

EFFECTS OF PRESCRIBED FIRE ON WINTERING,
BARK-FORAGING BIRDS IN NORTHERN ARIZONA

By Theresa L. Pope

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Approved:

William M. Block, Ph.D., Co-chair

Paul Beier, Ph.D., Co-chair

Peter Z. Fulé, Ph.D.

Thomas D. Sisk, Ph.D.

ABSTRACT

EFFECTS OF PRESCRIBED FIRE ON WINTERING, BARK-FORAGING BIRDS IN NORTHERN ARIZONA

THERESA L. POPE

Forest management practices of the past century have led to an increase in unnatural and destructive crown fires in ponderosa pine (*Pinus ponderosa*) forests of the southwest. To combat large fires, forest managers are attempting to simulate past fire regimes of low-intensity surface fires using prescribed fire. While there have been many studies investigating the effects of crown fires on birds, few studies exist on the effects of prescribed fire on birds, especially during winter. Winter may be a critical time for resident species since food is generally limited. Any information on how resident species and food availability are affected by prescribed fire in winter is useful. This study examines the effects of prescribed fire on wintering, bark-foraging birds in northern Arizona, including hairy woodpecker (*Picoides villosus*), pygmy nuthatch (*Sitta pygmaea*) and white-breasted nuthatch (*Sitta carolinensis*). Distance sampling to assess bird density (analyzed using Distance 5.0 Release 3), foraging observations and bark beetle surveys were conducted during the 2004 – 2005 and 2005 – 2006 winter seasons. Bark beetle presence was twice as great in the burn units as in the control units (12% of ponderosa pines had bark beetle activity as compared to 6%), potentially providing additional food for bark-foraging birds during winter where bark beetles occur. Hairy woodpeckers had greater estimated densities

(\pm SE) on the burn units than the control units ($\hat{D}=10.8 \pm 2.0$ to $\hat{D}=2.1 \pm 0.3$ per 100 ha, respectively). White-breasted nuthatches ($\hat{D}=9.8 \pm 0.9$ and $\hat{D}=11.6 \pm 1.4$ per 100 ha) and pygmy nuthatches ($\hat{D}=45.0 \pm 7.5$ and $\hat{D}=39.7 \pm 7.1$ per 100 ha) had similar estimated densities (\pm SE) in both units. Furthermore, hairy woodpeckers foraged on trees with bark beetle activity in greater proportion than trees with bark beetle activity were found on all of the units as a whole ($p < 0.01$). The pygmy ($p = 0.53$) and white-breasted nuthatches ($p = 0.91$) did not. Therefore, hairy woodpeckers may be utilizing the bark beetles as an additional food source in the winter. The nuthatches may not utilize this food source as much, relying more on other food resources during winter.

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PREFACE

This document is written in manuscript format. Chapter 2, “Effects of prescribed fire on wintering, bark-foraging birds in northern Arizona,” will be submitted to the Journal of Wildlife Management. Chapter 3, “Winter bird communities in ponderosa pine forests of northern Arizona,” will be submitted to the Wilson Journal of Ornithology. Any format differences and redundancy can be attributed to guidelines provided for submission of manuscripts to these journals.

CHAPTER 1

INTRODUCTION

LITERATURE REVIEW

Prescribed fire

After nearly a century of fire suppression activities, forest managers are realizing the adverse effects of fire suppression on forest ecosystems. Many forests, including the ponderosa pine (*Pinus ponderosa*) forests in northern Arizona, have had fire as a natural disturbance of the system until fire suppression efforts began in the early 20th century. Frequent, low-intensity fires were part of the ecology and evolutionary history of ponderosa pine forests (Cooper 1960, Covington and Moore 1994a,b; Swetnam and Baison 1996, Moir et al. 1997). These fires were also heterogeneous in nature, leading to a mosaic of burned and unburned patches on the landscape (Fulé et al. 1997). However, forest management since Euro-American settlement has caused a reduction in fire frequency and size. Several factors are associated with this reduction including roads and trails that broke up fuel continuity; domestic livestock grazing reducing herbaceous fuels; proficiency at extinguishing natural fires shortly after ignition; and active fire suppression that began near the turn of the 20th century (Covington and Moore 1994a). Heavy livestock grazing, timber harvesting, and fire exclusion have led to forests with many more younger and smaller trees, fewer older and larger trees, heavy forest floor fuel loads, and reduced herbaceous production (Cooper 1960, Covington and Moore 1994a,b, Swetnam and Baison 1996).

Prior to the 1950s, crown fires were extremely rare to non-existent (Cooper 1960). Structural changes shifted the fire regime in ponderosa pine forests from frequent, low-intensity surface fires to high-intensity crown fires (Covington and Moore 1994*a,b*). An example is the Rodeo-Chediski fire that burned over 189,000ha in Arizona during the summer of 2002. Over 400 structures burned and \$153 million was spent on fire suppression. Yet today, there is a general understanding that some fires can be beneficial to the landscape and that complete fire exclusion is not a sustainable course of action.

As part of the National Fire Plan and Healthy Forests Initiative, forest managers are looking at alternate forest management strategies that include using prescribed fire in fuel reduction and forest restoration treatments. State, federal, and local agencies put enormous resources into efforts to reduce fire occurrence while at the same time advocating the need to use fire to promote healthy ecosystems (SNEP 1996). The challenge they face is how to restore some aspects of a more natural fire regime while minimizing threat of wildfire to people and property (SNEP 1996). Prescribed fire is frequently advocated as a tool that can be used for landscape level fuel reduction while simultaneously restoring fire as an ecosystem process (McKelvey et al. 1996).

With the use of prescribed fire on the rise, forest managers must consider the possible effects of prescribed fire on wildlife, including birds. While there have been many studies investigating the effects of wildfires on birds (see Saab and Powell 2005), few studies exist on the effects of prescribed fire on birds (Bock and Bock 1983, Horton and Mannan 1988, Dickson 2006). Future research should emphasize avian response to low-intensity, surface fires (Bock and Block 2005). This research

can help overcome a major stumbling block, i.e. the lack of knowledge of the effects of prescribed fire on avian ecology, for developing ecologically sound fire management policies (Kotliar et al. 2005).

Bark beetle response to fire

Bark beetles are found under the bark of conifer species. Several species of bark beetles that are attracted to fire injured ponderosa pine trees persist at endemic population levels in northern Arizona, including western pine beetle (*Dendroctonus brevicomis*), mountain pine beetle (*D. ponderosae*), pine engraver beetle (*Ips* spp.) and red turpentine beetle (*D. valens*) (Flanagan 1996, Sanchez-Martinez and Wagner 2002).

Bark beetles are attracted to burned trees after fire. Miller and Patterson (1927) first reported the direct relationship between fire injury and subsequent insect damage on burned-over areas. Trees with scorched trunks or crowns can become susceptible to bark beetles, depending on the severity and pattern of scorching (Ferrell 1996, McCullough et al. 1998, Bradley and Tueller 2001). Ponderosa pines with heavy crown scorch tend to be more heavily colonized by bark beetles (Wallin et al. 2003, McHugh et al. 2003). While not all species of bark beetles are primary mortality agents (e.g. *D. valens*), they may predispose trees to other mortality agents.

Bark beetle infestations may be a serious problem especially following the initial reintroduction of fire into stands with high fuel levels due to long term fire exclusion (Ferrell 1996). An increase in the number of trees infested by bark beetles is a frequent aftermath of so-called “light” fires, which rarely kill mature trees (Miller and Patterson 1927). Machmer (2002) and Bradley and Tueller (2001) reported an

increase in the abundance of bark beetles and wood borers after prescribed fire. Understanding the ecology of fire-insect interactions is needed as we identify strategies to manage forest pests, enhance forest health, and maintain biological diversity in forest ecosystems (McCullough et al. 1998).

Woodpecker response to bark beetles

Information on the effects of fire on birds in southwestern ponderosa pine forests indicates that several species benefit from disturbance. Woodpeckers can be nomadic and tend to increase in number after fire in response to an increasing bark beetle food source (Koplin 1969, Brawn et al. 2001, Covert 2003). Koplin (1969) attributed an increase in woodpecker density following a wildfire in Colorado to aggregation in response to insect prey and not increased woodpecker reproduction. Birds tend to be opportunistic foragers and often make use of super-abundant food resources (Szaro et al. 1990). As prey density increases, predators may respond functionally by attacking more prey or numerically by aggregating or having increased survival rates (Garton 1979, Kroll and Fleet 1979, Machmer and Steeger 1995).

Hairy woodpeckers feed primarily on bark beetles and wood borers in winter and appear to be opportunistic feeders, shifting about in large numbers in winter in search of food and shelter (Beal 1911, Otvos 1965, Crockett and Hansley 1978). Woodpecker populations increased most significantly during the non-breeding season in response to bark beetle outbreaks (Crockett and Hansley 1978). Although woodpecker response to bark beetle outbreaks has been widely studied, literature on the response by other bark-foraging species, such as pygmy and white-breasted

nuthatches, is not currently available. However, hairy woodpeckers, pygmy nuthatches and white-breasted nuthatches are all known to have bark beetles as part of their winter diet (Beal 1911, Norris 1958, Stallcup 1968, Anderson 1976)

Avian foraging patterns in winter

Seasonal changes in availability of woodpecker food items might be expected to cause concurrent changes in foraging methods (Conner 1979, Szaro et al. 1990).

During winter, hairy woodpeckers use methods that penetrate trees deeper, such as scaling bark and excavating, than during milder seasons (Conner 1979).

Woodpeckers use subsurface foraging techniques to penetrate the subcambium layer where beetle larvae are extracted (Machmer and Steeger 1995). Bark scaling by woodpeckers also tends to make food more available to other bark foragers, such as brown creepers (Otvos 1979). Nuthatches and creepers use superficial foraging techniques to chip away bark flakes, locating insect prey on or near the bark surface (Stallcup 1968, Machmer and Steeger 1995).

Although these birds all use bark as a foraging substrate, investigations by Stallcup (1968) reveal that each species is segregated spatially and temporally, allowing these birds to coexist in the same region with reduced or no interspecific competition for food. In a study by McEllin (1979), white-breasted nuthatches had lower mean foraging heights than pygmy nuthatches during the non-reproductive season. Also, white-breasted nuthatches used a significantly smaller proportion of the total tree height during the non-reproductive period than pygmy nuthatches, using mainly the trunk and limbs. In contrast, pygmy nuthatches utilized all parts of the tree, including the twigs and needle clusters (Norris 1958, McEllin 1979). McEllin

(1979) additionally found that white-breasted nuthatches maintained an exclusivity strategy throughout the entire year through territorial defense, whereas pygmy nuthatches exhibited an inclusivity strategy in which tolerance and, possibly, cooperation resulted in group utilization of habitat space. This information suggests that several species of bark-foraging birds can make use of the same food source without much competition.

Effects of prescribed fire on wintering, bark-foraging birds and bird communities

Studies investigating the effects of fire on wintering birds are few (Blake 1982, Kreisel and Stein 1999, Covert 2003, Bock and Block 2005), especially for prescribed fire (King et al. 1998). More studies should investigate prescribed fire effects on wintering birds, since winter may be a crucial time for birds. First, food is generally limited in the winter; as insects are not as abundant and plants are dormant during this time. As a result, food is more patchily distributed and birds need to be opportunistic in their foraging ecology. Second, conditions are harsher during winter, with lower temperatures and greater snow cover. Therefore, species are more vulnerable to physiological stress and may not survive the winter conditions. Winter survival can have an effect on populations, since birds that survive the winter will be able to reproduce in the breeding season. Also, habitat requirements may be different in the winter, since habitat required for breeding may not be the same as habitat needed in the winter for providing food and shelter from the harsh conditions.

My research was conducted on the Birds and Burns Network study sites in northern Arizona. The Birds and Burns Network is a collaborative effort, led by the

U.S. Forest Service Rocky Mountain Research Station, examining fire effects on populations and habitats of breeding birds in ponderosa pine forests in eight states across the western United States. Collaborators include a number of national forests, Pacific Northwest Research Station, The Nature Conservancy and several universities. Cavity-nesting birds were chosen as the focus of this research because many of them depend on fire-maintained habitats for their dispersal and movements, food, and nest and roost sites; they are designated as Management Indicator Species and Sensitive Species by state and federal agencies; and they are responsive to timber and fire management activities. The goal of the project is to understand the ecological consequences of fire management for breeding birds in ponderosa pine forests.

For Chapter 2, I investigated the effects of prescribed fire on wintering, bark-foraging birds in ponderosa pine forests of northern Arizona. I choose three resident bark-foraging birds as focal species: hairy woodpeckers (*Picoides villosus*), white-breasted nuthatches (*Sitta carolinensis*), and pygmy nuthatches (*S. pygmaea*). Distance sampling to assess bird density, foraging observations, and bark beetle surveys were conducted in northern Arizona during the winters of 2004 – 2006, the first two winters after all burn units were treated by prescribed fire. Since bark beetles have been known to increase activity in burned areas (Miller and Patterson 1927, Ferrell 1996, McCullough et al. 1998, Bradley and Tueller 2001, Wallin et al. 2003, McHugh et al. 2003), they could provide more food for opportunistic bark-foraging birds. Therefore, I expected an increase in bark-foraging bird density and a change in foraging patterns in burn units as compared to the control units after being treated by prescribed fire.

I also investigated the effects of prescribed fire on winter bird communities in northern Arizona (Chapter 3). I used similarity indices and rank order of abundance to examine treatment (burn vs. control) and year differences between communities, with respect to the community composition and structure. Since prescribed fire generally does not have a great effect on the general structure of the forest, I expected that communities of birds in the winter should be similar in the burned and unburned forests.

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CHAPTER 2

EFFECTS OF PRESCRIBED FIRE ON WINTERING, BARK-FORAGING BIRDS IN NORTHERN ARIZONA

Abstract: This study examines the effects of prescribed fire on wintering, bark-foraging birds, including hairy woodpecker (*Picoides villosus*), pygmy nuthatch (*Sitta pygmaea*) and white-breasted nuthatch (*Sitta carolinensis*) in ponderosa pine (*Pinus ponderosa*) forests of northern Arizona. Distance sampling to assess bird density, foraging observations, and bark beetle surveys were conducted during the winters of 2004 – 2006. Bark beetle activity was twice as great in the burn units (12% of the ponderosa pine trees had bark beetle activity as compared to 6% in the control units), potentially providing additional food for bark-foraging birds during winter. Hairy woodpeckers had greater estimated densities (+/- SE) in the burn units than the control units ($\hat{D}=10.8 \pm 2.0$ to $\hat{D}=2.1 \pm 0.3$ per 100 ha, respectively). White-breasted nuthatches ($\hat{D}=9.8 \pm 0.9$ and $\hat{D}=11.6 \pm 1.4$ per 100 ha) and pygmy nuthatches ($\hat{D}=45.0 \pm 7.5$ and $\hat{D}=39.7 \pm 7.1$ per 100 ha) had similar estimated densities (+/- SE) in both units. Furthermore, hairy woodpeckers foraged on trees with bark beetle activity in greater proportion than trees were found with bark beetle activity on all of the units as a whole ($p < 0.01$). The pygmy ($p = 0.53$) and white-breasted nuthatches ($p = 0.91$) did not. Therefore, hairy woodpeckers may be utilizing the bark beetles as an additional food source in the winter. The nuthatches

may not utilize this food source as much, relying more on other food resources during winter.

Key Words: Arizona, bark-foraging birds, hairy woodpecker, *Picoides villosus*, prescribed fire, pygmy nuthatch, *Sitta carolinensis*, *Sitta pygmaea*, white-breasted nuthatch, winter

Introduction

After nearly a century of fire suppression activities, forest managers are realizing the adverse effects of fire suppression on forest ecosystems. Many forests, including the ponderosa pine (*Pinus ponderosa*) forests in northern Arizona, have had fire as a natural disturbance of the system until fire suppression efforts began in the early 20th century. Frequent, low-intensity fires were part of the ecology and evolutionary history of ponderosa pine forests (Cooper 1960, Covington and Moore 1994a,b; Swetnam and Baison 1996, Moir et al. 1997). These fires were also heterogeneous in nature, leading to a mosaic of burned and unburned patches on the landscape (Fulé et al. 1997). However, forest management since Euro-American settlement has caused a reduction in fire frequency and size. Several factors are associated with this reduction, including roads and trails that broke up fuel continuity; domestic livestock grazing reducing herbaceous fuels; proficiency at extinguishing natural fires shortly after ignition; and active fire suppression that began near the turn of the 20th century (Covington and Moore 1994a). Heavy livestock grazing, timber harvesting, and fire exclusion have led to forests with many more younger and smaller trees, fewer older and larger trees, heavy forest floor fuel loads, and reduced

herbaceous production (Cooper 1960, Covington and Moore 1994*a,b*, Swetnam and Baison 1996).

Prior to the 1950s, crown fires were extremely rare to non-existent (Cooper 1960). Structural changes shifted the fire regime in ponderosa pine forests from frequent, low-intensity surface fires to high-intensity crown fires (Covington and Moore 1994*a,b*). An example is the Rodeo-Chediski fire that burned over 189,000ha in Arizona during the summer of 2002. Over 400 structures burned and \$153 million was spent on fire suppression. As part of the National Fire Plan and Healthy Forests Initiative, forest managers are looking at alternate forest management strategies that include using prescribed fire in fuel reduction and forest restoration treatments.

With the use of prescribed fire on the rise, forest managers must consider the possible effects of prescribed fire on wildlife, including birds. While there have been many studies investigating the effects of wildfires on birds (see Saab and Powell 2005), few studies exist on the effects of prescribed fire on birds (Bock and Bock 1983, Horton and Mannan 1988, Dickson 2006). Future research should emphasize avian response to low-intensity, surface fires (Bock and Block 2005a). This research can help overcome a major stumbling block, i.e. the lack of knowledge of the effects of prescribed fire on avian ecology, for developing ecologically sound fire management policies (Kotliar et al. 2005).

Studies investigating the effects of fire on wintering birds are few (Blake 1982, Kreisel and Stein 1999, Covert 2003, Bock and Block 2005b), especially for prescribed fire (King et al. 1998). More studies should investigate prescribed fire effects on wintering birds, since winter may be a crucial time for birds. First, food is generally limited in the winter; as insects are not as abundant and many plants are

dormant during this time. Second, conditions are harsher during winter, with lower temperatures and greater snow cover. Therefore, species are more vulnerable to physiological stress and may not survive the winter conditions. Winter survival can have an effect on populations, since birds that survive the winter will be able to reproduce in the breeding season. Also, habitat requirements may be different in the winter, since habitat required for breeding may not be the same as habitat needed in the winter for providing food and shelter from the harsh conditions.

During winter, birds may travel outside of their breeding territory in search of food. According to Garton (1979), when a insect prey species increases in abundance in one type of patch, an optimally foraging bird will increase the percentage of its time spent foraging in that patch type until it has depleted that patch type down to a level comparable to other patch types. For example, Morrison et al. (1986) found an increase in the use of incense cedar (*Calocedrus decurrens*) by many birds in winter, which was attributed to the availability of a prey species, incense cedar scale (*Xylococcus macrocarpae*), overwintering under the loose bark. Bark-foraging birds in northern Arizona may respond in a similar fashion to bark beetle irruptions.

I investigated the effects of prescribed fire on wintering, bark-foraging birds in ponderosa pine forests of northern Arizona. I choose three resident bark-foraging birds as focal species: hairy woodpeckers (*Picoides villosus*), white-breasted nuthatches (*Sitta carolinensis*), and pygmy nuthatches (*S. pygmaea*). Distance sampling to assess bird density, foraging observations, and bark beetle surveys were conducted in northern Arizona during the winters of 2004 – 2006, the first two winters after all burn units were treated by prescribed fire. Since bark beetles have been known to increase activity in burned areas (Miller and Patterson 1927, Ferrell

1996, McCullough et al. 1998, Bradley and Tueller 2001, Wallin et al. 2003, McHugh et al. 2003), they could provide more food for opportunistic bark-foraging birds. Therefore, I expected an increase in bark-foraging bird density and a change in foraging patterns in burn units as compared to the control units after being treated by prescribed fire.

Study Area

My research was conducted on the Birds and Burns Network study sites in northern Arizona. The Birds and Burns Network is a collaborative effort, led by the U.S. Forest Service Rocky Mountain Research Station, examining fire effects on populations and habitats of breeding birds in ponderosa pine forests in eight states across the western United States. Collaborators include a number of national forests, Pacific Northwest Research Station, The Nature Conservancy and several universities. Cavity-nesting birds were chosen as the focus of this research because many of them depend on fire-maintained habitats for their dispersal and movements, food, and nest and roost sites; they are designated as Management Indicator Species and Sensitive Species by state and federal agencies; and they are responsive to timber and fire management activities. The goal of the project is to understand the ecological consequences of fire management for breeding birds in ponderosa pine forests of the Interior West.

The northern Arizona study sites were located in the Coconino and Kaibab National Forests, 60 km southeast and 35 km northwest of Flagstaff, respectively. Ponderosa pine was the dominant overstory tree on the three Kaibab study units. Pinyon pine (*P. edulis*) and one-seed (*Juniperus monosperma*) and alligator (*J.*

deppeana) junipers occurred on the control units, but contributed little to stand canopies. Ponderosa pine also dominated the overstory on the two Coconino study units, with Gambel oak (*Quercus gambelii*) contributing to canopy structure. Alligator juniper is also found on both of the Coconino units. Open grassland patches in all locations are dominated by bunchgrass species, including Arizona fescue (*Festuca arizonica*) and blue gramma (*Bouteloua gracilis*). The topography on the Kaibab is relatively flat, with elevations ranging from 2100m to 2300m. Topography on the Coconino varies from flat to steep hills, with elevations ranging from 2070m to 2160m.

Each study site had a burn treatment unit paired with control unit(s) of similar forest structure (Table 1). The Kaibab study area needed two control units to capture the large tree component that was present in the treatment unit. Treatment units were chosen in consultation with district fire managers on each forest. Control units were then placed in representative areas with similar stand structures within 1 km of the treatment unit where no management was planned by the forests. A systematic random sampling design was used for the placement of point count stations. The point count stations were assigned using a GIS algorithm, after randomizing the placement of the first station (see Dickson 2006). U.S. Forest Service personnel conducted the prescribed fires during Fall 2003 on the Coconino treatment unit and on the Kaibab treatment unit during Fall 2003 and Spring 2004.

All prescriptions were characterized as broadcast burns with expected fire behaviors of low to moderate intensity (see Dickson 2006). These fires were heterogeneous in nature, with areas that were not burned at all, as well as areas with burns severe enough to kill some trees. The average (+/- SE) maximum bole char

height for the two burn units was 1.2m +/- 0.04. The average (+/- SE) percent of the bole circumference that was charred at the base was 65.0 +/- 0.81. And the average (+/- SE) percent of needles scorched was 6.6 +/- 0.39. The fire effect variables for each individual burn unit are listed in Table 2. Since I was investigating general relationships between prescribed fire effects and wintering, bark-foraging bird density and foraging behavior, I pooled data across the burn units for analyses.

Methods

Bird surveys - I measured avian densities in each unit using point-transect distance sampling (Buckland et al. 2001). Distance sampling takes into account differences in detectability, which usually decreases with increasing distance from the observer, when computing density estimations. Burnham et al. (1980) recommended a minimum of 40 or more individuals per sampling period for density estimation, with preferred total counts of 60-80 individuals per sampling period. To obtain these recommendations, more effort is generally necessary than with traditional point count methodology (Verner and Ritter 1985, Rosenstock et al. 2002, Norvell et al. 2003). However, estimating density with distance sampling is considered less biased than relative abundance indices that assume constant detectability (Rosenstock et al. 2002, Norvell et al. 2003).

At each study area, one unit was treated with a low to moderate intensity prescribed fire a full growing season before point counts began. The Kaibab study area had 40 point count stations in the burn unit and 50 point count stations between the two control units. The Coconino study area had 40 point count stations in the burn unit and in the control unit. Therefore, there were a total of 170 point count

stations (80 in the burn units and 90 in the control units) (Table 1). Each station was approximately 300m apart and at least 200m from unit edges. At each station, I recorded distance (using categories up to 100m) and direction to all birds observed during a 5-minute survey period. The number of individuals, sex (if known), and type of detection were also recorded. Point counts began within 30 minutes of sunrise and concluded within 5 hours. I did not count in windy or wet (more than a light snow) conditions. I counted each station eight times between mid-October and mid-March, 2004 – 2006.

I estimated bird densities using Program Distance 5.0 Release 3 (Thomas et al. 2005). I used the conventional distance sampling (CDS) analysis engine. For each of my three focal species, I ran multiple models, post-stratifying by burn and control. Three models (uniform cosine, half-normal cosine, and hazard-rate cosine) were compared for each species and the best one was chosen using Akaike's Information Criterion (AIC) (Table 3). For hairy woodpecker (HAWO) and white-breasted nuthatch (WBNU), I was able to have separate detection functions for each stratum (burn vs. control). However, pygmy nuthatches (PYNU) were estimated using a global detection function. Therefore, the detection probability was assumed to be the same in each unit for the pygmy nuthatch. For hairy woodpecker and white-breasted nuthatch, the top two models had $\Delta AIC < 2.00$. Therefore, I used model averaging to estimate the densities (Buckland et al. 2001). For the pygmy nuthatch, only the half-normal cosine model had $\Delta AIC < 2.00$. Therefore, I used the density estimate given by this model.

Bark beetle surveys - I conducted bark beetle surveys using 10-m radius circular plots (0.03 ha) at each point count station. Four plots were assigned: one at the station center and three within 100m of the station center. To find the locations of the additional plots, I assigned one a random bearing and distance from the station center (between 20m and 90m, to prevent overlap with center plot and remain within 100-m radius of station center), using a random number table. The other two plots were offset from the first by 120 and 240 degrees, respectively, and were also assigned distances from a random number table. I determined the presence or absence of bark beetles in each ponderosa pine over 12.7cm DBH (to exclude saplings) by noting evidence of bark beetle activity (i.e. pitch tubes, frass, boring dust, and fading needles) and woodpecker foraging (i.e. bark flaking, hole drilling). I also recorded the following information about each marked tree: DBH (to 0.1cm), maximum bole char height (to 0.5m), bole char severity, percent of the bole circumference charred at base, pre-burn crown ratio (crown length/tree height), and crown volume burned (percent green, brown, and black) (USDI NPS Fire Monitoring Handbook 2003). Bark beetle surveys occurred during both field seasons.

A Pearson chi-squared test was used to determine if the proportion of ponderosa pine trees with signs of bark beetle activity were different among burn and control units. Analyses were run using JMP 6 (SAS Institute, Inc. 2005). SPSS for Windows Release 10.1.3 (SPSS, Inc. 1989-2000) was used to analyze the fire effects measured on all of these trees as well as trees where foraging observations occurred.

Foraging observations - I conducted foraging observations of hairy woodpeckers, pygmy nuthatches, and white-breasted nuthatches from mid-October until mid-

March, during winters of 2004 – 2005 and 2005 – 2006. At each point-count station, I searched the area within a 100-m radius of the station for foraging birds for 8 minutes. Once a bird was located, I watched the bird for 10 seconds before recording data. This period of time allowed the bird to resume “normal” activity patterns after being disturbed and also prevented only recording conspicuous behavior (Noon and Block 1990). After the initial 10 seconds, I recorded the first foraging event observed. For each tree where a foraging event occurred, I recorded the same information as I did for the bark beetle survey trees, including evidence of bark beetle activity. All stations were visited equally throughout the season.

Pearson chi-squared tests were used to determine if the proportion of ponderosa pine trees with signs of bark beetle activity used for foraging were different among burn and control units and between trees used to forage and those surveyed across the units. If so, Fisher’s Exact tests were used to determine if trees with bark beetle activity were used in greater proportion. These analyses were run using JMP 6 (SAS Institute, Inc. 2005). I also used log-linear analysis to determine significant factors in foraging behavior using SPSS for Windows Release 10.1.3 (SPSS, Inc. 1989-2000).

Results

Bark-foraging bird densities - Over eight visits to each point in the two winters following prescribed fire treatments, 1244 observations were made of hairy woodpeckers, pygmy nuthatches, and white-breasted nuthatches. Hairy woodpeckers were detected 113 times in the burn units and 47 times in the control units. Pygmy nuthatches were detected 341 times in the burn units and 337 times in the control

units. White-breasted nuthatches were detected 197 times in the burn units and 209 times in the control units. Therefore, total counts exceeded the recommended 60-80 individuals per sampling period in all cases except for hairy woodpeckers in control units, which exceeded the minimum recommendation of 40 individuals (Burnham et al. 1980). Taking into account differing detection probabilities, Program Distance 5 Release 3 calculated density estimates (per 100ha) for these species. Hairy woodpeckers had an estimated density (+/- SE) of $\hat{D}=10.8 \pm 2.0$ individuals per 100ha in the burn units and only $\hat{D}=2.1 \pm 0.3$ individuals per 100 ha in the control units. Pygmy nuthatches had an estimated density (+/- SE) of $\hat{D}=45.0 \pm 7.5$ individuals per 100ha in the burn units and $\hat{D}=39.7 \pm 7.1$ individuals per 100 ha in the control units. White-breasted nuthatches had an estimated density (+/- SE) of $\hat{D}=9.8 \pm 0.9$ individuals per 100ha in the burn units and $\hat{D}=11.6 \pm 1.4$ individuals per 100 ha in the control units. Therefore, hairy woodpecker density was five times greater in the burn units and the nuthatches did not have different densities in the burn and control units.

Bark beetle activity - I examined 4412 ponderosa pine trees for signs of bark beetle activity. In the burn units, 269 of 2320 (12%) ponderosa pines had signs of bark beetle activity. Only 118 of 2092 (6%) ponderosa pines in the control units did. Since there were twice as many ponderosa pine trees with evidence of bark beetle activity in the burn units than in the control units, bark beetle presence was greater in the burn units than the control units ($X^2 = 48.74$, $p < 0.01$).

Foraging behavior of bark-foraging birds - Foraging observations were recorded for all three focal species over the two winters of this study. Morrison (1984) and Brennan and Morrison (1990) indicated that 30-40 samples adequately described foraging behavior. Because of small samples, foraging observations from both seasons and between sexes (when sex could be determined) were combined.

Hairy woodpeckers were observed foraging 105 times, 71 in the burn units and 34 in the control units. Pygmy nuthatches were observed foraging 141 times, 74 in the burn units and 67 in the control units. White-breasted nuthatches were observed foraging a total of 179 times, 95 in the burn units and 84 in the control units.

Hairy woodpeckers showed a difference in the use of ponderosa pine trees with bark beetle activity between burn and control units ($X^2 = 4.28, p = 0.04$). Hairy woodpeckers foraged on trees showing evidence of bark beetle activity 61% of the time in the burn units and only 37% of the time in control units (Figure 3). Pygmy nuthatches foraged on trees showing evidence of bark beetle activity 11% of the time in the burn and control units (Figure 3). White-breasted nuthatches foraged on trees showing evidence of bark beetle activity 15% of the time in the burn units and 8% of the time in control units (Figure 3). Therefore, hairy woodpeckers foraging on ponderosa pine trees in the burn units were more likely to forage on trees with evidence of bark beetle activity than those observed in the control units ($p = 0.03$). Pygmy ($X^2 = 0.004, p = 0.95$) and white-breasted ($X^2 = 1.66, p = 0.20$) nuthatches used ponderosa pine trees with evidence of bark beetle activity in the same proportion in burn and control units.

For the observations on ponderosa pine trees, the proportion of events recorded that occurred on trees showing evidence of bark beetle activity was

compared to the proportion of ponderosa pine trees with bark beetle activity surveyed across the units. As noted earlier, 12% of the trees surveyed in the burn unit had evidence of bark beetle activity, while only 6% of the trees on the control units did. Again, hairy woodpeckers showed a difference in the proportion of trees with bark beetle activity used for foraging as compared to those surveyed across the units. On the burn units, hairy woodpeckers foraged on trees with evidence of bark beetle activity 61% of the time ($X^2 = 136.74, p < 0.01$) (Figure 4). On the control units, hairy woodpeckers foraged on trees with evidence of bark beetle activity 37% of the time ($X^2 = 46.30, p < 0.01$) (Figure 5). Therefore, hairy woodpeckers were using trees with signs of bark beetle activity in greater proportion than the trees surveyed across the units ($p < 0.01$).

This was not the case with pygmy and white-breasted nuthatches. For pygmy nuthatches, the proportion of ponderosa pine trees with bark beetle activity was not different between trees where foraging observations were recorded and trees surveyed in the burn ($X^2 = 0.02, p = 0.89$) (Figure 4) or control ($X^2 = 3.49, p = 0.06$) (Figure 5) units. Pygmy nuthatches foraged on trees with evidence of bark beetle activity 11% of the time on the burn and control units. For white-breasted nuthatches, the proportion of ponderosa pine trees with bark beetle activity was not different between where foraging observations were recorded and trees surveyed in the burn ($X^2 = 0.83, p = 0.36$) (Figure 4) or control ($X^2 = 0.87, p = 0.35$) (Figure 5) units. White-breasted nuthatches foraged on trees with evidence of bark beetle activity 15% of the time on the burn units and 8% of the time on the control units. Therefore, pygmy ($p = 0.53$) and white-breasted nuthatches ($p = 0.91$) did not forage on trees with evidence of bark beetle activity in greater proportion than trees surveyed on the units.

For the log-linear analysis, I used four factors as inputs into the model to find out which factors are the most important for each species while foraging. The factors I used were tree species, bark beetles, maximum bole char height, and DBH (Table 4). For the hairy woodpecker, the significant factors included the interaction between maximum bole char height and tree species, the interaction between bark beetles and tree species, and the interaction between DBH and maximum bole char height (Table 5). All of the factors were significant for pygmy nuthatch without any interaction terms (Table 5). And the significant factors for the white-breasted nuthatch included the interaction between tree species and bark beetles, the interaction between tree species and maximum bole char height, and DBH (Table 5). For hairy woodpecker and white-breasted nuthatch, the interaction between tree species and bark beetles is expected since bark beetles only attack ponderosa pine trees on these sites. Also, pygmy and white-breasted nuthatches used larger trees (DBH > 45.7cm) to forage more than were surveyed across the units ($p < 0.01$). Pygmy nuthatches also used Gambel oak to forage less than were surveyed across the units ($p < 0.01$). Of note, pygmy nuthatches had a slightly significant difference in the use of trees with bark beetle activity and trees surveyed across the units when Gambel oak is included ($X^2 = 4.02, p = 0.05$). They foraged on trees with bark beetle activity more than trees surveyed across the units ($p = 0.04$). Including the Gambel oak, which is not used by pygmy nuthatches very much for foraging, may reduce the proportion of trees with bark beetle activity enough to make this significant, since bark beetles do not attack this species.

Discussion

During the first two winters following prescribed fire, there were more bark beetles in the burn units than the control units. Breece (2006) conducted research on all four of the Birds and Burns Network sites in the Southwest, and found similar numbers of bark beetles in the burn units (13%). Bradley and Tueller (2001) had even higher levels of bark beetle attack (24%) after prescribed fire in the Sierra Nevadas. This indicates that even low-intensity fires can attract bark beetles to an area with burned trees. Bark beetles overwintering under the bark of ponderosa pine can provide a food source for bark-foraging birds during this critical time.

Although hairy woodpeckers, pygmy nuthatches and white-breasted nuthatches are known to eat bark beetles in winter (Beal 1911, Stallcup 1968, Anderson 1976), only hairy woodpeckers appear to focus on this winter food source after prescribed fire in the ponderosa pine forests of northern Arizona. Hairy woodpecker density was five times greater in the areas that were treated by prescribed fire. Also, hairy woodpeckers used trees that showed evidence of bark beetle activity in greater proportion than trees surveyed across the units. Pygmy and white-breasted nuthatches did not have any difference in density or the use of trees with bark beetle activity between units treated by prescribed fire and the control units. This could be due to nuthatches having a more diverse diet in the winter, including seeds cached in the bark of trees and insects gleaned from bark fissures and needles (Norris 1958, Stallcup 1968, Anderson 1976, McEllin 1979, *personal observation*).

Blake (1982) compared winter bird communities in areas burned by wildfire and unburned areas in northern Arizona. In his study, hairy woodpeckers were more common in the burned areas. The same was true in this study, even though the fire

was of lower intensity. He also found that species that search bark crevices for insects, such as nuthatches, were more common in the unburned sites in winter. This was not the case in this study, since nuthatches had similar densities in the burned and unburned areas. This may be due to the fact that low-intensity prescribed fire does not alter forest stand structure as much as crown fire.

After wildfire, hairy woodpeckers will remain in the burned areas for several years, declining over time as food sources are depleted (Kreisel and Stein 1999, Covert 2003). However, with prescribed fire, not all trees are killed, so some remain susceptible to bark beetle attack for longer periods of time as broods of bark beetles fly to new trees. Therefore, I speculate that the effect of prescribed fire on bark beetle activity could last longer into the future than with wildfire. Of course, there will be a time when the trees weakened by prescribed fire have died and the remaining trees are healthier, at which time the bark beetles should return to endemic population levels.

Forest managers should be aware that an increase in bark beetle activity might follow the use of prescribed fire. Yet not all bark beetles that are attracted to burned trees actually kill the trees, such as *D. valens*. These bark beetles were fairly common on the study sites and may represent a large portion of the trees with signs of bark beetle activity. In fact, there was low mortality of ponderosa pine (8% on the burn units) associated with bark beetle attacks for sites in Arizona and New Mexico (Breece 2006). Even low-intensity fires, such as prescribed fire, can have a positive short-term effect for hairy woodpeckers in northern Arizona by providing an additional winter food source in the years immediately following the prescribed fire. However, it is worth noting that populations of pygmy and white-breasted nuthatches

were not negatively affected by prescribed fire, since their densities remained similar on burn and control units.

Consequently, the availability of bark beetles in the winter may lead to higher survival for individuals, and thus higher population levels for species that utilize this additional food source. Since prescribed fire can positively affect hairy woodpeckers, it can also have a positive effect for secondary cavity nesting birds in the area, such as western bluebirds and nuthatches, that rely on hairy woodpeckers and other primary cavity nesters to provide nesting habitat. Although an initial positive effect for the nuthatches was not detected in this study, future studies in the area may detect a delayed positive response once these species begin using nesting cavities abandoned by the hairy woodpeckers.

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Table 1. Description of Bird and Burn Network study units on the Coconino and Kaibab National Forests, including unit name, treatment (burn and control), area (ha), number of point count stations, and dates of prescribed fire treatment.

National Forest	Unit Name	Treatment	Area (ha)	# count stations	Dates burned
Kaibab	Kendrick	Burn	369	40	10/27/03 11/6/03 3/25/04
Kaibab	Moritz	Control	360	40	NA
Kaibab	Beale	Control	127	10	NA
Coconino	Imax	Burn	405	40	9/15/03 9/18/03 9/19/03
Coconino	Buck Mtn.	Control	404	40	NA

Table 2. Description of the fire effects of the prescribed fires on each of the burn units and combined for both burn units. Factors listed include the average (+/- SE) maximum bole char height, the average (+/- SE) percent of the needles scorched, and the average (+/- SE) percent of the bole circumference that was charred at the base.

	Imax	Kendrick	Combined Sites
Max bole char ht	0.7m +/- 0.02	2.6m +/- 0.11	1.2m +/- 0.04
Percent scorch	2.3 +/- 0.29	17.6 +/- 1.05	6.6 +/- 0.39
Percent of bole char	56.9 +/- 1.00	85.8 +/- 1.01	65.0 +/- 0.81

Table 3. Three models (uniform cosine, half-normal cosine, and hazard-rate cosine) were compared for hairy woodpeckers (HAWO), pygmy nuthatches (PYNU), and white-breasted nuthatches (WBNU) and the best one was chosen using Δ AIC. Models with Δ AIC <2.0 were averaged for the density estimates. If not, the model designated with an * was selected. All densities are given per 100ha.

Species	Model	ΔAIC	AIC	\hat{D} burn	\hat{D} control	%CV burn	%CV control	\hat{p} burn	\hat{p} control
HAWO	Uniform cosine	0.0	412.4	10.8	2.1	18.5	14.0	0.5	1.0
	Half-normal cosine	1.0	413.4	10.1	1.2	19.4	40.4	0.6	1.7
	Hazard-rate cosine	2.9	415.3	9.7	2.5	27.1	16.8	0.6	0.8
WBNU	Uniform cosine	0.0	997.0	9.8	9.3	9.6	8.9	1.0	1.0
	Half-normal cosine	0.7	997.7	10.4	11.6	16.1	12.3	0.9	0.8
	Hazard-rate cosine	4.8	1001.8	10.2	11.6	19.6	12.3	1.0	0.8
PYNU	Uniform cosine	30.7	1733.3	20.5	18.1	13.1	14.5	0.8	0.8
	*Half-normal cosine	0.0	1702.6	45.0	39.7	16.7	17.8	0.4	0.4
	Hazard-rate cosine	31.2	1733.8	18.4	16.3	12.0	13.5	0.9	0.9

Table 4. Factors included in log-linear analysis of foraging observations for hairy woodpecker, pygmy nuthatch, and white-breasted nuthatch during the winters of 2004 – 2006 using SPSS for Windows Release 10.1.3.

Factor	Range	Description
Bark Beetles	0	No bark beetles
	1	Bark beetle activity
Tree Species	1	Ponderosa pine (<i>Pinus ponderosa</i>)
	2	Gambel oak (<i>Quercus gambelii</i>)
Maximum Bole Char Height	0	None
	1	0.5 – 3.0 m
	2	> 3.0 m
DBH (based on VSS Class)	1	0 – 12.7 cm
	2	12.8 – 45.7 cm
	3	> 45.7 cm

Table 5. Factors selected by log-linear analysis as being significant for hairy woodpecker, pygmy nuthatch, and white-breasted nuthatch foraging behavior in northern Arizona during the winters of 2004 – 2006.

Species	Significant Factors
Hairy Woodpecker	Maximum bole char height*Tree species Bark beetles*Tree Species DBH*Maximum bole char height
Pygmy Nuthatch	Maximum bole char height Bark beetles DBH Tree species
White-breasted Nuthatch	Bark beetles*Tree species Bark beetles*Maximum bole char height DBH

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Figure 1. Map of the Coconino National Forest study area (Imax) with the number of trees at each point center showing signs of bark beetle activity in the circles (from light to dark) overlaid with foraging event locations for hairy woodpeckers (HAWO), pygmy nuthatches (PYNU), and white-breasted nuthatches (WBNU).

Figure 2. Map of the Kaibab National Forest study area (Kendrick) with the number of trees at each point center showing signs of bark beetle activity in the circles (from light to dark) overlaid with foraging event locations for hairy woodpeckers (HAWO), pygmy nuthatches (PYNU), and white-breasted nuthatches (WBNU). The “No Trees” category represents point centers where no trees were located, and therefore no bark beetle activity or foraging event locations.

Figure 3. The proportion of trees used for foraging by hairy woodpeckers (HAWO), pygmy nuthatches (PYNU), and white-breasted nuthatches (WBNU) that show evidence of bark beetle activity on burn and control units.

Figure 4. The proportion of trees with evidence of bark beetle activity used for foraging in the burn units by hairy woodpeckers (HAWO), pygmy nuthatches (PYNU), and white-breasted nuthatches (WBNU) and surveyed across the burn units.

Figure 5. The proportion of trees with evidence of bark beetle activity used for foraging in the control units by hairy woodpeckers (HAWO), pygmy nuthatches (PYNU), and white-breasted nuthatches (WBNU) and surveyed across the control units.

Figure 1.

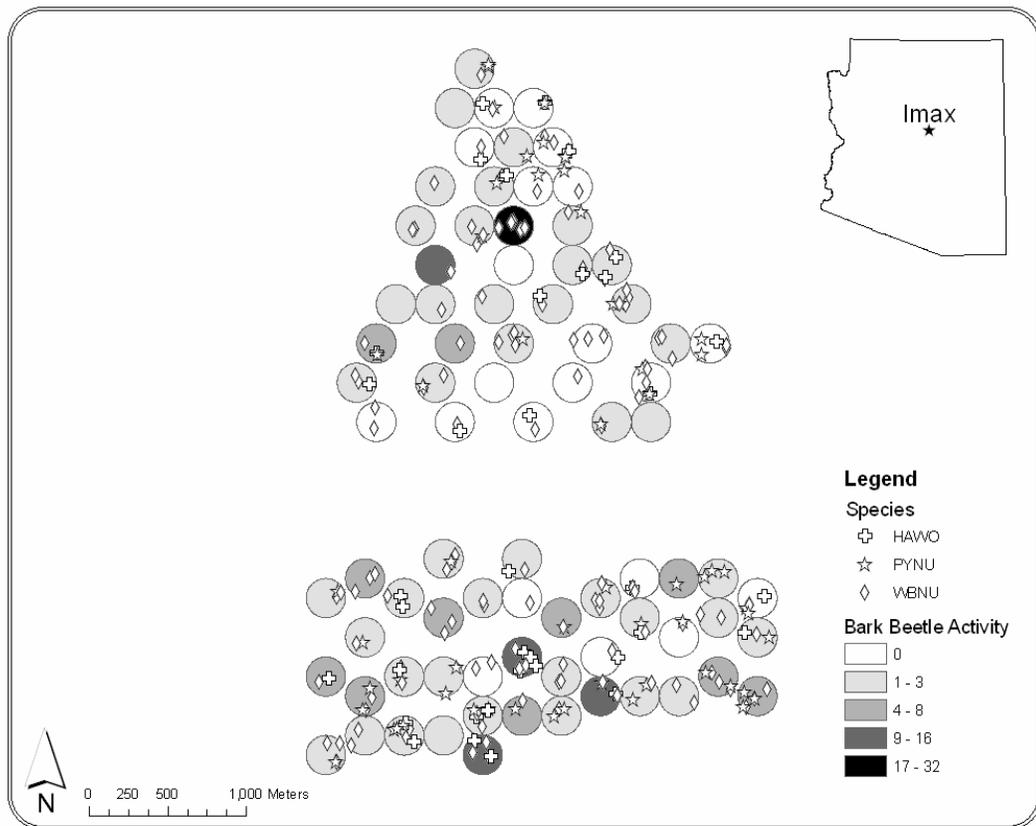


Figure 2.

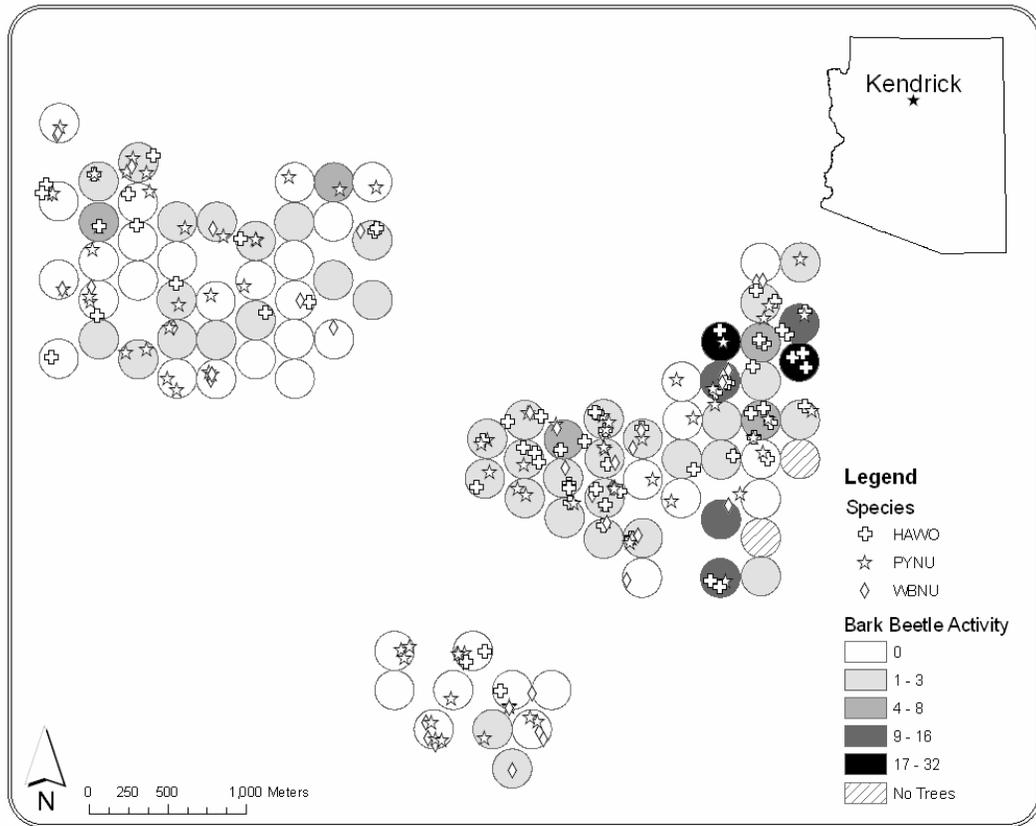


Figure 3.

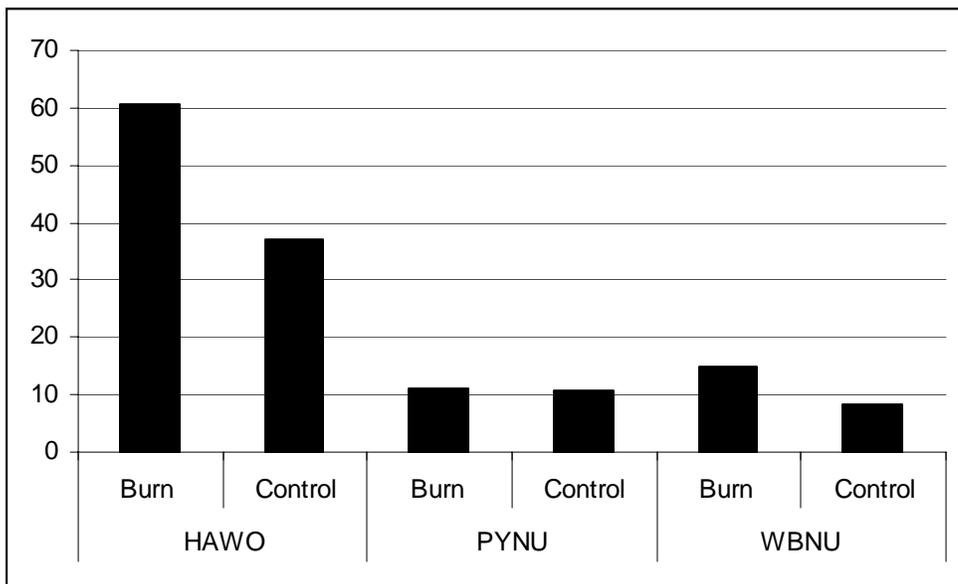


Figure 4.

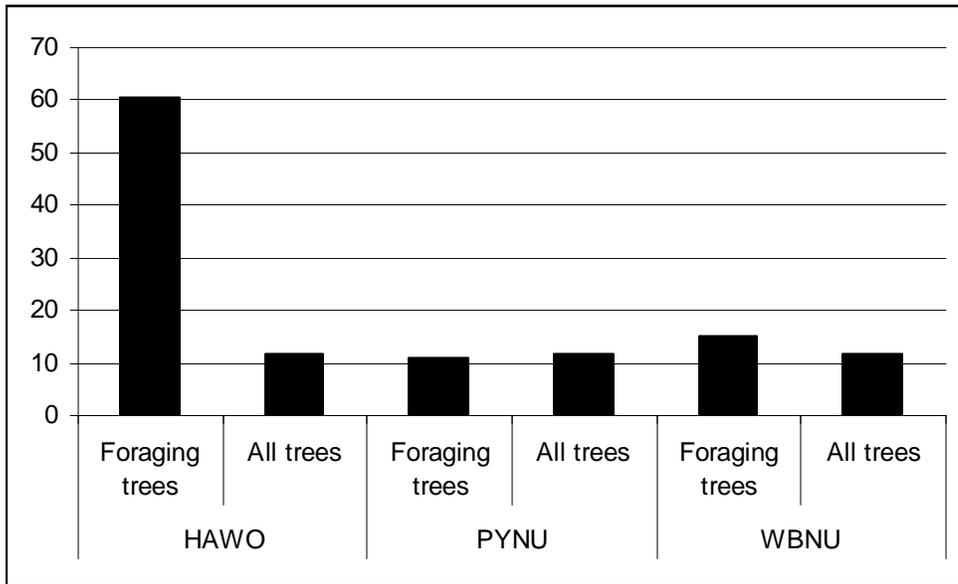
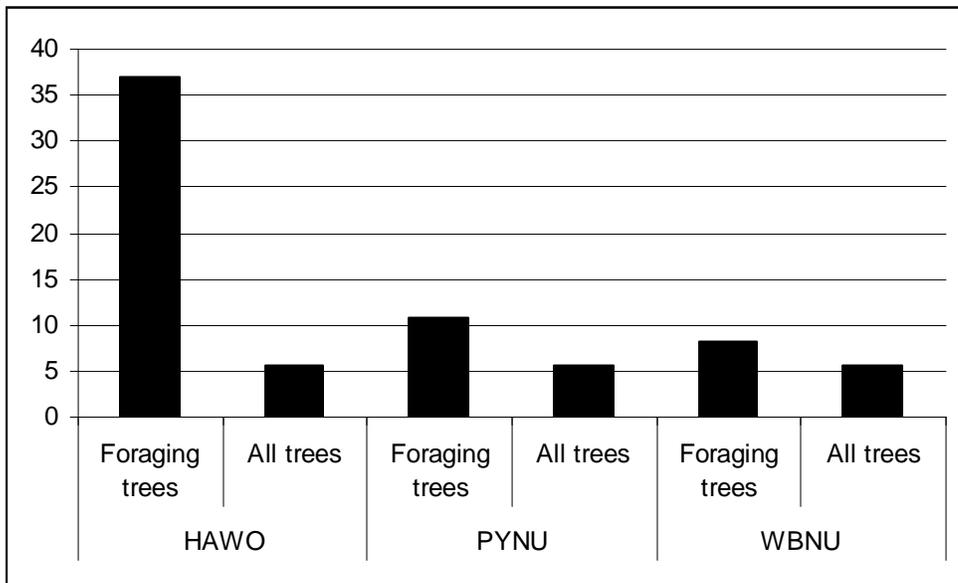


Figure 5.



CHAPTER 3

WINTER BIRD COMMUNITIES IN PONDEROSA PINE FORESTS OF NORTHERN ARIZONA

Abstract

Winter bird studies are rare in wildlife ecology. However, winter can be a critical time for birds. This study examines winter bird communities in the ponderosa pine (*Pinus ponderosa*) forests of northern Arizona. Point counts were conducted on two sites in northern Arizona from mid-October to mid-March 2004 – 2006. Each site had one unit treated by prescribed fire a full growing season before the point counts began, paired with control unit(s) of similar forest structure. Thirty-nine bird species were detected during the survey period. Nine species comprised 81% of all detections; eight of these were year-round residents of the area. Dark-eyed Juncos (*Junco hyemalis*) were the most numerous, comprising 23% of all detections. The bird communities were very similar between treatments ($C = 0.85$) and years ($C = 0.85$), and the rank abundance of species between burn and control units were correlated (Spearman's rank correlation coefficient was $\rho = 0.83$). Therefore, winter bird communities were similar among areas treated by prescribed fire and unburned areas of the ponderosa pine forests in northern Arizona.

Introduction

Although most studies of avian ecology occur during the breeding the season, winter ecology is also very important to bird populations. First, food is generally

limited in the winter; as insects are not as abundant and many plants are dormant during this time. As a result, food is more patchily distributed and birds are more opportunistic in their foraging ecology. Second, conditions are harsher during winter, with lower temperatures and greater snow cover. Therefore, species are more vulnerable to physiological stress and may not survive winter conditions. Winter survival can have an effect on populations, since birds that survive the winter will be able to reproduce in the breeding season. Also, habitat requirements may be different in the winter, since habitat required for breeding may not be the same as habitat needed in the winter for providing food and shelter from the harsh conditions.

Yet there is a paucity of studies on the winter ecology of birds. While numerous studies exist investigating the effects of wildfires on birds (see Saab and Powell 2005), few studies exist on the effects of prescribed fire on birds (Bock and Bock 1983, Horton and Mannan 1988, Dickson 2006). Studies investigating the effects of fire on wintering birds are few (Blake 1982, Kreisel and Stein 1999, Covert 2003, Bock and Block 2005), especially for prescribed fire (King et al. 1998). Few studies of wintering bird communities have occurred in northern Arizona (Haldeman et al. 1973, Blake 1982, Bock and Block 2005). This study compares winter bird communities in areas recently treated by prescribed fire with untreated controls. Data collection began the winter after the first growing season following prescribed fire treatment.

Study Area

My research was conducted on the Birds and Burns Network study sites in northern Arizona. The U.S. Forest Service Rocky Mountain Research Station directs

the Birds and Burns Network, a collaborative effort examining fire effects on populations and habitats of breeding birds in ponderosa pine forests in eight states across the western United States. Collaborators include a number of national forests, Pacific Northwest Research Station, The Nature Conservancy and several universities. The goal of the project is to understand the ecological consequences of fire management for breeding birds in ponderosa pine forests of the Interior West.

The northern Arizona study sites were located in the Coconino and Kaibab National Forests, 60 km southeast and 35 km northwest of Flagstaff, respectively. Ponderosa pine dominated the overstory on the Coconino study units, with Gambel oak (*Quercus gambelii*) contributing to canopy structure. Alligator juniper (*Juniperus deppeana*) is also found on the Coconino units. Ponderosa pine was also the dominant overstory tree on each of the Kaibab study units. Pinyon pine (*P. edulis*) and one-seed (*J. monosperma*) and alligator (*J. deppeana*) junipers occurred on the control units, but contributed little to stand canopies. Open grassland patches on both sites are dominated by bunchgrass species, including Arizona fescue (*Festuca arizonica*) and blue gramma (*Bouteloua gracilis*). Topography on the Coconino varies from flat to steep hills, with elevations ranging from 2070m to 2160m, while the Kaibab is relatively flat, with elevations ranging from 2100m to 2300m.

Each study site had a burn treatment unit paired with control unit(s) of similar forest structure (Table 1). The Kaibab study area needed two control units to capture the large tree component that was present in the treatment unit. Treatment units were chosen in consultation with district fire managers on each forest. Control units were then placed in representative areas with similar stand structures within 1 km of the treatment unit where no management was planned by the forests. A systematic

random sampling design was used for the placement of point count stations. The placement of the first point was randomized, with the remaining points assigned using a GIS algorithm (see Dickson 2006). U.S. Forest Service personnel conducted the prescribed fires during Fall 2003 on the Coconino treatment unit and on the Kaibab treatment unit during Fall 2003 and Spring 2004.

All prescriptions were characterized as broadcast burns with expected fire behaviors of low to moderate intensity (see Dickson 2006). These fires were heterogeneous in nature, with areas that were not burned at all, as well as areas with burns severe enough to kill some trees. The average (\pm SE) maximum bole char height for the two burn units was 1.2m \pm 0.04. The average (\pm SE) percent of the bole circumference that was charred at the base was 65.0 \pm 0.81. And the average (\pm SE) percent of needles scorched was 6.6 \pm 0.39. Since I was investigating general relationships between prescribed fire effects and winter bird communities, I pooled data across the burn units for analyses.

Methods

I conducted winter point counts on the Birds and Burns Network study sites in northern Arizona. The Coconino study area had 40 point count stations each in the burn and control units. The Kaibab study area had 40 point count stations in the burn unit and 50 point count stations between the two control units. Therefore, there were a total of 170 point count stations (80 in the burn units and 90 in the control units) (Table 1). Each station was approximately 300m apart and at least 200m from unit edges. At each station, I recorded number of individuals, sex (if known), and type of detection of all birds observed during a 5-minute survey period. Point counts began

within 30 minutes of sunrise and concluded within 5 hours. I did not count in windy or wet (more than a light snow) conditions. I counted each station eight times between mid-October and mid-March, 2004 – 2006.

To describe the general winter bird community patterns, I used a similarity index and a test of rank order correlation of abundances. The similarity of winter bird communities between treatments (pooling across years) and years (pooling across treatments) was investigated using the Sorenson similarity index, $C = 2j/(a + b)$, where j = number of species common to both treatment units/years, a = total number of species detected in the burn unit/year 1, and b = total number of species detected in the control unit/year 2 (Magurran 1988). The similarity equals a number between 0 and 1, with the higher value representing greater similarity. This value represents how similar the winter bird communities were with respect to species composition in each treatment and year.

While Sorenson similarity index investigated the similarity of species composition between treatments and years, the rank order of abundance represents some aspects of community structure for each year and treatment. The species were ranked in order of abundance, based on the number of individual detections. Spearman's rank order correlation coefficient (ρ) (Conover 1999) was calculated using SPSS for Windows Release 10.1.3. A higher value of ρ represents a higher correlation in the rank order of species between treatments and years. For example, $\rho = 1.00$ will have the species ranked in the same order for each treatment or year.

Results

Thirty-nine bird species were detected during the winters of 2004 – 05 and 2005 – 06 (Table 2). Five of these species were incidental (observed in the unit but not within 100m of any point count station). All other species were assigned to one of four foraging groups, as long as the foraging group was represented by more than one species. These groups were seed-eating (e.g. Dark-eyed Junco), bark-foraging/sapsucking (e.g. Hairy Woodpecker), gleaning insectivores (e.g. Western Bluebird), and generalists (e.g. Steller's Jay).

Twelve species were detected only during one field season. Of these, ten of the species only occurred as incidentals or were only detected one or two times. The exceptions were American Robin (*Turdus migratorius*) and Violet-green Swallow (*Tachycineta thalassina*). Violet-green Swallows were only detected during the first season because counts went longer into March and the swallows had returned from migration at that point. American Robins were prevalent during the first season yet were not detected during the second season. Also, eight of the twelve species were only detected the first year, two species in both units, five in the burn units only and one in the control unit only. Of the four species only detected in year 2, three were detected in the burn only and one in the control only.

Nine species comprised the major percentage of observations (81%). In descending order, these were Dark-eyed Junco (*Junco hyemalis*), Pygmy Nuthatch (*Sitta pygmaea*), Western Bluebird (*Sialia mexicana*), White-breasted Nuthatch (*S. carolinensis*), Mountain Chickadee (*Poecile gambeli*), Ruby-crowned Kinglet (*Regulus calendula*), Steller's Jay (*Cyanocitta stelleri*), Hairy Woodpecker (*Picoides villosus*), and Brown Creeper (*Certhia americana*). All of these species, except for

Ruby-crowned Kinglet are year-round residents of northern Arizona ponderosa pine forests. Dark-eyed Juncos represented the greatest number of detections during the study. Although they were not recorded to subspecies, several subspecies were observed in winter in addition to the local breeding Gray-headed (*J. h. caniceps*) and Red-backed (*J. h. dorsalis*) subspecies, including Pink-sided (*J. h. mearnsi*), Slate-colored (*J. h. hyemalis*) and Oregon subspecies (*J. h. oregonus*) (*personal observation*).

I found that the bird communities were fairly similar in composition and structure between years and between burn and control units. Using the Sorenson similarity index (C) to examine all species detected during point counts, except for the incidental detections, the similarity between treatments was $C = 0.85$. The similarity between years was also $C = 0.85$. Therefore, the species that were detected were very similar between treatment units and years. This pattern continued when looking at individual foraging groups. The seed-eating group had $C = 1.00$ between treatments and $C = 0.80$ between years. Gleaning insectivores had $C = 0.95$ between treatments and $C = 0.89$ between years. Bark-foraging/sapsucking birds had $C = 0.89$ for both treatment and year. The generalists had $C = 0.85$ between years, however, they had $C = 0.66$ between treatments. This is because two of the four species, Pinyon Jay (*Gymnorhinus cyanocephalus*) and American Crow (*Corvus brachyrhynchos*), were only detected in the control units.

Spearman's rank order correlation coefficient (ρ) was calculated for each treatment between years for all non-incidental species. In the burned unit, $\rho = 0.69$ ($p < 0.01$). In the control unit, $\rho = 0.70$ ($p < 0.01$). Since there were no yearly differences, both years were combined for a $\rho = 0.83$ ($p < 0.01$) correlation between

treatments. Therefore, when looking at all non-incidental species, there were no real differences in the structure of winter bird communities between the burn and control units.

However, when looking at individual foraging groups, there were some differences between years in each of the treatment units. Seed-eating birds had $\rho = 0.50$ ($p = 0.92$) between years in the burn units and $\rho = 1.00$ ($p < 0.01$) in the control units. This group had $\rho = 0.50$ ($p = 0.92$) correlation between treatments for both years combined. The generalist group had $\rho = 1.00$ ($p < 0.01$) in the burn units and the control units between years, and $\rho = 0.63$ ($p = 0.37$) correlation between treatments. The gleaning insectivore group had $\rho = 0.47$ ($p = 0.17$) in burn units, $\rho = 0.28$ ($p = 0.43$) in control units, yet $\rho = 0.91$ ($p < 0.01$) correlation between treatments. The bark-foraging/sapsucking group had $\rho = 0.95$ ($p < 0.01$) in the burn units, $\rho = 0.96$ ($p < 0.01$) in the control units, and $\rho = 0.92$ ($p < 0.01$) correlation between treatments.

Discussion

Blake (1982) compared winter bird communities in areas burned by wildfire and unburned areas in northern Arizona. In his study, Hairy Woodpeckers were more common in the burned areas. The same was true in this study, even though the fire was of lower intensity. He also found that species that search bark crevices for insects, such as nuthatches, were more common in the unburned sites in winter. This was not the case in this study, since nuthatches had similar abundances in the burned and unburned areas. This may be due to the fact that low-intensity prescribed fire does not alter forest stand structure as much as crown fire.

Haldeman et al. (1973) recorded 18 species wintering in ponderosa pine forests of northern Arizona in undisturbed sites. Bock and Block (2005) found a similar trend in species numbers during the nonbreeding season, with 26 species in the unburned forests. Each is lower than the 31 species detected in the unburned areas of this study. However, this study and Bock and Block (2005) both had 35 species detected in areas of moderate or prescribed burns. The most common species Haldeman et al. (1973) reported was the Pygmy Nuthatch, which was second most common species behind Dark-eyed Juncos in this study. However, the other most common species were the same for both studies, including Mountain Chickadee, White-breasted Nuthatch and Western Bluebird.

In this study, 25 of the 34 species were common between treatments and years. The Sorenson similarity index (C) between years and treatments was $C = 0.85$ for all the bird species combined. When divided into foraging groups, Sorenson similarity indices remained high, with the lowest of the indices ($C = 0.66$) for generalists between treatments. The rank abundance of species between burn and control units were also correlated (Spearman's rank order correlation coefficient was $\rho = 0.83$). The gleaning insectivore and bark-foraging/sapsucking groups appear to have more influence over the Spearman's rank correlation coefficient due to the number of species in each group. The bark-foraging/sapsucking foraging group was the most similar between the burn and control units, with $\rho = 0.92$. The seed-eating species were the same for both units between years, however, Pine Siskin and Red Crossbill switched in rank order of abundance in the second year. Though the rank order of abundance for the generalist species was the same in each treatment unit, Pinyon Jays and American Crow were only detected in the control units. The greatest

variability in rank order of abundance occurred in the gleaning insectivore foraging group. While most of the species were found in both units and both years, there were a few that were only present in one year or treatment. Yet, the species with the highest rank in order of abundance between treatment units were the same each year (Year 1 = American Robin, Mountain Chickadee, and Western Bluebird; Year 2 = Western Bluebird, Ruby-crowned Kinglet, and Mountain Chickadee), with two of these species being ranked in the top three both years (Western Bluebird and Mountain Chickadee). Therefore, bird communities in northern Arizona were similar in composition and structure among treatments and years during the first two winters following a full growing season after low-intensity prescribed fire treatments.

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Table 1. Description of Bird and Burn Network study units on the Coconino and Kaibab National Forests, including unit name, treatment (burn and control), area (ha), number of point count stations, and dates of prescribed fire treatment.

National Forest	Unit Name	Treatment	Area (ha)	# count stations	Dates burned
Kaibab	Kendrick	Burn	369	40	10/27/03 11/6/03 3/25/04
Kaibab	Moritz	Control	360	40	NA
Kaibab	Beale	Control	127	10	NA
Coconino	Imax	Burn	405	40	9/15/03 9/18/03 9/19/03
Coconino	Buck Mtn.	Control	404	40	NA

Table 2. List of all species detected during point counts from mid-October to mid-March, 2004-2006, including incidental species (I) that were only detected beyond 100-m radius of point count stations. Species, scientific name, foraging group, number of detections, percent of total detections, year(s) detected and treatment(s) detected are listed. Incidental species and those that are the only representative of their foraging group were not assigned to foraging groups analyzed in rank order of abundance.

Species	Scientific name	Foraging group	# of detections (w/in 100m)	% of total detections (n=4639)	Year(s) detected	Treatment(s) detected
Bald Eagle	<i>Haliaeetus leucocephalus</i>	-	1	-	both	control only
Northern Goshawk	<i>Accipiter gentiles</i>	-	1	-	1	burn only
Red-tailed Hawk	<i>Buteo jamaicensis</i>	-	2	0.04	both	both
Swainson's Hawk	<i>Buteo swainsoni</i>	-	1	-	1	burn only
Wild Turkey	<i>Meleagris gallopavo</i>	-	3	0.06	both	both
Mourning Dove	<i>Zenaida macroura</i>	-	1	0.02	2	burn only
Long-eared Owl	<i>Asio otus</i>	-	1	0.02	2	control only
Great Horned Owl	<i>Bubo virginianus</i>	-	1	0.02	1	burn only
Northern Pygmy-owl	<i>Glaucidium gnoma</i>	-	1	0.02	1	burn only
Northern Flicker	<i>Colaptes auratus</i>	Bark-foraging/ Sapsucking	62	1.34	both	both
Williamson's Sapsucker	<i>Sphyrapicus thyroideus</i>	Bark-foraging/ Sapsucking	11	0.24	both	both

Red-naped Sapsucker	<i>Sphyrapicus nuchalis</i>	Bark-foraging/ Sapsucking	7	0.15	both	both
Downy Woodpecker	<i>Picoides pubescens</i>	Bark-foraging/ Sapsucking	2	0.04	1	burn only
Hairy Woodpecker	<i>Picoides villosus</i>	Bark-foraging/ Sapsucking	160	3.45	both	both
Three-toed Woodpecker	<i>Picoides tridactylus</i>	Bark-foraging/ Sapsucking	9	0.19	both	both
Steller's Jay	<i>Cyanocitta stelleri</i>	Generalist	176	3.79	both	both
Clark's Nutcracker	<i>Nucifraga columbiana</i>	-	1	-	2	burn only
Pinyon Jay	<i>Gymnorhinus cyanocephalus</i>	Generalist	28	0.60	both	control only
American Crow	<i>Corvus brachyrhynchos</i>	Generalist	2	0.04	both	control only
Common Raven	<i>Corvus corax</i>	Generalist	50	1.08	both	both
Violet-green Swallow	<i>Tachycineta thalassina</i>	-	8	0.17	1	both
Mountain Chickadee	<i>Poecile gambeli</i>	Gleaning Insectivore	405	8.73	both	both
Bushtit	<i>Psaltriparus minimus</i>	Gleaning Insectivore	78	1.68	both	both
Brown Creeper	<i>Certhia americana</i>	Bark-foraging/ Sapsucking	133	2.87	both	both
White-breasted Nuthatch	<i>Sitta carolinensis</i>	Bark-foraging/ Sapsucking	406	8.75	both	both
Red-breasted Nuthatch	<i>Sitta canadensis</i>	Bark-foraging/ Sapsucking	2	0.04	2	burn only
Pygmy Nuthatch	<i>Sitta pygmaea</i>	Bark-foraging/ Sapsucking	678	14.62	both	both

Golden-crowned Kinglet	<i>Regulus satrapa</i>	Gleaning Insectivore	34	0.73	both	both
Ruby-crowned Kinglet	<i>Regulus calendula</i>	Gleaning Insectivore	220	4.74	both	both
Western Bluebird	<i>Sialia mexicana</i>	Gleaning Insectivore	508	10.95	both	both
Mountain Bluebird	<i>Sialia currucoides</i>	Gleaning Insectivore	1	0.02	1	control only
Townsend's Solitaire	<i>Myadestes townsendi</i>	Gleaning Insectivore	10	0.22	both	both
American Robin	<i>Turdus migratorius</i>	Gleaning Insectivore	84	1.81	1	both
Yellow-rumped Warbler	<i>Dendroica coronata</i>	Gleaning Insectivore	32	0.69	both	both
Olive Warbler	<i>Peucedramus taeniatus</i>	Gleaning Insectivore	16	0.34	both	both
Dark-eyed Junco	<i>Junco hyemalis</i>	Seed-eating	1065	22.96	both	both
Cassin's Finch	<i>Carpodacus cassinii</i>	-	1	-	both	both
Red Crossbill	<i>Loxia curvirostra</i>	Seed-eating	38	0.82	both	both
Pine Siskin	<i>Carduelis pinus</i>	Seed-eating	32	0.69	both	both

Table 3. List of Spearman's rank order correlation coefficients (ρ) for each foraging group (seed-eating, generalist, gleaning insectivores, and bark-foraging/sapsucking) and all non-incidental species between years for each treatment and between treatments (both years combined).

Foraging Group	Burn Unit Between Years	Control Unit Between Years	Both Years Between Treatments
Seed-eating	0.50 ($p = 0.92$)	1.00 ($p < 0.01$)	0.50 ($p = 0.92$)
Generalist	1.00 ($p < 0.01$)	1.00 ($p < 0.01$)	0.63 ($p = 0.37$)
Gleaning Insectivore	0.47 ($p = 0.17$)	0.28 ($p = 0.43$)	0.91 ($p < 0.01$)
Bark-foraging/Sapsucking	0.95 ($p < 0.01$)	0.96 ($p < 0.01$)	0.92 ($p < 0.01$)
All species	0.69 ($p < 0.01$)	0.70 ($p < 0.01$)	0.83 ($p < 0.01$)