>100 m and ≤150 m	Mid-level
White-breasted Nuthatch	>150 m	Mid-level
Western Bluebird	≤50 m	Top
Western Kingbird	>150 m	Top
Western Meadowlark	>150 m	Ground
Western Scrub Jay	>150 m	Top
Western Kingbird	>100 m and ≤150 m	Top
Western Wood Pewee	>150 m	Top
White-throated Swallow	>150 m	Aerial

Table A4.2. The median value of the posterior distribution of site abundance (N_i) and associated 95% credible intervals for each bird species at fire (F), fire reference (FR), mechanically disturbed (H), and reference (R) sites in the Piceance Basin of Northwest Colorado.

Species	Site	2013			2014		
		N_i	LCI	UCI	N_i	LCI	UCI
Blue-gray Gnatcatcher	F1	0	0	1	0	0	1
Blue-gray Gnatcatcher	F10	0	0	1	0	0	1
Blue-gray Gnatcatcher	F11	1	1	3	0	0	2
Blue-gray Gnatcatcher	F12	0	0	2	0	0	2
Blue-gray Gnatcatcher	F13	0	0	2	0	0	2
Blue-gray Gnatcatcher	F14	0	0	2	0	0	2
Blue-gray Gnatcatcher	F15	0	0	2	0	0	2
Blue-gray Gnatcatcher	F16	0	0	2	0	0	2
Blue-gray Gnatcatcher	F17	0	0	2	0	0	2
Blue-gray Gnatcatcher	F18	0	0	2	0	0	2
Blue-gray Gnatcatcher	F19	0	0	2	0	0	2
Blue-gray Gnatcatcher	F2	0	0	2	1	1	3
Blue-gray Gnatcatcher	F20	0	0	2	1	1	3
Blue-gray Gnatcatcher	F21	0	0	2	0	0	2
Blue-gray Gnatcatcher	F22	0	0	2	0	0	2
Blue-gray Gnatcatcher	F3	0	0	2	0	0	2
Blue-gray Gnatcatcher	F4	1	1	3	0	0	2
Blue-gray Gnatcatcher	F5	1	1	4	2	2	5
Blue-gray Gnatcatcher	F6	2	1	5	2	1	5
Blue-gray Gnatcatcher	F7	0	0	2	0	0	2
Blue-gray Gnatcatcher	F8	0	0	2	0	0	2
Blue-gray Gnatcatcher	F9	0	0	2	0	0	2
Blue-gray Gnatcatcher	FR1	0	0	4	0	0	4
Blue-gray Gnatcatcher	FR10	0	0	4	0	0	4
Blue-gray Gnatcatcher	FR11	2	1	7	3	1	8
Blue-gray Gnatcatcher	FR12	1	0	5	2	1	6
Blue-gray Gnatcatcher	FR13	3	2	8	1	0	5
Blue-gray Gnatcatcher	FR14	1	0	7	2	1	8
Blue-gray Gnatcatcher	FR15	1	0	7	1	0	7
Blue-gray Gnatcatcher	FR16	1	0	7	3	1	8
Blue-gray Gnatcatcher	FR17	3	1	8	3	1	8
Blue-gray Gnatcatcher	FR18	3	2	9	5	4	11
Blue-gray Gnatcatcher	FR19	2	1	8	4	2	10
Blue-gray Gnatcatcher	FR2	2	1	7	1	0	6
Blue-gray Gnatcatcher	FR20	4	2	10	4	2	9
Blue-gray Gnatcatcher	FR21	1	0	7	1	0	7
Blue-gray Gnatcatcher	FR22	3	1	8	2	1	8

Blue-gray Gnatcatcher	FR3	1	0	6	2	1	7
Blue-gray Gnatcatcher	FR4	1	0	6	3	1	8
Blue-gray Gnatcatcher	FR5	4	1	14	5	2	15
Blue-gray Gnatcatcher	FR6	5	2	15	3	0	12
Blue-gray Gnatcatcher	FR7	4	1	10	3	1	9
Blue-gray Gnatcatcher	FR8	2	1	8	1	0	8
Blue-gray Gnatcatcher	FR9	4	2	11	3	2	10
Blue-gray Gnatcatcher	H1	0	0	3	0	0	3
Blue-gray Gnatcatcher	H10	1	1	4	1	1	4
Blue-gray Gnatcatcher	H11	1	1	4	1	1	4
Blue-gray Gnatcatcher	H12	0	0	3	0	0	3
Blue-gray Gnatcatcher	H13	0	0	3	2	1	5
Blue-gray Gnatcatcher	H14	1	1	4	0	0	3
Blue-gray Gnatcatcher	H15	2	1	6	0	0	3
Blue-gray Gnatcatcher	H16	0	0	3	0	0	3
Blue-gray Gnatcatcher	H17	0	0	3	0	0	3
Blue-gray Gnatcatcher	H18	0	0	3	0	0	3
Blue-gray Gnatcatcher	H19	0	0	3	1	1	4
Blue-gray Gnatcatcher	H2	1	1	4	2	1	5
Blue-gray Gnatcatcher	H20	0	0	3	0	0	3
Blue-gray Gnatcatcher	H21	0	0	3	0	0	3
Blue-gray Gnatcatcher	H22	0	0	3	0	0	3
Blue-gray Gnatcatcher	H23	1	1	4	1	1	4
Blue-gray Gnatcatcher	H24	3	2	6	0	0	3
Blue-gray Gnatcatcher	H25	1	1	4	0	0	3
Blue-gray Gnatcatcher	H3	1	1	4	0	0	3
Blue-gray Gnatcatcher	H4	0	0	3	1	1	4
Blue-gray Gnatcatcher	H5	2	1	5	0	0	3
Blue-gray Gnatcatcher	H6	0	0	3	0	0	3
Blue-gray Gnatcatcher	H7	1	1	4	2	1	5
Blue-gray Gnatcatcher	H8	0	0	3	0	0	3
Blue-gray Gnatcatcher	H9	0	0	3	0	0	3
Blue-gray Gnatcatcher	R1	1	0	5	1	0	4
Blue-gray Gnatcatcher	R10	2	1	6	2	1	6
Blue-gray Gnatcatcher	R11	4	2	8	1	0	5
Blue-gray Gnatcatcher	R12	2	1	6	1	0	5
Blue-gray Gnatcatcher	R13	1	0	5	2	1	5
Blue-gray Gnatcatcher	R14	2	1	6	1	0	5
Blue-gray Gnatcatcher	R15	3	2	7	1	0	5
Blue-gray Gnatcatcher	R16	1	0	5	2	1	5
Blue-gray Gnatcatcher	R17	3	2	7	1	0	5
Blue-gray Gnatcatcher	R18	1	0	5	3	2	7
Blue-gray Gnatcatcher	R19	1	0	5	2	1	6

Blue-gray Gnatcatcher	R2	2	1	6	2	1	6
Blue-gray Gnatcatcher	R20	2	1	6	3	1	7
Blue-gray Gnatcatcher	R21	1	0	5	1	0	5
Blue-gray Gnatcatcher	R22	2	1	6	1	0	4
Blue-gray Gnatcatcher	R23	1	0	5	1	0	5
Blue-gray Gnatcatcher	R24	2	1	6	1	0	5
Blue-gray Gnatcatcher	R25	1	0	5	1	0	5
Blue-gray Gnatcatcher	R3	3	2	7	1	0	5
Blue-gray Gnatcatcher	R4	1	0	4	1	0	5
Blue-gray Gnatcatcher	R5	1	0	4	1	0	5
Blue-gray Gnatcatcher	R6	1	0	4	3	1	7
Blue-gray Gnatcatcher	R7	2	1	6	1	0	5
Blue-gray Gnatcatcher	R8	1	0	4	1	0	5
Blue-gray Gnatcatcher	R9	1	0	5	3	2	7
Brewer's Sparrow	F1	0	0	8	0	0	8
Brewer's Sparrow	F10	0	0	7	0	0	7
Brewer's Sparrow	F11	0	0	7	0	0	7
Brewer's Sparrow	F12	1	0	13	1	0	13
Brewer's Sparrow	F13	1	0	13	2	1	14
Brewer's Sparrow	F14	0	0	6	0	0	6
Brewer's Sparrow	F15	0	0	6	0	0	6
Brewer's Sparrow	F16	0	0	6	0	0	6
Brewer's Sparrow	F17	0	0	6	0	0	6
Brewer's Sparrow	F18	0	0	6	0	0	6
Brewer's Sparrow	F19	0	0	6	1	1	7
Brewer's Sparrow	F2	0	0	7	0	0	7
Brewer's Sparrow	F20	0	0	6	0	0	6
Brewer's Sparrow	F21	0	0	6	0	0	6
Brewer's Sparrow	F22	0	0	6	0	0	6
Brewer's Sparrow	F3	0	0	7	0	0	7
Brewer's Sparrow	F4	0	0	7	0	0	7
Brewer's Sparrow	F5	0	0	6	0	0	6
Brewer's Sparrow	F6	0	0	6	0	0	6
Brewer's Sparrow	F7	0	0	6	0	0	6
Brewer's Sparrow	F8	0	0	6	0	0	6
Brewer's Sparrow	F9	0	0	6	0	0	6
Brewer's Sparrow	FR1	2	0	25	2	0	25
Brewer's Sparrow	FR10	2	0	22	2	0	22
Brewer's Sparrow	FR11	2	0	22	2	0	22
Brewer's Sparrow	FR12	3	0	40	3	0	39
Brewer's Sparrow	FR13	4	1	40	3	0	39
Brewer's Sparrow	FR14	2	0	19	2	0	19
Brewer's Sparrow	FR15	2	0	19	2	0	19

Brewer's Sparrow	FR16	2	0	18	4	2	21
Brewer's Sparrow	FR17	2	0	19	2	0	19
Brewer's Sparrow	FR18	2	0	19	2	0	19
Brewer's Sparrow	FR19	2	0	19	2	0	19
Brewer's Sparrow	FR2	2	0	22	2	0	22
Brewer's Sparrow	FR20	2	0	19	2	0	19
Brewer's Sparrow	FR21	2	0	18	2	0	19
Brewer's Sparrow	FR22	2	0	19	2	0	19
Brewer's Sparrow	FR3	2	0	22	2	0	22
Brewer's Sparrow	FR4	3	1	23	2	0	22
Brewer's Sparrow	FR5	2	0	20	2	0	20
Brewer's Sparrow	FR6	2	0	20	2	0	21
Brewer's Sparrow	FR7	1	0	19	1	0	19
Brewer's Sparrow	FR8	1	0	19	1	0	19
Brewer's Sparrow	FR9	1	0	19	1	0	19
Brewer's Sparrow	H1	1	0	10	1	0	10
Brewer's Sparrow	H10	1	0	10	1	0	10
Brewer's Sparrow	H11	1	0	10	1	0	11
Brewer's Sparrow	H12	1	0	10	1	0	10
Brewer's Sparrow	H13	1	0	10	2	1	11
Brewer's Sparrow	H14	1	0	10	1	0	10
Brewer's Sparrow	H15	1	0	10	1	0	11
Brewer's Sparrow	H16	1	0	10	1	0	10
Brewer's Sparrow	H17	1	0	10	1	0	10
Brewer's Sparrow	H18	1	0	10	1	0	10
Brewer's Sparrow	H19	1	0	11	2	1	11.025
Brewer's Sparrow	H2	1	0	11	1	0	10
Brewer's Sparrow	H20	2	1	11	1	0	10
Brewer's Sparrow	H21	1	0	11	1	0	10
Brewer's Sparrow	H22	1	0	10	1	0	11
Brewer's Sparrow	H23	1	0	10	1	0	10
Brewer's Sparrow	H24	1	0	10	1	0	10
Brewer's Sparrow	H25	1	0	10	1	0	10
Brewer's Sparrow	H3	1	0	10	1	0	10
Brewer's Sparrow	H4	1	0	10	1	0	10
Brewer's Sparrow	H5	1	0	10	1	0	10
Brewer's Sparrow	H6	1	0	10	1	0	10
Brewer's Sparrow	H7	1	0	10	1	0	10
Brewer's Sparrow	H8	1	0	10	1	0	10
Brewer's Sparrow	H9	1	0	10	1	0	10
Brewer's Sparrow	R1	4	0	39	4	0	39
Brewer's Sparrow	R10	4	0	39	4	0	39
Brewer's Sparrow	R11	4	0	39	4	0	39

Brewer's Sparrow	R12	4	0	39	4	0	39
Brewer's Sparrow	R13	4	0	39	4	0	39
Brewer's Sparrow	R14	4	0	39	7	3	42
Brewer's Sparrow	R15	5	1	40	4	0	39
Brewer's Sparrow	R16	4	0	39	4	0	39
Brewer's Sparrow	R17	4	0	39	4	0	39
Brewer's Sparrow	R18	4	0	39	4	0	38
Brewer's Sparrow	R19	4	0	39	4	0	39
Brewer's Sparrow	R2	4	0	39	4	0	38
Brewer's Sparrow	R20	6	2	41	4	0	39
Brewer's Sparrow	R21	4	0	39	4	0	39
Brewer's Sparrow	R22	4	0	39	4	0	39
Brewer's Sparrow	R23	4	0	39	4	0	39
Brewer's Sparrow	R24	4	0	39	4	0	39
Brewer's Sparrow	R25	5	1	40	4	0	39
Brewer's Sparrow	R3	4	0	39	5	1	40
Brewer's Sparrow	R4	4	0	39	4	0	39
Brewer's Sparrow	R5	6	2	42	5	1	41
Brewer's Sparrow	R6	4	0	38	4	0	39
Brewer's Sparrow	R7	4	0	39	4	0	39
Brewer's Sparrow	R8	4	0	39	4	0	39
Brewer's Sparrow	R9	4	0	39	4	0	39
Broad-tailed Hummingbird	F1	2	0	54	2	0	54
Broad-tailed Hummingbird	F10	2	0	50	2	0	50
Broad-tailed Hummingbird	F11	3	0	73	3	0	73
Broad-tailed Hummingbird	F12	3	0	77	3	0	77
Broad-tailed Hummingbird	F13	3	0	76.025	3	0	76.025
Broad-tailed Hummingbird	F14	2	1	27	1	0	26
Broad-tailed Hummingbird	F15	1	0	26	1	0	26
Broad-tailed Hummingbird	F16	1	0	26	1	0	26
Broad-tailed Hummingbird	F17	1	0	26	1	0	26
Broad-tailed Hummingbird	F18	1	0	26	1	0	26
Broad-tailed Hummingbird	F19	1	0	26	1	0	26
Broad-tailed Hummingbird	F2	3	0	74	5	1	76
Broad-tailed Hummingbird	F20	1	0	26	1	0	26
Broad-tailed Hummingbird	F21	1	0	26	1	0	26
Broad-tailed Hummingbird	F22	1	0	26	1	0	26
Broad-tailed Hummingbird	F3	3	0	74	4	1	75
Broad-tailed Hummingbird	F4	3	0	74	4	1	74
Broad-tailed Hummingbird	F5	3	0	67	4	1	68
Broad-tailed Hummingbird	F6	3	0	67	3	0	67
Broad-tailed Hummingbird	F7	4	0	99	4	0	100
Broad-tailed Hummingbird	F8	4	0	100	4	0	100

Broad-tailed Hummingbird	F9	4	0	100	5	1	101
Broad-tailed Hummingbird	FR1	2	0	46	2	0	46
Broad-tailed Hummingbird	FR10	2	0	46	2	0	46
Broad-tailed Hummingbird	FR11	3	0	57	3	0	57
Broad-tailed Hummingbird	FR12	3	0	61	4	1	62
Broad-tailed Hummingbird	FR13	4	1	63	3	0	61
Broad-tailed Hummingbird	FR14	1	0	25	1	0	25
Broad-tailed Hummingbird	FR15	1	0	25	1	0	25
Broad-tailed Hummingbird	FR16	1	0	25	1	0	25
Broad-tailed Hummingbird	FR17	1	0	25	1	0	25
Broad-tailed Hummingbird	FR18	1	0	25	1	0	25
Broad-tailed Hummingbird	FR19	1	0	25	1	0	25
Broad-tailed Hummingbird	FR2	3	0	57	3	0	57
Broad-tailed Hummingbird	FR20	1	0	25	1	0	25
Broad-tailed Hummingbird	FR21	1	0	25	1	0	25
Broad-tailed Hummingbird	FR22	1	0	25	1	0	25
Broad-tailed Hummingbird	FR3	3	0	57	3	0	57
Broad-tailed Hummingbird	FR4	3	0	56	3	0	57
Broad-tailed Hummingbird	FR5	3	0	58	4	1	59
Broad-tailed Hummingbird	FR6	3	0	59	3	0	58
Broad-tailed Hummingbird	FR7	4	0	78	4	0	78
Broad-tailed Hummingbird	FR8	5	1	80	5	1	80
Broad-tailed Hummingbird	FR9	5	1	80	4	0	79
Broad-tailed Hummingbird	H1	3	0	51	4	1	52
Broad-tailed Hummingbird	H10	3	0	51	4	1	52
Broad-tailed Hummingbird	H11	3	0	51	3	0	51
Broad-tailed Hummingbird	H12	3	0	51	3	0	51
Broad-tailed Hummingbird	H13	4	1	52	3	0	51
Broad-tailed Hummingbird	H14	4	1	52	3	0	51
Broad-tailed Hummingbird	H15	3	0	51	3	0	51
Broad-tailed Hummingbird	H16	3	0	51	4	1	52
Broad-tailed Hummingbird	H17	3	0	51	3	0	51
Broad-tailed Hummingbird	H18	3	0	51	3	0	51
Broad-tailed Hummingbird	H19	3	0	51	3	0	51
Broad-tailed Hummingbird	H2	3	0	51	3	0	51
Broad-tailed Hummingbird	H20	3	0	51	3	0	51
Broad-tailed Hummingbird	H21	3	0	51	3	0	51
Broad-tailed Hummingbird	H22	3	0	51.025	3	0	52
Broad-tailed Hummingbird	H23	3	0	51	3	0	51
Broad-tailed Hummingbird	H24	3	0	51	3	0	51
Broad-tailed Hummingbird	H25	4	1	52	3	0	51
Broad-tailed Hummingbird	H3	3	0	51	3	0	51
Broad-tailed Hummingbird	H4	3	0	51	3	0	51

Broad-tailed Hummingbird	H5	4	1	52	3	0	52
Broad-tailed Hummingbird	H6	3	0	51	3	0	51
Broad-tailed Hummingbird	H7	3	0	51	3	0	51
Broad-tailed Hummingbird	H8	3	0	51	4	1	52
Broad-tailed Hummingbird	H9	3	0	51	3	0	51
Broad-tailed Hummingbird	R1	3	0	63	3	0	63
Broad-tailed Hummingbird	R10	5	1	65	3	0	63
Broad-tailed Hummingbird	R11	3	0	63	3	0	63
Broad-tailed Hummingbird	R12	3	0	63	3	0	64
Broad-tailed Hummingbird	R13	4	1	65	3	0	64
Broad-tailed Hummingbird	R14	4	1	64	3	0	63
Broad-tailed Hummingbird	R15	3	0	63	3	0	63
Broad-tailed Hummingbird	R16	3	0	64	3	0	63
Broad-tailed Hummingbird	R17	3	0	64	4	1	64.025
Broad-tailed Hummingbird	R18	3	0	63	3	0	63
Broad-tailed Hummingbird	R19	3	0	63	3	0	63
Broad-tailed Hummingbird	R2	4	1	64	3	0	63
Broad-tailed Hummingbird	R20	3	0	63	3	0	63
Broad-tailed Hummingbird	R21	3	0	63	3	0	64
Broad-tailed Hummingbird	R22	3	0	64	3	0	63
Broad-tailed Hummingbird	R23	3	0	63	3	0	63
Broad-tailed Hummingbird	R24	3	0	63	3	0	63
Broad-tailed Hummingbird	R25	3	0	63	3	0	63
Broad-tailed Hummingbird	R3	6	1	66	5	1	65
Broad-tailed Hummingbird	R4	3	0	64	3	0	63
Broad-tailed Hummingbird	R5	3	0	63	3	0	63
Broad-tailed Hummingbird	R6	3	0	63	3	0	63
Broad-tailed Hummingbird	R7	3	0	63	3	0	63
Broad-tailed Hummingbird	R8	3	0	63	3	0	64
Broad-tailed Hummingbird	R9	3	0	63	3	0	63
<i>Empidonax</i> flycatcher	F1	0	0	1	0	0	1
<i>Empidonax</i> flycatcher	F10	0	0	3	0	0	3
<i>Empidonax</i> flycatcher	F11	0	0	1	0	0	1
<i>Empidonax</i> flycatcher	F12	0	0	1	0	0	1
<i>Empidonax</i> flycatcher	F13	0	0	1	0	0	1
<i>Empidonax</i> flycatcher	F14	0	0	2	0	0	2
<i>Empidonax</i> flycatcher	F15	0	0	2	0	0	2
<i>Empidonax</i> flycatcher	F16	0	0	2	0	0	2
<i>Empidonax</i> flycatcher	F17	1	1	3	0	0	2
<i>Empidonax</i> flycatcher	F18	0	0	2	0	0	2
<i>Empidonax</i> flycatcher	F19	1	1	3	0	0	2
<i>Empidonax</i> flycatcher	F2	0	0	1	0	0	1
<i>Empidonax</i> flycatcher	F20	0	0	2	0	0	2

<i>Empidonax flycatcher</i>	F21	0	0	2	0	0	2
<i>Empidonax flycatcher</i>	F22	0	0	2	0	0	2
<i>Empidonax flycatcher</i>	F3	1	1	2	0	0	1
<i>Empidonax flycatcher</i>	F4	0	0	1	0	0	1
<i>Empidonax flycatcher</i>	F5	0	0	2	0	0	1
<i>Empidonax flycatcher</i>	F6	0	0	1	2	2	4
<i>Empidonax flycatcher</i>	F7	0	0	1	0	0	1
<i>Empidonax flycatcher</i>	F8	0	0	1	0	0	1
<i>Empidonax flycatcher</i>	F9	0	0	1	0	0	1
<i>Empidonax flycatcher</i>	FR1	0	0	4	0	0	4
<i>Empidonax flycatcher</i>	FR10	3	1	15	5	2	17
<i>Empidonax flycatcher</i>	FR11	0	0	4	0	0	4
<i>Empidonax flycatcher</i>	FR12	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	FR13	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	FR14	4	1	12	6	3	16
<i>Empidonax flycatcher</i>	FR15	3	0	11	5	2	14
<i>Empidonax flycatcher</i>	FR16	2	0	11	4	1	13
<i>Empidonax flycatcher</i>	FR17	3	0	11	4	1	12
<i>Empidonax flycatcher</i>	FR18	6	3	14	4	1	13
<i>Empidonax flycatcher</i>	FR19	5	2	14	3	0	11
<i>Empidonax flycatcher</i>	FR2	2	1	5	0	0	4
<i>Empidonax flycatcher</i>	FR20	5	2	13	4	1	12
<i>Empidonax flycatcher</i>	FR21	6	2	15	4	1	13
<i>Empidonax flycatcher</i>	FR22	4	1	13	5	2	14
<i>Empidonax flycatcher</i>	FR3	0	0	4	0	0	4
<i>Empidonax flycatcher</i>	FR4	0	0	4	0	0	4
<i>Empidonax flycatcher</i>	FR5	1	0	6	1	0	6
<i>Empidonax flycatcher</i>	FR6	2	1	7	1	0	6
<i>Empidonax flycatcher</i>	FR7	0	0	3	1	1	4
<i>Empidonax flycatcher</i>	FR8	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	FR9	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H1	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H10	2	1	5	1	1	4
<i>Empidonax flycatcher</i>	H11	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H12	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H13	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H14	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H15	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H16	0	0	3	2	2	5
<i>Empidonax flycatcher</i>	H17	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H18	0	0	3	2	2	5
<i>Empidonax flycatcher</i>	H19	0	0	3	3	2	5
<i>Empidonax flycatcher</i>	H2	0	0	3	3	2	6

<i>Empidonax flycatcher</i>	H20	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H21	0	0	3	1	1	4
<i>Empidonax flycatcher</i>	H22	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H23	0	0	3	1	1	4
<i>Empidonax flycatcher</i>	H24	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H25	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H3	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H4	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H5	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H6	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H7	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H8	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	H9	0	0	3	0	0	3
<i>Empidonax flycatcher</i>	R1	1	0	5	1	0	5
<i>Empidonax flycatcher</i>	R10	1	0	5	3	1	7
<i>Empidonax flycatcher</i>	R11	1	0	5	1	0	5
<i>Empidonax flycatcher</i>	R12	1	0	5	3	1	7
<i>Empidonax flycatcher</i>	R13	1	0	5	1	0	5
<i>Empidonax flycatcher</i>	R14	1	0	5	1	0	5
<i>Empidonax flycatcher</i>	R15	1	0	5	2	1	6
<i>Empidonax flycatcher</i>	R16	3	1	8	2	1	6
<i>Empidonax flycatcher</i>	R17	1	0	5	1	0	5
<i>Empidonax flycatcher</i>	R18	2	1	6	1	0	5
<i>Empidonax flycatcher</i>	R19	1	0	5	2	1	6
<i>Empidonax flycatcher</i>	R2	2	1	6	3	1	8
<i>Empidonax flycatcher</i>	R20	1	0	5	2	1	6
<i>Empidonax flycatcher</i>	R21	1	0	5	2	1	6
<i>Empidonax flycatcher</i>	R22	1	0	5	3	1	7
<i>Empidonax flycatcher</i>	R23	1	0	5	1	0	5
<i>Empidonax flycatcher</i>	R24	2	1	6	3	2	8
<i>Empidonax flycatcher</i>	R25	3	2	8	2	1	6
<i>Empidonax flycatcher</i>	R3	1	0	5	2	1	6
<i>Empidonax flycatcher</i>	R4	1	0	5	3	1	8
<i>Empidonax flycatcher</i>	R5	1	0	5	1	0	5
<i>Empidonax flycatcher</i>	R6	2	1	6	1	0	5
<i>Empidonax flycatcher</i>	R7	1	0	5	2	1	6
<i>Empidonax flycatcher</i>	R8	2	1	6	1	0	5
<i>Empidonax flycatcher</i>	R9	1	0	5	2	1	6
Green-tailed Towhee	F1	0	0	2	0	0	2
Green-tailed Towhee	F10	0	0	2	0	0	2
Green-tailed Towhee	F11	1	0	4	1	0	4
Green-tailed Towhee	F12	1	0	4	1	0	4
Green-tailed Towhee	F13	1	0	4	3	2	6

Green-tailed Towhee	F14	0	0	2	0	0	2
Green-tailed Towhee	F15	0	0	2	0	0	2
Green-tailed Towhee	F16	0	0	2	0	0	2
Green-tailed Towhee	F17	0	0	2	0	0	2
Green-tailed Towhee	F18	0	0	2	0	0	2
Green-tailed Towhee	F19	0	0	2	0	0	2
Green-tailed Towhee	F2	2	1	5	1	0	4
Green-tailed Towhee	F20	0	0	2	1	1	4
Green-tailed Towhee	F21	0	0	2	0	0	2
Green-tailed Towhee	F22	0	0	2	0	0	2
Green-tailed Towhee	F3	1	0	4	4	3	8
Green-tailed Towhee	F4	1	0	4	2	1	5
Green-tailed Towhee	F5	0	0	3	1	1	4
Green-tailed Towhee	F6	0	0	3	2	1	4
Green-tailed Towhee	F7	2	1	6	3	1	7
Green-tailed Towhee	F8	2	1	6	2	1	7
Green-tailed Towhee	F9	1	0	5	1	0	5
Green-tailed Towhee	FR1	0	0	3	0	0	3
Green-tailed Towhee	FR10	0	0	3	0	0	3
Green-tailed Towhee	FR11	2	0	8	2	0	8
Green-tailed Towhee	FR12	1	0	7	3	1	8
Green-tailed Towhee	FR13	1	0	7	6	3	12
Green-tailed Towhee	FR14	1	0	4	2	1	5
Green-tailed Towhee	FR15	1	0	4	1	0	4
Green-tailed Towhee	FR16	2	1	5	1	0	4
Green-tailed Towhee	FR17	2	1	5	2	1	5
Green-tailed Towhee	FR18	2	1	5	1	0	4
Green-tailed Towhee	FR19	1	0	4	4	3	7
Green-tailed Towhee	FR2	3	1	9	2	0	8
Green-tailed Towhee	FR20	1	0	4	2	1	5
Green-tailed Towhee	FR21	2	1	5	2	1	5
Green-tailed Towhee	FR22	1	0	4	1	0	4
Green-tailed Towhee	FR3	2	0	8	2	0	8
Green-tailed Towhee	FR4	7	4	14	7	3	14
Green-tailed Towhee	FR5	1	0	5	2	1	6
Green-tailed Towhee	FR6	1	0	5	2	1	6
Green-tailed Towhee	FR7	3	1	11	5	2	12
Green-tailed Towhee	FR8	3	1	11	6	3	13
Green-tailed Towhee	FR9	2	0	10	4	2	12
Green-tailed Towhee	H1	0	0	2	0	0	2
Green-tailed Towhee	H10	0	0	2	0	0	2
Green-tailed Towhee	H11	0	0	2	0	0	2
Green-tailed Towhee	H12	0	0	2	0	0	2

Green-tailed Towhee	H13	0	0	2	3	3	5
Green-tailed Towhee	H14	0	0	2	0	0	2
Green-tailed Towhee	H15	0	0	2	1	1	3
Green-tailed Towhee	H16	0	0	2	0	0	2
Green-tailed Towhee	H17	0	0	2	0	0	2
Green-tailed Towhee	H18	0	0	2	0	0	2
Green-tailed Towhee	H19	0	0	2	1	1	3
Green-tailed Towhee	H2	0	0	2	1	1	3
Green-tailed Towhee	H20	0	0	2	0	0	2
Green-tailed Towhee	H21	0	0	2	0	0	2
Green-tailed Towhee	H22	0	0	2	0	0	2
Green-tailed Towhee	H23	0	0	2	0	0	2
Green-tailed Towhee	H24	0	0	2	0	0	2
Green-tailed Towhee	H25	1	1	3	1	1	3
Green-tailed Towhee	H3	0	0	2	0	0	2
Green-tailed Towhee	H4	0	0	2	0	0	2
Green-tailed Towhee	H5	1	1	3	1	1	3
Green-tailed Towhee	H6	0	0	2	0	0	2
Green-tailed Towhee	H7	0	0	2	0	0	2
Green-tailed Towhee	H8	0	0	2	0	0	2
Green-tailed Towhee	H9	0	0	2	0	0	2
Green-tailed Towhee	R1	0	0	2	0	0	2
Green-tailed Towhee	R10	0	0	2	1	1	3
Green-tailed Towhee	R11	0	0	2	0	0	2
Green-tailed Towhee	R12	0	0	2	0	0	2
Green-tailed Towhee	R13	0	0	2	0	0	2
Green-tailed Towhee	R14	1	1	3	3	2	5
Green-tailed Towhee	R15	0	0	2	0	0	2
Green-tailed Towhee	R16	1	1	3	0	0	2
Green-tailed Towhee	R17	0	0	2	0	0	2
Green-tailed Towhee	R18	0	0	2	0	0	2
Green-tailed Towhee	R19	0	0	2	2	1	4
Green-tailed Towhee	R2	0	0	2	2	2	5
Green-tailed Towhee	R20	0	0	2	0	0	2
Green-tailed Towhee	R21	0	0	2	0	0	2
Green-tailed Towhee	R22	0	0	2	1	1	3
Green-tailed Towhee	R23	0	0	2	0	0	2
Green-tailed Towhee	R24	0	0	2	1	1	3
Green-tailed Towhee	R25	0	0	2	0	0	2
Green-tailed Towhee	R3	0	0	2	2	1	4
Green-tailed Towhee	R4	0	0	2	0	0	2
Green-tailed Towhee	R5	0	0	2	0	0	2
Green-tailed Towhee	R6	0	0	2	0	0	2

Green-tailed Towhee	R7	0	0	2	0	0	3
Green-tailed Towhee	R8	0	0	2	0	0	2
Green-tailed Towhee	R9	1	1	3	0	0	2
Hairy Woodpecker	F1	4	1	21	3	1	20
Hairy Woodpecker	F10	1	0	10	1	0	10
Hairy Woodpecker	F11	0	0	3	0	0	3
Hairy Woodpecker	F12	2	1	16	2	0	15
Hairy Woodpecker	F13	4	2	18	2	0	15
Hairy Woodpecker	F14	0	0	3	0	0	3
Hairy Woodpecker	F15	0	0	3	0	0	3
Hairy Woodpecker	F16	0	0	3	0	0	3
Hairy Woodpecker	F17	0	0	3	0	0	3
Hairy Woodpecker	F18	0	0	3	0	0	3
Hairy Woodpecker	F19	0	0	3	0	0	3
Hairy Woodpecker	F2	0	0	3	0	0	3
Hairy Woodpecker	F20	0	0	3	0	0	3
Hairy Woodpecker	F21	0	0	3	0	0	3
Hairy Woodpecker	F22	0	0	3	0	0	3
Hairy Woodpecker	F3	0	0	3	0	0	3
Hairy Woodpecker	F4	0	0	3	0	0	3
Hairy Woodpecker	F5	0	0	6	1	1	7
Hairy Woodpecker	F6	0	0	6	0	0	6
Hairy Woodpecker	F7	1	0	8	1	0	8
Hairy Woodpecker	F8	2	1	9	1	0	8
Hairy Woodpecker	F9	1	0	8	2	1	10
Hairy Woodpecker	FR1	1	0	7	1	0	7
Hairy Woodpecker	FR10	0	0	4	1	1	5
Hairy Woodpecker	FR11	0	0	1	0	0	1
Hairy Woodpecker	FR12	0	0	6	1	0	6
Hairy Woodpecker	FR13	0	0	6	1	0	6
Hairy Woodpecker	FR14	0	0	1	0	0	1
Hairy Woodpecker	FR15	0	0	1	0	0	1
Hairy Woodpecker	FR16	0	0	1	0	0	1
Hairy Woodpecker	FR17	0	0	1	0	0	1
Hairy Woodpecker	FR18	0	0	1	0	0	1
Hairy Woodpecker	FR19	0	0	1	1	1	2
Hairy Woodpecker	FR2	0	0	1	0	0	1
Hairy Woodpecker	FR20	0	0	1	0	0	1
Hairy Woodpecker	FR21	0	0	1	0	0	1
Hairy Woodpecker	FR22	0	0	1	0	0	1
Hairy Woodpecker	FR3	0	0	1	0	0	1
Hairy Woodpecker	FR4	0	0	1	0	0	1
Hairy Woodpecker	FR5	0	0	3	0	0	3

Hairy Woodpecker	FR6	0	0	3	0	0	3
Hairy Woodpecker	FR7	0	0	3	0	0	3
Hairy Woodpecker	FR8	0	0	3	0	0	3
Hairy Woodpecker	FR9	0	0	3	0	0	3
Hairy Woodpecker	H1	0	0	3	0	0	3
Hairy Woodpecker	H10	0	0	3	0	0	3
Hairy Woodpecker	H11	0	0	3	0	0	3
Hairy Woodpecker	H12	0	0	3	0	0	3
Hairy Woodpecker	H13	0	0	3	0	0	3
Hairy Woodpecker	H14	0	0	3	0	0	3
Hairy Woodpecker	H15	0	0	3	0	0	3
Hairy Woodpecker	H16	0	0	3	0	0	3
Hairy Woodpecker	H17	0	0	3	0	0	3
Hairy Woodpecker	H18	0	0	3	1	1	4
Hairy Woodpecker	H19	0	0	3	0	0	3
Hairy Woodpecker	H2	0	0	3	1	1	4
Hairy Woodpecker	H20	0	0	3	0	0	3
Hairy Woodpecker	H21	0	0	3	0	0	3
Hairy Woodpecker	H22	0	0	3	2	2	6
Hairy Woodpecker	H23	0	0	3	0	0	3
Hairy Woodpecker	H24	0	0	3	0	0	3
Hairy Woodpecker	H25	0	0	3	0	0	3
Hairy Woodpecker	H3	0	0	3	0	0	3
Hairy Woodpecker	H4	0	0	3	0	0	3
Hairy Woodpecker	H5	0	0	3	0	0	3
Hairy Woodpecker	H6	0	0	3	2	2	6
Hairy Woodpecker	H7	0	0	3	0	0	3
Hairy Woodpecker	H8	0	0	3	0	0	3
Hairy Woodpecker	H9	0	0	3	0	0	3
Hairy Woodpecker	R1	0	0	3	0	0	3
Hairy Woodpecker	R10	0	0	3	0	0	3
Hairy Woodpecker	R11	1	1	4	3	3	7
Hairy Woodpecker	R12	0	0	3	0	0	3
Hairy Woodpecker	R13	0	0	3	0	0	3
Hairy Woodpecker	R14	0	0	3	0	0	3
Hairy Woodpecker	R15	0	0	3	0	0	3
Hairy Woodpecker	R16	0	0	3	0	0	3
Hairy Woodpecker	R17	0	0	3	0	0	3
Hairy Woodpecker	R18	0	0	3	0	0	3
Hairy Woodpecker	R19	0	0	3	0	0	3
Hairy Woodpecker	R2	0	0	3	0	0	3
Hairy Woodpecker	R20	0	0	3	0	0	3
Hairy Woodpecker	R21	0	0	3	0	0	3

Hairy Woodpecker	R22	0	0	3	0	0	3
Hairy Woodpecker	R23	0	0	3	0	0	3
Hairy Woodpecker	R24	0	0	3	1	1	4
Hairy Woodpecker	R25	0	0	3	0	0	3
Hairy Woodpecker	R3	0	0	3	0	0	3
Hairy Woodpecker	R4	0	0	3	0	0	3
Hairy Woodpecker	R5	0	0	3	0	0	3
Hairy Woodpecker	R6	0	0	3	0	0	3
Hairy Woodpecker	R7	0	0	3	0	0	3
Hairy Woodpecker	R8	0	0	3	0	0	3
Hairy Woodpecker	R9	0	0	3	0	0	3
House Wren	F1	0	0	2	0	0	2
House Wren	F10	0	0	1	0	0	2
House Wren	F11	0	0	1	0	0	1
House Wren	F12	0	0	2	0	0	2
House Wren	F13	0	0	2	1	1	3
House Wren	F14	0	0	1	0	0	1
House Wren	F15	0	0	1	0	0	1
House Wren	F16	0	0	1	0	0	1
House Wren	F17	0	0	1	0	0	1
House Wren	F18	0	0	1	0	0	1
House Wren	F19	0	0	1	0	0	1
House Wren	F2	0	0	1	0	0	1
House Wren	F20	0	0	1	0	0	1
House Wren	F21	0	0	1	0	0	1
House Wren	F22	0	0	1	0	0	1
House Wren	F3	0	0	1	0	0	1
House Wren	F4	0	0	1	0	0	1
House Wren	F5	1	1	5	1	1	6
House Wren	F6	0	0	4	0	0	4
House Wren	F7	0	0	2	0	0	2
House Wren	F8	0	0	2	0	0	2
House Wren	F9	0	0	2	0	0	2
House Wren	FR1	0	0	3	0	0	3
House Wren	FR10	0	0	2	0	0	2
House Wren	FR11	0	0	2	0	0	2
House Wren	FR12	0	0	3	0	0	2
House Wren	FR13	0	0	3	0	0	2
House Wren	FR14	0	0	1	0	0	1
House Wren	FR15	0	0	1	0	0	1
House Wren	FR16	0	0	1	0	0	1
House Wren	FR17	0	0	1	0	0	1
House Wren	FR18	0	0	1	0	0	1

House Wren	FR19	0	0	1	0	0	1
House Wren	FR2	0	0	2	0	0	2
House Wren	FR20	0	0	1	0	0	1
House Wren	FR21	0	0	1	0	0	1
House Wren	FR22	1	1	2	0	0	1
House Wren	FR3	0	0	2	0	0	2
House Wren	FR4	0	0	2	0	0	2
House Wren	FR5	0	0	5	0	0	5
House Wren	FR6	0	0	5	0	0	5
House Wren	FR7	0	0	3	0	0	3
House Wren	FR8	1	1	4	1	1	4
House Wren	FR9	0	0	3	0	0	3
House Wren	H1	0	0	2	0	0	2
House Wren	H10	0	0	2	1	1	3
House Wren	H11	0	0	2	0	0	2
House Wren	H12	0	0	2	1	1	3
House Wren	H13	0	0	2	0	0	2
House Wren	H14	0	0	2	0	0	2
House Wren	H15	0	0	2	0	0	2
House Wren	H16	0	0	2	0	0	2
House Wren	H17	0	0	2	0	0	2
House Wren	H18	0	0	2	0	0	2
House Wren	H19	0	0	2	0	0	2
House Wren	H2	0	0	2	0	0	2
House Wren	H20	0	0	2	0	0	2
House Wren	H21	0	0	2	0	0	2
House Wren	H22	0	0	2	0	0	2
House Wren	H23	0	0	2	0	0	2
House Wren	H24	0	0	2	0	0	2
House Wren	H25	0	0	2	0	0	2
House Wren	H3	0	0	2	0	0	2
House Wren	H4	0	0	2	0	0	2
House Wren	H5	0	0	2	0	0	2
House Wren	H6	0	0	2	0	0	2
House Wren	H7	0	0	2	3	3	5
House Wren	H8	0	0	2	0	0	2
House Wren	H9	0	0	2	0	0	2
House Wren	R1	0	0	3	0	0	3
House Wren	R10	0	0	3	0	0	3
House Wren	R11	0	0	3	0	0	3
House Wren	R12	0	0	3	0	0	2
House Wren	R13	0	0	3	0	0	2
House Wren	R14	0	0	3	1	1	4

House Wren	R15	0	0	3	0	0	2
House Wren	R16	0	0	3	0	0	3
House Wren	R17	0	0	3	0	0	3
House Wren	R18	0	0	3	0	0	2
House Wren	R19	0	0	3	0	0	3
House Wren	R2	0	0	3	0	0	2
House Wren	R20	0	0	3	0	0	3
House Wren	R21	0	0	3	0	0	2
House Wren	R22	2	2	6	2	2	6
House Wren	R23	0	0	3	0	0	3
House Wren	R24	0	0	3	0	0	2
House Wren	R25	0	0	3	0	0	3
House Wren	R3	0	0	3	0	0	3
House Wren	R4	0	0	3	0	0	2
House Wren	R5	0	0	2	0	0	2
House Wren	R6	1	1	3	0	0	3
House Wren	R7	0	0	3	0	0	2
House Wren	R8	0	0	3	0	0	3
House Wren	R9	0	0	3	0	0	2
Lark Sparrow	F1	0	0	2	0	0	2
Lark Sparrow	F10	0	0	2	0	0	2
Lark Sparrow	F11	0	0	10	0	0	10
Lark Sparrow	F12	0	0	1	0	0	1
Lark Sparrow	F13	0	0	1	0	0	1
Lark Sparrow	F14	1	0	27	3	2	30
Lark Sparrow	F15	1	0	28	1	0	27
Lark Sparrow	F16	1	0	27	1	0	27
Lark Sparrow	F17	1	0	27	1	0	28
Lark Sparrow	F18	1	0	28	2	1	28
Lark Sparrow	F19	3	2	30	2	1	29
Lark Sparrow	F2	0	0	10	0	0	10
Lark Sparrow	F20	2	1	28	4	2	32
Lark Sparrow	F21	1	0	28	3	2	31
Lark Sparrow	F22	2	1	29	1	0	27
Lark Sparrow	F3	4	4	15	0	0	10
Lark Sparrow	F4	0	0	10	0	0	10
Lark Sparrow	F5	0	0	1	0	0	1
Lark Sparrow	F6	0	0	1	0	0	1
Lark Sparrow	F7	0	0	1	0	0	1
Lark Sparrow	F8	0	0	1	0	0	1
Lark Sparrow	F9	0	0	1	0	0	1
Lark Sparrow	FR1	0	0	1	0	0	1
Lark Sparrow	FR10	0	0	1	0	0	1

Lark Sparrow	FR11	0	0	6	0	0	6
Lark Sparrow	FR12	0	0	1	0	0	1
Lark Sparrow	FR13	0	0	1	0	0	1
Lark Sparrow	FR14	0	0	17	4	3	21
Lark Sparrow	FR15	0	0	18	0	0	17
Lark Sparrow	FR16	0	0	17	4	3	21
Lark Sparrow	FR17	0	0	17	0	0	17
Lark Sparrow	FR18	1	1	18	2	1	20
Lark Sparrow	FR19	0	0	17	0	0	17
Lark Sparrow	FR2	0	0	6	0	0	6
Lark Sparrow	FR20	0	0	18	0	0	17
Lark Sparrow	FR21	0	0	17	0	0	17
Lark Sparrow	FR22	0	0	17	0	0	17
Lark Sparrow	FR3	0	0	6	0	0	6
Lark Sparrow	FR4	0	0	6	0	0	6
Lark Sparrow	FR5	0	0	1	0	0	1
Lark Sparrow	FR6	0	0	1	0	0	1
Lark Sparrow	FR7	0	0	1	0	0	1
Lark Sparrow	FR8	0	0	1	0	0	1
Lark Sparrow	FR9	0	0	1	0	0	1
Lark Sparrow	H1	0	0	0	0	0	0
Lark Sparrow	H10	0	0	0	0	0	0
Lark Sparrow	H11	0	0	0	0	0	0
Lark Sparrow	H12	0	0	0	0	0	0
Lark Sparrow	H13	0	0	0	0	0	0
Lark Sparrow	H14	0	0	0	0	0	0
Lark Sparrow	H15	0	0	0	0	0	0
Lark Sparrow	H16	0	0	0	0	0	0
Lark Sparrow	H17	0	0	0	0	0	0
Lark Sparrow	H18	0	0	0	0	0	0
Lark Sparrow	H19	0	0	0	0	0	0
Lark Sparrow	H2	0	0	0	0	0	0
Lark Sparrow	H20	0	0	0	0	0	0
Lark Sparrow	H21	0	0	0	0	0	0
Lark Sparrow	H22	0	0	0	0	0	0
Lark Sparrow	H23	0	0	0	0	0	0
Lark Sparrow	H24	0	0	0	0	0	0
Lark Sparrow	H25	0	0	0	0	0	0
Lark Sparrow	H3	0	0	0	0	0	0
Lark Sparrow	H4	0	0	0	0	0	0
Lark Sparrow	H5	0	0	0	0	0	0
Lark Sparrow	H6	0	0	0	0	0	0
Lark Sparrow	H7	0	0	0	0	0	0

Lark Sparrow	H8	0	0	0	0	0	0
Lark Sparrow	H9	0	0	0	0	0	0
Lark Sparrow	R1	0	0	1	0	0	1
Lark Sparrow	R10	0	0	1	0	0	1
Lark Sparrow	R11	0	0	1	0	0	1
Lark Sparrow	R12	0	0	1	0	0	1
Lark Sparrow	R13	0	0	1	0	0	1
Lark Sparrow	R14	0	0	1	0	0	1
Lark Sparrow	R15	0	0	1	0	0	1
Lark Sparrow	R16	0	0	1	0	0	1
Lark Sparrow	R17	0	0	1	0	0	1
Lark Sparrow	R18	0	0	1	0	0	1
Lark Sparrow	R19	0	0	1	0	0	1
Lark Sparrow	R2	0	0	1	0	0	1
Lark Sparrow	R20	0	0	1	2	2	3
Lark Sparrow	R21	0	0	1	0	0	1
Lark Sparrow	R22	0	0	1	0	0	1
Lark Sparrow	R23	0	0	1	0	0	1
Lark Sparrow	R24	0	0	1	0	0	1
Lark Sparrow	R25	0	0	1	0	0	1
Lark Sparrow	R3	0	0	1	0	0	1
Lark Sparrow	R4	0	0	1	0	0	1
Lark Sparrow	R5	0	0	1	0	0	1
Lark Sparrow	R6	0	0	1	0	0	1
Lark Sparrow	R7	0	0	1	0	0	1
Lark Sparrow	R8	0	0	1	0	0	1
Lark Sparrow	R9	0	0	1	0	0	1
Mountain Bluebird	F1	15	2	262	14	1	260
Mountain Bluebird	F10	15	2	270	14	1	269
Mountain Bluebird	F11	14	1	260	16	3	263
Mountain Bluebird	F12	17	3	283	15	1	281
Mountain Bluebird	F13	16	2	281	17	3	283
Mountain Bluebird	F14	20	4	311	17	1	308
Mountain Bluebird	F15	17	1	308	17	1	308
Mountain Bluebird	F16	17	1	308	21	5	312
Mountain Bluebird	F17	17	1	308	17	1	307
Mountain Bluebird	F18	17	1	308	21	5	312
Mountain Bluebird	F19	19	3	310	17	1	307
Mountain Bluebird	F2	14	1	261	14	1	261
Mountain Bluebird	F20	17	1	307	17	1	308
Mountain Bluebird	F21	22	6	313	17	1	308
Mountain Bluebird	F22	19	3	310	17	1	308
Mountain Bluebird	F3	19	6	266	15	2	262

Mountain Bluebird	F4	14	1	261	15	2	262
Mountain Bluebird	F5	16	2	292	18	4	295
Mountain Bluebird	F6	15	1	292	16	2	294
Mountain Bluebird	F7	19	2	363	20	3	364
Mountain Bluebird	F8	19	2	364	24	6	368
Mountain Bluebird	F9	20	3	363	21	4	365
Mountain Bluebird	FR1	5	0	100	5	0	100
Mountain Bluebird	FR10	5	0	101	5	0	101
Mountain Bluebird	FR11	6	0	101	6	0	101
Mountain Bluebird	FR12	6	0	110	6	0	109
Mountain Bluebird	FR13	6	0	109	6	0	110
Mountain Bluebird	FR14	7	0	116	7	0	116
Mountain Bluebird	FR15	7	0	116	7	0	116
Mountain Bluebird	FR16	7	0	116	8	1	117
Mountain Bluebird	FR17	8	1	117	7	0	116
Mountain Bluebird	FR18	9	1	118	7	0	116
Mountain Bluebird	FR19	8	1	117	8	1	117
Mountain Bluebird	FR2	6	0	102	6	0	102
Mountain Bluebird	FR20	7	0	116	9	2	118
Mountain Bluebird	FR21	9	2	118	9	1	118
Mountain Bluebird	FR22	8	1	117	7	0	116
Mountain Bluebird	FR3	6	0	102	6	0	101
Mountain Bluebird	FR4	6	0	101	6	0	102
Mountain Bluebird	FR5	6	0	114	6	0	114
Mountain Bluebird	FR6	6	0	114	6	0	113
Mountain Bluebird	FR7	7	0	139	9	2	141
Mountain Bluebird	FR8	8	1	140	7	0	138
Mountain Bluebird	FR9	7	0	139	9	2	141
Mountain Bluebird	H1	10	0	178	14	4	183
Mountain Bluebird	H10	10	0	178	10	0	178
Mountain Bluebird	H11	12	2	180	12	2	180
Mountain Bluebird	H12	10	0	178	10	0	178
Mountain Bluebird	H13	10	0	177	10	0	178
Mountain Bluebird	H14	11	1	179	10	0	178
Mountain Bluebird	H15	11	1	179	10	0	178
Mountain Bluebird	H16	11	1	179	11	1	178
Mountain Bluebird	H17	11	1	179	10	0	178
Mountain Bluebird	H18	10	0	178	15	4	183
Mountain Bluebird	H19	12	2	180	12	2	180
Mountain Bluebird	H2	10	0	178	10	0	178
Mountain Bluebird	H20	10	0	178	14	3	182
Mountain Bluebird	H21	10	0	177	10	0	178
Mountain Bluebird	H22	10	0	178	11	1	178

Mountain Bluebird	H23	12	2	180	10	0	178
Mountain Bluebird	H24	10	0	178	10	0	178
Mountain Bluebird	H25	13	3	181	10	0	178
Mountain Bluebird	H3	10	0	178	10	0	178
Mountain Bluebird	H4	10	0	178	10	0	178
Mountain Bluebird	H5	10	0	178	10	0	178
Mountain Bluebird	H6	10	0	178	13	3	181
Mountain Bluebird	H7	10	0	178	10	0	178
Mountain Bluebird	H8	10	0	178	10	0	178
Mountain Bluebird	H9	10	0	178	10	0	178
Mountain Bluebird	R1	6	0	113	7	1	115
Mountain Bluebird	R10	6	0	114	7	1	114
Mountain Bluebird	R11	6	0	114	6	0	113
Mountain Bluebird	R12	7	1	115	6	0	114
Mountain Bluebird	R13	6	0	113	7	1	115
Mountain Bluebird	R14	6	0	113	6	0	113
Mountain Bluebird	R15	6	0	114	6	0	113
Mountain Bluebird	R16	6	0	114	6	0	113
Mountain Bluebird	R17	6	0	114	6	0	114
Mountain Bluebird	R18	6	0	114	6	0	114
Mountain Bluebird	R19	6	0	113	7	1	115
Mountain Bluebird	R2	8	1	115	6	0	113
Mountain Bluebird	R20	6	0	114	6	0	113
Mountain Bluebird	R21	6	0	113	6	0	114
Mountain Bluebird	R22	10	3	118	6	0	114
Mountain Bluebird	R23	6	0	113	6	0	113
Mountain Bluebird	R24	6	0	113	8	2	115
Mountain Bluebird	R25	6	0	113	8	2	115
Mountain Bluebird	R3	6	0	113	6	0	114
Mountain Bluebird	R4	6	0	114	6	0	114
Mountain Bluebird	R5	6	0	113	6	0	114
Mountain Bluebird	R6	8	2	116	6	0	114
Mountain Bluebird	R7	6	0	113	6	0	113
Mountain Bluebird	R8	9	3	117	8	2	116
Mountain Bluebird	R9	6	0	113	6	0	113
Mourning Dove	F1	2	1	6	0	0	4
Mourning Dove	F10	0	0	3	0	0	3
Mourning Dove	F11	0	0	2	0	0	2
Mourning Dove	F12	2	1	7	3	1	8
Mourning Dove	F13	1	0	6	1	0	7
Mourning Dove	F14	1	0	6	1	0	5
Mourning Dove	F15	1	0	6	1	0	6
Mourning Dove	F16	1	0	5	1	0	5

Mourning Dove	F17	1	0	6	3	2	8
Mourning Dove	F18	1	0	6	4	3	9
Mourning Dove	F19	1	0	6	2	1	7
Mourning Dove	F2	1	1	3	0	0	2
Mourning Dove	F20	1	0	5	2	1	7
Mourning Dove	F21	1	0	6	1	0	6
Mourning Dove	F22	1	0	6	1	0	6
Mourning Dove	F3	0	0	2	0	0	2
Mourning Dove	F4	0	0	2	0	0	2
Mourning Dove	F5	0	0	3	0	0	3
Mourning Dove	F6	0	0	3	0	0	3
Mourning Dove	F7	0	0	3	0	0	3
Mourning Dove	F8	0	0	3	0	0	3
Mourning Dove	F9	0	0	3	0	0	3
Mourning Dove	FR1	1	0	6	1	0	6
Mourning Dove	FR10	0	0	4	0	0	4
Mourning Dove	FR11	0	0	3	0	0	3
Mourning Dove	FR12	2	0	9	2	0	9
Mourning Dove	FR13	4	2	11	3	1	10
Mourning Dove	FR14	2	0	8	2	0	8
Mourning Dove	FR15	3	1	9	2	0	8
Mourning Dove	FR16	3	1	9	2	0	8
Mourning Dove	FR17	3	1	9	2	0	8
Mourning Dove	FR18	4	2	10	3	1	9
Mourning Dove	FR19	6	3	12	2	0	8
Mourning Dove	FR2	0	0	3	0	0	3
Mourning Dove	FR20	4	2	10	3	1	9
Mourning Dove	FR21	2	0	8	2	0	8
Mourning Dove	FR22	2	0	8	2	0	8
Mourning Dove	FR3	0	0	3	0	0	3
Mourning Dove	FR4	0	0	3	0	0	3
Mourning Dove	FR5	1	0	4	1	0	4
Mourning Dove	FR6	2	1	5	2	1	5
Mourning Dove	FR7	0	0	4	1	0	4
Mourning Dove	FR8	0	0	4	2	2	6
Mourning Dove	FR9	0	0	4	1	0	4
Mourning Dove	H1	0	0	2	0	0	2
Mourning Dove	H10	1	1	3	0	0	2
Mourning Dove	H11	0	0	2	0	0	2
Mourning Dove	H12	0	0	2	1	1	3
Mourning Dove	H13	0	0	2	0	0	2
Mourning Dove	H14	0	0	2	0	0	2
Mourning Dove	H15	0	0	2	0	0	2

Mourning Dove	H16	0	0	2	0	0	2
Mourning Dove	H17	0	0	2	0	0	2
Mourning Dove	H18	0	0	2	0	0	2
Mourning Dove	H19	0	0	2	0	0	2
Mourning Dove	H2	0	0	2	0	0	2
Mourning Dove	H20	0	0	2	0	0	2
Mourning Dove	H21	0	0	2	0	0	2
Mourning Dove	H22	0	0	2	0	0	2
Mourning Dove	H23	0	0	2	0	0	2
Mourning Dove	H24	0	0	2	0	0	2
Mourning Dove	H25	1	1	3	0	0	2
Mourning Dove	H3	1	1	3	2	2	4
Mourning Dove	H4	0	0	2	0	0	2
Mourning Dove	H5	0	0	2	0	0	2
Mourning Dove	H6	0	0	2	0	0	2
Mourning Dove	H7	0	0	2	0	0	2
Mourning Dove	H8	0	0	2	0	0	2
Mourning Dove	H9	0	0	2	0	0	2
Mourning Dove	R1	1	0	4	1	0	4
Mourning Dove	R10	1	0	4	1	0	4
Mourning Dove	R11	1	0	4	1	0	4
Mourning Dove	R12	1	0	4	1	0	4
Mourning Dove	R13	1	0	4	1	0	4
Mourning Dove	R14	1	0	4	1	0	4
Mourning Dove	R15	3	2	7	1	0	4
Mourning Dove	R16	3	2	6	1	0	4
Mourning Dove	R17	1	0	4	1	0	4
Mourning Dove	R18	1	0	4	2	1	5
Mourning Dove	R19	1	0	4	1	0	4
Mourning Dove	R2	1	0	4	1	0	4
Mourning Dove	R20	2	1	5	1	0	4
Mourning Dove	R21	1	0	4	1	0	4
Mourning Dove	R22	5	3	9	1	0	4
Mourning Dove	R23	1	0	4	1	0	4
Mourning Dove	R24	1	0	4	1	0	4
Mourning Dove	R25	1	0	4	1	0	4
Mourning Dove	R3	1	0	4	1	0	4
Mourning Dove	R4	1	0	4	1	0	4
Mourning Dove	R5	1	0	4	1	0	4
Mourning Dove	R6	2	1	5	1	0	4
Mourning Dove	R7	1	0	4	2	1	5
Mourning Dove	R8	2	1	5	1	0	4
Mourning Dove	R9	2	1	6	1	0	4

Spotted Towhee	F1	0	0	1	1	1	2
Spotted Towhee	F10	0	0	1	0	0	1
Spotted Towhee	F11	0	0	2	0	0	2
Spotted Towhee	F12	0	0	2	0	0	2
Spotted Towhee	F13	0	0	2	0	0	2
Spotted Towhee	F14	0	0	1	0	0	1
Spotted Towhee	F15	0	0	1	0	0	1
Spotted Towhee	F16	0	0	1	0	0	1
Spotted Towhee	F17	0	0	1	0	0	1
Spotted Towhee	F18	0	0	1	0	0	1
Spotted Towhee	F19	0	0	1	0	0	1
Spotted Towhee	F2	0	0	2	1	1	3
Spotted Towhee	F20	1	1	2	1	1	2
Spotted Towhee	F21	0	0	1	0	0	1
Spotted Towhee	F22	0	0	1	0	0	1
Spotted Towhee	F3	0	0	2	0	0	2
Spotted Towhee	F4	1	1	4	1	1	3
Spotted Towhee	F5	0	0	2	2	2	4
Spotted Towhee	F6	0	0	2	2	2	5
Spotted Towhee	F7	0	0	1	0	0	1
Spotted Towhee	F8	0	0	1	0	0	1
Spotted Towhee	F9	0	0	1	0	0	2
Spotted Towhee	FR1	0	0	3	0	0	3
Spotted Towhee	FR10	3	2	6	1	0	4
Spotted Towhee	FR11	4	2	9	7	4	12
Spotted Towhee	FR12	2	1	6	1	0	6
Spotted Towhee	FR13	6	3	11	4	2	9
Spotted Towhee	FR14	0	0	2	0	0	2
Spotted Towhee	FR15	0	0	2	0	0	2
Spotted Towhee	FR16	0	0	2	0	0	2
Spotted Towhee	FR17	1	1	3	1	1	3
Spotted Towhee	FR18	0	0	2	1	1	3
Spotted Towhee	FR19	0	0	2	1	1	3
Spotted Towhee	FR2	5	2	11	6	3	12
Spotted Towhee	FR20	0	0	2	0	0	2
Spotted Towhee	FR21	0	0	2	0	0	2
Spotted Towhee	FR22	0	0	2	0	0	2
Spotted Towhee	FR3	2	0	7	2	0	7
Spotted Towhee	FR4	5	3	11	6	3	11
Spotted Towhee	FR5	6	3	13	6	3	12
Spotted Towhee	FR6	5	2	10	4	1	10
Spotted Towhee	FR7	1	0	4	2	1	6
Spotted Towhee	FR8	3	2	7	2	1	5

Spotted Towhee	FR9	4	2	8	4	3	8
Spotted Towhee	H1	0	0	2	0	0	2
Spotted Towhee	H10	0	0	2	0	0	2
Spotted Towhee	H11	0	0	2	1	1	3
Spotted Towhee	H12	0	0	2	0	0	2
Spotted Towhee	H13	1	1	3	2	2	5
Spotted Towhee	H14	0	0	2	0	0	2
Spotted Towhee	H15	0	0	2	0	0	2
Spotted Towhee	H16	0	0	2	0	0	2
Spotted Towhee	H17	0	0	2	0	0	2
Spotted Towhee	H18	0	0	2	1	1	3
Spotted Towhee	H19	0	0	2	1	1	3
Spotted Towhee	H2	1	1	3	1	1	3
Spotted Towhee	H20	0	0	2	0	0	2
Spotted Towhee	H21	0	0	2	0	0	2
Spotted Towhee	H22	0	0	2	1	1	3
Spotted Towhee	H23	0	0	2	0	0	2
Spotted Towhee	H24	3	3	6	1	1	3
Spotted Towhee	H25	0	0	2	1	1	3
Spotted Towhee	H3	3	3	5	3	2	5
Spotted Towhee	H4	0	0	2	1	1	3
Spotted Towhee	H5	2	1	4	0	0	2
Spotted Towhee	H6	0	0	2	0	0	2
Spotted Towhee	H7	0	0	2	2	1	4
Spotted Towhee	H8	0	0	2	0	0	2
Spotted Towhee	H9	0	0	2	0	0	2
Spotted Towhee	R1	0	0	3	1	1	4
Spotted Towhee	R10	2	1	4	3	2	6
Spotted Towhee	R11	0	0	3	0	0	3
Spotted Towhee	R12	0	0	2	0	0	3
Spotted Towhee	R13	1	1	4	0	0	3
Spotted Towhee	R14	0	0	3	0	0	3
Spotted Towhee	R15	0	0	3	0	0	3
Spotted Towhee	R16	1	1	4	0	0	2
Spotted Towhee	R17	3	2	6	4	3	6
Spotted Towhee	R18	1	1	4	0	0	3
Spotted Towhee	R19	3	2	5	4	4	7
Spotted Towhee	R2	0	0	3	2	1	4
Spotted Towhee	R20	0	0	3	0	0	3
Spotted Towhee	R21	0	0	2	0	0	3
Spotted Towhee	R22	0	0	2	0	0	3
Spotted Towhee	R23	0	0	2	0	0	3
Spotted Towhee	R24	2	2	5	2	2	5

Spotted Towhee	R25	0	0	3	0	0	3
Spotted Towhee	R3	2	1	4	2	2	5
Spotted Towhee	R4	0	0	2	0	0	3
Spotted Towhee	R5	0	0	3	0	0	3
Spotted Towhee	R6	0	0	3	0	0	3
Spotted Towhee	R7	1	1	4	3	2	5
Spotted Towhee	R8	0	0	3	0	0	3
Spotted Towhee	R9	0	0	3	2	1	4
White-breasted Nuthatch	F1	0	0	4	1	1	5
White-breasted Nuthatch	F10	0	0	3	0	0	3
White-breasted Nuthatch	F11	0	0	2	0	0	2
White-breasted Nuthatch	F12	0	0	4	0	0	4
White-breasted Nuthatch	F13	2	2	6	0	0	4
White-breasted Nuthatch	F14	0	0	1	0	0	1
White-breasted Nuthatch	F15	0	0	1	0	0	1
White-breasted Nuthatch	F16	0	0	1	0	0	1
White-breasted Nuthatch	F17	0	0	1	0	0	1
White-breasted Nuthatch	F18	0	0	1	0	0	1
White-breasted Nuthatch	F19	0	0	1	0	0	1
White-breasted Nuthatch	F2	0	0	2	0	0	2
White-breasted Nuthatch	F20	0	0	1	0	0	1
White-breasted Nuthatch	F21	0	0	1	0	0	1
White-breasted Nuthatch	F22	0	0	1	0	0	1
White-breasted Nuthatch	F3	0	0	2	0	0	2
White-breasted Nuthatch	F4	0	0	2	0	0	2
White-breasted Nuthatch	F5	0	0	4	1	1	5
White-breasted Nuthatch	F6	0	0	4	1	1	5
White-breasted Nuthatch	F7	2	2	5	0	0	3
White-breasted Nuthatch	F8	0	0	3	0	0	3
White-breasted Nuthatch	F9	0	0	3	0	0	3
White-breasted Nuthatch	FR1	1	0	6	2	1	7
White-breasted Nuthatch	FR10	1	0	5	2	1	6
White-breasted Nuthatch	FR11	0	0	3	0	0	3
White-breasted Nuthatch	FR12	4	2	11	1	0	7
White-breasted Nuthatch	FR13	1	0	7	1	0	7
White-breasted Nuthatch	FR14	0	0	2	0	0	2
White-breasted Nuthatch	FR15	0	0	2	0	0	2
White-breasted Nuthatch	FR16	0	0	2	0	0	2
White-breasted Nuthatch	FR17	0	0	2	0	0	2
White-breasted Nuthatch	FR18	0	0	2	0	0	2
White-breasted Nuthatch	FR19	0	0	2	0	0	2
White-breasted Nuthatch	FR2	2	1	5	0	0	3
White-breasted Nuthatch	FR20	0	0	2	1	1	3

White-breasted Nuthatch	FR21	0	0	2	0	0	2
White-breasted Nuthatch	FR22	1	1	3	0	0	2
White-breasted Nuthatch	FR3	0	0	3	0	0	3
White-breasted Nuthatch	FR4	0	0	3	0	0	3
White-breasted Nuthatch	FR5	1	0	7	1	0	7
White-breasted Nuthatch	FR6	3	1	8	2	1	8
White-breasted Nuthatch	FR7	2	1	6	2	1	6
White-breasted Nuthatch	FR8	1	0	5	1	0	5
White-breasted Nuthatch	FR9	1	0	5	1	0	5
White-breasted Nuthatch	H1	2	1	6	3	2	7
White-breasted Nuthatch	H10	6	4	11	3	1	7
White-breasted Nuthatch	H11	3	1	7	1	0	5
White-breasted Nuthatch	H12	1	0	5	1	0	5
White-breasted Nuthatch	H13	1	0	5	1	0	5
White-breasted Nuthatch	H14	1	0	5	1	0	5
White-breasted Nuthatch	H15	1	0	5	1	0	5
White-breasted Nuthatch	H16	1	0	5	1	0	5
White-breasted Nuthatch	H17	1	0	5	1	0	5
White-breasted Nuthatch	H18	2	1	6	1	0	5
White-breasted Nuthatch	H19	1	0	5	2	1	6
White-breasted Nuthatch	H2	1	0	5	2	1	6
White-breasted Nuthatch	H20	1	0	5	4	3	9
White-breasted Nuthatch	H21	2	1	6	1	0	5
White-breasted Nuthatch	H22	1	0	5	1	0	5
White-breasted Nuthatch	H23	3	2	7	1	0	5
White-breasted Nuthatch	H24	1	0	5	1	0	5
White-breasted Nuthatch	H25	2	1	6	2	1	6
White-breasted Nuthatch	H3	2	1	6	1	0	5
White-breasted Nuthatch	H4	2	1	6	1	0	5
White-breasted Nuthatch	H5	2	1	6	1	0	5
White-breasted Nuthatch	H6	1	0	5	2	1	6
White-breasted Nuthatch	H7	2	1	6	1	0	5
White-breasted Nuthatch	H8	1	0	5	1	0	5
White-breasted Nuthatch	H9	1	0	5	1	0	5
White-breasted Nuthatch	R1	5	4	11	3	1	8
White-breasted Nuthatch	R10	1	0	6	2	1	7
White-breasted Nuthatch	R11	4	2	10	5	2	10
White-breasted Nuthatch	R12	4	2	9	1	0	6
White-breasted Nuthatch	R13	1	0	6	1	0	6
White-breasted Nuthatch	R14	1	0	6	1	0	6
White-breasted Nuthatch	R15	1	0	6	1	0	6
White-breasted Nuthatch	R16	1	0	6	1	0	6
White-breasted Nuthatch	R17	3	1	8	1	0	6

White-breasted Nuthatch	R18	4	2	10	1	0	6
White-breasted Nuthatch	R19	2	1	7	1	0	6
White-breasted Nuthatch	R2	1	0	6	1	0	6
White-breasted Nuthatch	R20	4	2	9	3	1	8
White-breasted Nuthatch	R21	1	0	6	1	0	6
White-breasted Nuthatch	R22	1	0	6	1	0	6
White-breasted Nuthatch	R23	1	0	6	1	0	6
White-breasted Nuthatch	R24	1	0	6	1	0	6
White-breasted Nuthatch	R25	2	1	7	2	1	7
White-breasted Nuthatch	R3	1	0	6	1	0	6
White-breasted Nuthatch	R4	1	0	6	1	0	6
White-breasted Nuthatch	R5	1	0	6	1	0	6
White-breasted Nuthatch	R6	2	1	7	2	1	7
White-breasted Nuthatch	R7	3	1	8	1	0	6
White-breasted Nuthatch	R8	2	1	7	1	0	6
White-breasted Nuthatch	R9	2	1	7	2	1	7

Table A4.3. Comprehensive list of birds detected in the Piceance Basin study site, Colorado, USA, and their classification by foraging guild (De Graaf et al. 1985), nesting guild (Degraaf and Wentworth 1986), and habitat preference (Rodewald 2015).

Common Name	Latin Name	# of detections			Foraging Guild	Nesting Guild	Habitat group
		Mech. Dist	Fi re	Refer ence			
American robin	<i>Turdus migratorius</i>	5	11	9	Ground	Tree	Generalist
Ash-throated flycatcher	<i>Myiarchus cinerascens</i>	4	12	9	Shrub gleaner, upper canopy gleaner, Air sallier	Cavity	Woodland
Audubon yellow-rumped warbler	<i>Setophaga coronata</i>	4	2	0	Shrub forager, shrub gleaner	Tree	Woodland
Black-chinned hummingbird	<i>Archilochus alexandri</i>	1	0	1	Floral-hover gleaner	Tree	Woodland
Bewick's wren	<i>Thryomanes bewickii</i>	0	0	12	Ground	Cavity	Generalist
Blue-gray gnatcatcher	<i>Polioptila caerulea</i>	37	17	106	Upper canopy forager	Tree	Generalist
Brown-headed cowbird	<i>Molothrus ater</i>	12	9	4	Ground	Tree	Shrubland/grassland
Brewer's sparrow	<i>Spizella breweri</i>	3	2	17	Ground	Shrub	Shrubland/grassland
Broad-tailed hummingbird	<i>Selasphorus platycercus</i>	9	8	17	Floral-hover gleaner	Tree	Woodland
Black-throated gray warbler	<i>Setophaga nigrescens</i>	33	2	89	Shrub gleaner, upper canopy gleaner	Tree	Woodland
Bullock's Oriole	<i>Icterus bullockii</i>	0	3	0	Upper canopy forager	Tree	Woodland
Bushtit	<i>Psaltriparus minimus</i>	7	0	6	Shrub forager, shrub gleaner, upper canopy gleaner	Tree	Generalist
Cassin's finch	<i>Haemorhous cassinii</i>	37	22	34	Ground	Tree	Woodland
Canyon wren	<i>Catherpes mexicanus</i>	0	1	0	Ground	Permenant Structure	N/A

Chipping sparrow	<i>Spizella passerina</i> <i>Nucifraga</i>	61	46	67	Ground	Shrub	Generalist
Clark's nutcracker	<i>columbiana</i>	0	12	11	Ground	Tree	Woodland
Common nighthawk	<i>Chordeiles minor</i>	1	0	6	Air screener	Ground	Generalist
Cooper's hawk	<i>Accipiter cooperii</i>	0	3	8	Air hawk	Tree Permenant	Woodland
Common raven	<i>Corvus corax</i>	2	3	6	Ground	Structure	Generalist
Dark-eyed junco	<i>Junco hyemalis</i>	8	0	2	Ground	Ground	Generalist
Empidonax flycatcher	<i>Empidonax sp.</i>	24	6	80	Air sallier	Shrub	Generalist Shrubland/gr assland
Green-tailed towhee	<i>Pipilo chlorurus</i>	12	24	67	Ground	Shrub	Woodland
Hairy woodpecker	<i>Picoides villosus</i>	9	14	10	Bark gleaner	Cavity	Woodland
Hermit thrush	<i>Catharus guttatus</i> <i>Haemorhous</i>	1	0	2	Ground	Ground	Woodland
House finch	<i>mexicanus</i>	0	1	7	Ground	Tree	Generalist
House wren	<i>Troglodytes aedon</i> <i>Baeolophus</i>	8	5	14	Shrub forager	Cavity	Woodland
Juniper titmouse	<i>ridgwayi</i> <i>Chondestes</i>	7	4	20	Shrub forager	Cavity	Woodland Shrubland/gr assland
Lark sparrow	<i>grammacus</i>	0	23	14	Ground Shrub gleaner, upper canopy gleaner, Ground	Ground	assland
Lesser goldfinch	<i>Spinus psaltria</i> <i>Melospiza</i>	5	0	3	Ground forager, ground gleaner	Tree	Generalist Shrubland/gr assland
Lincoln's sparrow	<i>lincolni</i>	0	1	0	Ground forager, ground gleaner	Ground	assland
MacGillivray's warbler	<i>Geothlypis tolmiei</i>	0	0	3	Shrub forager	Shrub	Generalist Shrubland/gr assland
Mountain bluebird	<i>Sialia currucoides</i>	44	57	41	Shrub forager, Ground Shrub forager, shrub gleaner, upper canopy gleaner	Cavity	assland
Mountain chickadee	<i>Poecile gambeli</i>	29	9	67	Shrub forager, shrub gleaner, upper canopy gleaner	Cavity	Woodland
Mourning dove	<i>Zenaida macroura</i>	10	19	43	Ground	Tree	Generalist
Northern flicker	<i>Colaptes auratus</i>	2	3	10	Ground	Cavity	Woodland

Northern harrier	<i>Circus cyaneus</i>	0	0	1	Ground hawk	Ground	Shrubland/grassland
Orange-crowned warbler	<i>Oreothlypis celata</i>	0	0	2	Shrub forager, shrub gleaner, upper canopy gleaner	Ground	Generalist
Pinyon jay	<i>Gymnorhinus cyanocephalus</i>	0	5	3	Ground	Tree	Generalist
Plumbeous vireo	<i>Vireo plumbeus</i>	22	5	48	Shrub forager, shrub gleaner, upper canopy gleaner	Tree	Woodland
Red-breasted nuthatch	<i>Sitta canadensis</i>	2	1	7	Bark gleaner	Cavity	Woodland
Ruby-crowned kinglet	<i>Regulus calendula</i>	3	0	1	Shrub gleaner, upper canopy gleaner	Tree	Woodland
Rock wren	<i>Salpinctes obsoletus</i>	1	19	31	Ground	Ground	Generalist
Red-tailed hawk	<i>Buteo jamaicensis</i>	0	0	1	Ground	Tree	Generalist
Say's phoebe	<i>Sayornis saya</i>	0	2	1	Air sallier	Permenant Structure	Shrubland/grassland
Spotted towhee	<i>Pipilo maculatus</i>	43	23	137	Ground	Ground	Generalist
Vesper sparrow	<i>Pooecetes gramineus</i>	0	11	2	Ground	Ground	Shrubland/grassland
Violet-green swallow	<i>Tachycineta thalassina</i>	36	13	43	Air screener	Cavity	Generalist
Virginia warbler	<i>Oreothlypis virginiae</i>	0	1	7	Shrub gleaner, upper canopy gleaner	Ground	Woodland
White-breasted nuthatch	<i>Sitta carolinensis</i>	39	8	56	Bark gleaner	Cavity	Woodland
Western bluebird	<i>Sialia mexicana</i>	0	6	0	Ground gleaner, shrub forager	Cavity	Woodland
Western kingbird	<i>Tyrannus verticalis</i>	1	6	5	Air sallier	Tree	Generalist
Western meadowlark	<i>Sturnella neglecta</i>	0	5	1	Ground	Ground	Shrubland/grassland
Western scrub-jay	<i>Aphelocoma californica</i>	4	0	14	Ground	Tree	Generalist

Western Tanager	<i>Piranga ludoviciana</i>	0	0	1	Upper canopy forager	Tree	Woodland
Western wood pewee	<i>Contopus sordidulus</i>	0	1	1	Air sallier	Tree	Woodland
White-throated swift	<i>Aeronautes saxatalis</i>	0	0	3	Air screener	Permenant Structure	N/A

Table A4.4. Full list of mammal species and the number and proportion of sites each species was detected on remotely-triggered wildlife cameras at historically chained and reference sites in Pinyon-juniper woodlands from April-September 2014 in the Piceance Basin of Northwest Colorado.

Species	Mech. Dist.	Prop. of sites	Fir e	Prop. of sites	Referen ce	Prop. of sites
American Badger	3	0.14	2	0.09	12	0.55
Black Bear	1	0.05	3	0.14	8	0.36
Black-tailed Jackrabbit	1	0.05	1	0.05	3	0.14
Bobcat	9	0.41	7	0.32	19	0.86
Cow	12	0.55	11	0.50	14	0.64
Coyote	13	0.59	18	0.82	32	1.45
Domestic Dog	3	0.14	2	0.09	5	0.23
Wild Horse	3	0.14	2	0.09	4	0.18
Elk	14	0.64	21	0.95	33	1.50
Golden-mantled Ground Squirrel	9	0.41	5	0.23	23	1.05
Gray Fox	0	0.00	1	0.05	1	0.05
Human	7	0.32	3	0.14	5	0.23
Mountain Cottontail	19	0.86	13	0.59	42	1.91
Mule Deer	22	1.00	17	0.77	36	1.64
Pronghorn	0	0.00	2	0.09	1	0.05
Red Fox	1	0.05	1	0.05	2	0.09
Rock Squirrel	8	0.36	1	0.05	10	0.45
Chipmunk	1	0.05	0	0.00	7	0.32
Long-tailed Weasel	2	0.09	0	0.00	1	0.05
Mountain Lion	7	0.32	0	0.00	14	0.64
Striped Skunk	0	0.00	0	0.00	2	0.09
Western Spotted Skunk	0	0.00	0	0.00	3	0.14
White-tailed Jackrabbit	0	0.00	0	0.00	1	0.05
Bushy-tailed Woodrat	0	0.00	0	0.00	2	0.09
Domestic sheep	0	0.00	0	0.00	3	0.14
Wyoming Ground Squirrel	0	0.00	0	0.00	1	0.05