

**EFFECTS OF SNOWPACK ON AVIAN ABUNDANCE AND
SPECIES RICHNESS IN SIERRA NEVADA FORESTS**

by

TYLER HULL STUART

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This thesis for Master of Arts degree by
Tyler Hull Stuart
has been approved for the
Department of Geography and Environmental Studies
by

David G. Havlick, Chair

Curtis D. Holder

Steven A. Jennings

Date

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CHAPTER I

INTRODUCTION

Over 50% of bird species populations in the Sierra Nevada, California may be declining (Siegel and DeSante, 1999), based on analysis of Breeding Bird Survey routes and MAPS stations (Monitoring Avian Productivity and Survivorship). Of seventy-seven species that were detected with high enough frequency to analyze, forty (52%) show decreasing trends, and eighteen (23%) of these are significantly decreasing. Notably, many of the species that were not analyzed by Siegel and DeSante (1999) because of low detection rates are already known to be imperiled (e.g., California Spotted Owl and Great Gray Owl).

Each animal species serves an important function in the dynamics of an ecosystem, and the functions of the suite of birds in the Sierra Nevada parallel the remarkable diversity of this taxon. As predators of small mammals, fish, reptiles, and other birds; prey for large mammals, snakes, and other birds; scavengers; insect control; pollinators; seed dispersers; and a food source for decomposers, the ecosystem services of birds are significant. To name but a few consequences of declines in specific avian guilds, we would see reduced plant recruitment, increases in insect and rodent outbreaks, and increases in carcasses and associated plagues (Sekercioglu *et al.*, 2004).

Purpose of the Study

The National Park Service (NPS) has partnered with the Institute for Bird Populations (IBP) and their personnel to fulfill the following objectives for SIEN Landbird Monitoring, as stated in Siegel *et al.* (2010):

1. Detect trends in density of bird species that are monitored well by point counts (including most passerines, near-passerines, and galliformes), throughout accessible areas of Sierra Nevada Network parks, during the breeding season.
2. Track changes in breeding-season distribution of bird species throughout accessible areas of Sierra Nevada Network parks.

This study examined avian population dynamics over a four year span (2011-2014) in four forest types in Kings Canyon, Sequoia, and Yosemite National Parks, California. I related each year's population values to the preceding season's snowpack, specifically focusing on spring snowpack (April monthly mean). According to the California Department of Water Resources, the snowpack on April 1 each year has particular hydrologic significance because snow tends to accumulate "up to April 1, and then melts thereafter" (Cunningham, 2009). In addition, many of the bird species that breed in the Sierra Nevada begin to arrive from wintering grounds and establish territories during April. In this study, I focused on the bird species of four forest habitat types: Sierran mixed conifer, red fir, lodgepole pine, and subalpine conifer. Some portions of the Sierra Nevada have a distinct, pure white fir zone between Sierran mixed conifer and red fir in elevation, and white fir also occurs to varying degrees within Sierran mixed conifer forest. Although the white fir zone does occur in my study areas, I did not include it in this study because very few avian surveys were conducted within pure white fir. Within the four habitats of focus, I measured avian abundance and species richness.

Problem Statement

My research question is: do bird populations show fluctuations in abundance and species richness depending on early spring snowpack levels in various forest types of the Sierra Nevada? Review of related literature exposes that most studies do not examine population fluctuations across a broad range of habitats, with only one study (Purcell, 2002) assessing avian population values as related to winter weather in four forest types of the Sierra Nevada. Purcell (2002) found fifteen species that shifted downward in elevation and six species with upslope movements after two

severe winters. Similarly, numerous bird species in Yosemite National Park showed higher productivity in years with less spring snowpack (Saracco *et al.*, *unpublished*, cited in Siegel *et al.*, 2014). The present study investigated a different set of four forest types, with a focus on abundance and species richness rather than avian productivity.

Avian abundance is defined for this study as the calculated number of individuals of a species or all species combined along a point transect or within a habitat type based on the program, DISTANCE (*see Chapter III for explanation of DISTANCE*). Avian species richness is a measure of the raw number of species detected at a given site, or within a habitat type or study year.

Based on previous studies related to snowpack in the Sierra Nevada and other locations (Hejl *et al.*, 1988; Purcell, 2002; Hendricks, 2003), I hypothesize that avian abundance and species richness will decrease in higher elevation forests (red fir, lodgepole pine, and subalpine conifer) in years of high early spring snowpack. Sierran mixed conifer forest will show slight increases in avian abundance and species richness in years of high snowpack as a result of downslope migration from the higher elevation forests. Years of lower snowpack will show higher values in the upper elevation forests and slightly lower values in Sierran mixed conifer forest, relative to years of high snowpack. However, a threshold snowpack may exist, such that winters with extremely low snowpack, such as early 2013, show reduced bird populations in all forest types, perhaps as a result of limited food availability and less abundant vegetation for nesting of some species (Bolger *et al.*, 2004).

This thesis addresses current issues within the fields of biogeography and avian ecology as well as aiding to direct the focus of future studies of birds, their environment, and climate change within the Sierra Nevada and beyond. A review of literature and placement of this study in relation to previous studies follows in Chapter II. Chapter III addresses the methodologies and analyses used, and Chapter IV presents the results. Finally, Chapter V discusses the findings, including suggestions for improvements and further studies.

CHAPTER II

REVIEW OF THE LITERATURE

Previous research examines the effects of climate change on the distributions of bird species, especially in terms of elevational shifts, and also focuses on the relationships between severe weather, especially winter precipitation and temperatures, and avian population dynamics, including abundance, richness, fecundity, and survival. As climate change alters temperatures, vegetation, and frequency of severe weather in specific habitats, bird distributions will change as well. By understanding avian responses to short-term weather events and longer-term shifts due to climate change, we can better predict how bird populations will change in coming decades.

Climate Change-induced Elevational Shifts

Studies have shown that climate change, specifically increasing temperature, has forced elevational shifts in ranges for birds (Pounds *et al.*, 2005; Shoo *et al.*, 2006; Siegel *et al.*, 2011; Gardali *et al.*, 2012), insects (Wilson *et al.*, 2005; Franco *et al.*, 2006), and other animals and plants (Pounds *et al.*, 1999; Parmesan, 2006; Thomas *et al.*, 2006). Such shifts in range are anticipated to result in the extinction of between 1.3% and 30% of landbirds in the western hemisphere by 2100 (Sekercioglu *et al.*, 2008), and a decline of countless other species to threatened or near-threatened levels. As with any predictive study of climate change effects on plant and animal populations, there is variability in the factors causing the change, and estimates of extinctions and declines vary considerably (Sekercioglu *et al.*, 2004; Sekercioglu *et al.*, 2008; Harris and Pimm, 2008; Gardali *et al.*, 2012). Precision in predictions of decline is likely to increase when species are considered at the local level, but regardless, “quantitatively predicting future extinctions is essential for effective

management” (Harris and Pimm, 2008, 164). Shifts in elevation have ecological, phenological, and biological consequences for birds.

Biological differences between elevations. A single bird species at different elevations can have significantly different biological adaptations for producing and raising offspring (Badyaev and Ghalambor, 2001; Martin and Wiebe, 2004; Bears *et al.*, 2009). For example, the colder temperatures, seasonality, and shorter breeding season of higher elevations result in smaller clutch size, and longer incubation and nestling periods in order for adult birds to contribute more energy per nestling to offspring survival (Badyaev and Ghalambor, 2001). As a result, fledgling Dark-eyed Juncos at “high elevations were *c.* 11% heavier and the average fat score in fledglings was more than double fledglings at low elevations” (Bears *et al.*, 2009, 370). Higher elevation nestlings also had higher survival rates than lower elevation Dark-eyed Junco nestlings (Bears *et al.*, 2009). Lower elevation populations of a species, which tend to lay more eggs at the expense of lower survival rates, may be poorly adapted to raise young at higher elevations. Based on a rate of 5°C/km, meaning the change in temperature per vertical distance moved, and a moderate surface temperature increase estimate of 2.8°C by 2100, a bird species would have to permanently shift its range 560 m upslope to stay within the same vegetation type and temperature regime (Sekercioglu *et al.*, 2008). In many regions, this extreme of a shift is not possible, and this change is particularly threatening to already-high elevation species with low abilities to disperse.

Elevational shifts challenge ecological balances. Fluctuations in ratios of predators, prey, scavengers, and producers due to elevation shifts will undoubtedly have consequences for ecosystem balance (Machmer and Steeger, 1995; Sekercioglu *et al.*, 2004). Foliage-gleaning and bark-foraging birds (e.g., chickadees, woodpeckers, and nuthatches), in particular, help to reduce outbreaks of “pest” insect populations. Density-dependent prey regulation often occurs in predator-prey relationships, such that as prey quantity increases, predator foraging activity and efficiency increase as well (Machmer and Steeger, 1995). Nonetheless, range shifts for these birds, insects, or

the vegetation on which they feed could lead to such imbalances in the quantity of each that insect outbreaks and likely tree mortality would greatly increase. However, a survey of studies found prey mortality caused by bark-foraging birds to vary from 0.5% to 98% (Machmer and Steeger, 1995), depending on how each study measured prey mortality. Machmer and Steeger (1995) explain that the removal of bark by woodpeckers alters microhabitats for their prey, indirectly resulting in mortality. It is difficult to make broad predictions about future consequences of these relationships based on such a wide range of mortality rates, and more studies are needed.

Severe Weather and Bird Population Fluctuations

Because heavy snowfall winters occur on average every five years in the Sierra Nevada (Hejl *et al.*, 1988), it is important to understand avian population dynamics as they relate to severe winters. Current and impending changes in bird distributions and bird-habitat relationships as a result of climate change create a need to study how specific populations interact with adverse weather conditions. Severe winters have resulted in short-term, elevational shifts in breeding species distributions in the Sierra Nevada, California (Purcell, 2002), and this type of shift may create challenges for some species. Purcell (2002) found 18% of the bird species (or seventeen species) she studied occurring in a single habitat type. For example, Purcell found fourteen of her study species occurring only within ponderosa pine forest. This may relate to biological or ecological dependencies on specific habitats (Siegel and DeSante, 1999; Julliard *et al.*, 2004). On the other hand, 35% of species were found in all four habitats in Purcell's study. Purcell noted that presence of a species in a given habitat is not necessarily indicative of a healthy population, though, since presence does not confirm high-quality habitat. Abundance is a better measure of habitat quality (Berg, 1997), or at least of preference. In fact, species occupation of a marginal habitat may exacerbate the effects of severe weather conditions (Raphael *et al.*, 1987). A species inhabiting a marginal habitat for its needs may not find adequate food sources or shelter that it may have

otherwise easily found in higher-quality habitat, resulting in greater challenges in the face of severe weather.

Bird populations often decline in response to unfavorable weather, such as higher snowfall and colder temperatures (Szaro and Balda, 1986; Hejl *et al.*, 1988). For example, the highest snowfall on record for northern Arizona ponderosa pine forests resulted in the lowest densities in a study of local bird populations (Szaro and Balda, 1986). Avian abundance and species richness also decreased in true fir forests of the Sierra Nevada after the winter of 1982-1983 (Hejl *et al.*, 1988; DeSante, 1990), most likely as a result of low survival and recruitment of young birds during the record snowfall that year. The population dips were not a product of adult mortality or reduced fecundity (DeSante, 1990). In both white fir and red fir forests, abundance and species richness increased in 1984 and 1985, though more so in white fir than red fir (Hejl *et al.*, 1988). Interestingly, overall abundance declined from 1983 to 1985 in Sierran mixed-conifer forests, the habitat type immediately downslope from white fir (Morrison *et al.*, 1987). Hejl *et al.* (1988) states that this may suggest a short-term downslope shift (in 1983, when conditions were too harsh at higher elevations), followed by a return of birds to higher habitats after conditions improved (in 1984 and 1985). Morrison *et al.* (1987), Hejl *et al.* (1988), and DeSante (1990) all offer greater application of avian population dynamics in the Sierra Nevada because their studies examined sites over a broad area and are more generally representative of this mountain range.

Forest Structure and Bird Communities in Severe Weather

Forest structure in the Sierra Nevada and northern Rocky Mountains defines the avian composition during breeding and post-breeding seasons (Beedy, 1981; Verner and Larson, 1989; Sallabanks *et al.*, 2006), at least in part because of specific food availability and nesting substrate (Karr and Roth, 1971; Holmes *et al.*, 1979; Verner and Larson, 1989). Total crown volume of trees and shrubs is the best variable for predicting bird species richness in mixed-conifer forests of the Sierra Nevada (Verner and Larson, 1989). Areas of high crown volume, which tend to host higher

species richness, may therefore be important source population locations for many species. Severe winters force specific foraging guilds of birds into suboptimal forest types where food availability does not allow for the same carrying capacity. Ground-foraging species are especially sensitive to forest structure (Beedy, 1981) and vulnerable to diminished food access due to frozen or snow-covered ground (Robinson *et al.*, 2007). Additionally, prolonged low temperatures may lead to increased bird mortality (Graber and Graber, 1979) and reduced reproductive output (Chase *et al.*, 2005). Robinson *et al.* (2007) suggest, however, that the indirect effects of severe winter (e.g. limited food accessibility and delayed onset of nesting) are more commonly the cause of population declines.

In studies of cavity-nesting bird populations in the Sierra Nevada as they are related to precipitation, conclusions are variable (Raphael and White, 1984; Raphael *et al.*, 1987). Raphael and White (1984) found precipitation to be the driving factor in temporal variation of population numbers among cavity-nesting birds, though this was limited to a single drainage in the northern Sierra Nevada. Also, this dataset did not show any correlation between non-cavity-nesting bird densities and precipitation. On the other hand, analysis of the dataset three years later suggested no correlation between precipitation and cavity-nesting birds (Raphael *et al.*, 1987). Overall, the area investigated in these studies is probably too limited in range and geography to apply to broader scales.

Consequences of Severe Weather for Breeding and Nesting

Severe weather has resulted in periodic, population-wide breeding failure of one or more species (Bolger *et al.*, 2004; Schaming, 2015). A population of Clark's Nutcracker in northwestern Wyoming skipped breeding altogether in years of high snowpack (Schaming, 2015). Schaming (2015) concluded that low cone crop production from whitebark pines, the primary food source for Clark's Nutcracker, in the year preceding each failure to breed was the most likely factor for non-breeding. However, Schaming (2015) could not eliminate the coinciding high snowpack in early

breeding season of those years as a possible factor for non-breeding. In contrast, Bolger *et al.* (2004) revealed breeding failure in Rufous-crowned Sparrow, California Towhee, Spotted Towhee, and Wrentit as a result of extreme low precipitation in southern California, similar to the drought conditions throughout California between 2012 and 2015 (Griffin and Anchukaitis, 2014). In both studies (Bolger *et al.*, 2004 and Schaming, 2015), the birds exhibited breeding failure through lack of attempting to build nests, rather than failure to successfully raise the young. The authors in both studies attributed the breeding failure at least partially to low food availability (“bottom-up limitation”), and Bolger *et al.* (2004) also acknowledged a potential for fewer available locations to conceal nests because of reduced vegetation.

Extreme variations in snowpack challenge the breeding habitat arrival times and nesting onset of elevational migrants, such as White-crowned Sparrow and American Pipit (Hendricks, 2003; Hahn *et al.*, 2004), as well as year-round residents (DeSante, 1990). American Pipits migrate from low elevation wintering grounds to alpine habitats, and are now arriving at the nesting grounds earlier than they did historically (Hendricks, 2003), presumably because of changing environmental cues. Consequences of earlier arrival for this ground-nesting species include limited food availability and inaccessibility of nesting habitat due to snow. Hendricks’ (2003) study in northern Wyoming revealed a delay in the onset of nesting by as much as seventeen days on average because of heavy snowpack. Consequently, because American Pipits had smaller clutch sizes with later laying dates, delayed nesting led to reduced overall numbers of offspring. Seasonal elevational movements for White-crowned Sparrow in western North America are also affected by spring storms and temperatures (Hahn *et al.*, 2004). White-crowned Sparrows arrive as much as a month before nesting in Yosemite National Park, but fluctuate daily in elevation during this time because of adverse weather. Though deep, residual, spring snow would seem to postpone food availability for ground-foraging species, Hahn *et al.* (2004) found White-crowned Sparrows similarly present regardless of total snowpack. Instead, new snowfall and consequent lack of access to food sources caused daily elevational movement.

In some areas, including the Nearctic, mean snowmelt date has occurred earlier by 2.1 days per decade since 1980 (Grabowski *et al.*, 2013). However, considerable inter-annual variation in snowmelt date still occurs (Hahn *et al.*, 2004). In contrast, mean snowmelt date at a site in the Colorado Rocky Mountains did not change over 25 years, from 1975 to 1999, causing a mismatched phenology for American Robins that are arriving for nesting up to two weeks earlier than at the beginning of the study period (Inouye *et al.*, 2000). Earlier snowmelts may allow earlier nesting and, in some areas, a chance to rear a second brood if the first fails (Hendricks, 2003; Grabowski *et al.*, 2013). Increasing frequency of severe storms (Robinson *et al.*, 2007), particularly those in the late spring or early summer (Hendricks and Norment, 1992), however, could decrease recruitment into breeding populations (Hejl *et al.*, 1988; DeSante, 1990), potentially depressing avian abundance. A snowstorm during nesting season in southern Montana caused 79% mortality in alpine American Pipit nestlings and seven percent mortality in subalpine nestlings (Hendricks and Norment, 1992). Nestling age at the time of a severe storm may define survivorship, as 100% of nestlings over ten days old survived through the first day of the storm, compared to 11% of nestlings less than ten days old. As other studies have supported (Chase *et al.*, 2005; Robinson *et al.*, 2007), Hendricks and Norment (1992) also attributed a large portion of survival and mortality to food availability, though exposure to the extreme conditions likely killed the youngest nestlings (Hendricks and Norment, 1992).

The harsh weather conditions of spring at high elevation result in reduced body condition in adult female White-tailed Ptarmigan, challenging their effectiveness in rearing young, though this does not affect survivorship (Martin and Wiebe, 2004). Adult males offer greater care for offspring in high elevation populations of a bird species than in the same species at lower elevations (Badyaev and Ghalambor, 2001). This may be an evolutionary adaptation (Bears *et al.*, 2009) that indirectly functions to maintain adult female survivorship (i.e. greater balance of brood-raising equals less strain on female health). Overall, few studies have examined differences in life-history adaptations in avian species over an elevational gradient. Photoperiod triggers biological response

for breeding in birds (Hamner, 1966), but as photoperiod does not change by elevation at the same latitude, cues such as weather and food availability may instead cause intraspecific elevational differences (Bears *et al.*, 2009). Developing a better understanding of intraspecific elevational differences in behavior and biology would allow better predictions for how shifts as a result of severe weather (short-term) or climate change (long-term) affect avian population dynamics.

Predicting Population Effects of Climate Change

In an effort to mitigate the damage future climate change may cause, many studies have developed models to predict which species will severely decline or go extinct over the next century (Sekercioglu *et al.*, 2008), how habitats can be managed to benefit the greatest number of species (Hockin *et al.*, 1992; Barrows *et al.*, 2005), and which species may soon be facing extinction or decline, but are not obviously so yet (Harris and Pimm, 2008). These predictions vary considerably, and generally each employs a range of scenarios, from conservation measures preventing most declines to worst case scenarios of widespread extinction and decline (Sekercioglu *et al.*, 2004; Sekercioglu *et al.*, 2008). There are few climate change prediction studies (Ehrlich *et al.*, 1972), however, that have studied how the paired effects of climate change reducing distribution and population size and the increased frequency of severe weather events could alter entire populations at the local level.

Justification for Study

Climate change is likely to increase the frequency of extreme weather events (Julliard *et al.*, 2004; Robinson, 2007). It is necessary to consider relationships between forced elevation shifts due to warming temperatures and the consequences of severe winters for bird populations. The Sierra Nevada, with elevations ranging from just above sea level to over 14,000 feet (4267 m), incredible habitat diversity, and periodic severe winters, offers an ideal study site for such interactions. Novel environmental conditions at different elevations will challenge the adaptations of bird species to acquire food, evade predators, and raise young (Badyaev and Ghalambor, 2001;

Martin and Wiebe, 2004; Bears *et al.*, 2009). These relationships are undoubtedly complex, including changing ecological interactions, biological requirements, trophic balances, and habitat characteristics (Sekercioglu *et al.*, 2004; Pounds *et al.*, 2005; Thomas *et al.*, 2006). They will need to be studied piecewise on both local and broader scales to synthesize a more complete understanding of the responses of bird populations to severe weather as climate changes. This study provides a local scale examination of avian response to varying snow conditions across a broad elevational gradient and within habitats that are representative of many forests of the western United States.

Chapter III

METHODOLOGY AND ANALYSIS

METHODOLOGY

Study Sites

As contract biologists for the Institute for Bird Populations (IBP), other individuals and I conducted bird surveys in three large national parks in California, United States, as part of the National Park Service's (NPS) Sierra Nevada Network (SIEN) Landbird Monitoring, from 2011 through 2014. Sequoia and Kings Canyon are adjacent to one another and jointly administered as Sequoia and Kings Canyon National Parks. Yosemite National Park ranges in elevation from approximately 650 m to nearly 4,000 m, and covers an area of 3,080 km². Sequoia and Kings Canyon span from 400 m to 4,421 m in elevation, with an area of 3,504 km². These parks host large areas of montane and subalpine conifer forests, plus oak woodlands, hardwood and riparian forests, chaparral, and alpine habitats. The extreme elevation gradients and habitat diversity of these parks provide habitat for approximately 200 birds species that breed, winter, or migrate through the Sierra Nevada, which led to the American Bird Conservancy's designation of all three parks as "Globally Important Bird Areas."

Avian Count Sampling Design

The following explanation of sampling design and protocols is based on Siegel *et al.* (2010) and three years of using the following methods as a contract biologist with IBP, and subcontractor for the NPS. For the analysis portion of this research, I have accessed the data through the NPS SIEN Inventory and Monitoring Program (I&M), Bird Monitoring database. Funding for the

collection of these data was also provided by the NPS I&M Program. I have obtained permission to use the data for this study through completion of the SIEN Data Sharing and Confidentiality Agreement (Appendix A).

The bird surveys used a spatially-intensive, variable circular plot method (hereafter “point counts”) design in transects, with each transect beginning on a trail or road. A point count is a method of counting birds in which an observer records birds that are seen and heard from a specified point for a set amount of time. Point counts are typically conducted in transects or grids to determine the abundance, species richness, and other population measures of the study taxa. The annual survey effort in each park was divided, such that half of transects were revisited annually, and the other half were one of a set of four alternating panels that were visited every four years. My analyses incorporate data collected at the annual transects, but none of the alternating transects.

Annual and alternating transects were stratified by elevation (low, mid, and high) in each park to ensure balanced coverage of the vegetation and habitat types. Due to differences in relative amounts of habitat between the parks, and a considerable foothill habitat zone in Sequoia National Park, the elevation boundaries for the strata differed between parks. Habitat type has been used as a general proxy for elevation in this study (*see Siegel et al., 2010, p. 20 for an explanation of strata elevation boundaries*).

For the four years of bird monitoring, 2011 to 2014, surveys began on or after May 18 and were completed by July 23. Each year we surveyed a maximum of twelve annual transects and fifteen alternating transects each at Yosemite and Sequoia and Kings Canyon. In 2012 and 2013, we surveyed all twenty-seven scheduled transects in each park. In 2011, we completed fifty-one of the fifty-four total transects, due to high snow levels and inaccessibility of several transects (annual transect 8011 was not surveyed). We surveyed all transects in both parks except annual transect 8008 in Kings Canyon in 2014 (*see Figure 3.2, p. 20*). These annual and alternating transect quantities differ from Siegel *et al.* (2010), as it was determined in May 2012 that the original design was oversampling the parks. The minor restructuring before the 2012 season also meant that annual

transects 6025 and 6027 were first surveyed in 2012. Consequently, in each year field crews aimed to survey two low-elevation transects (one of which was surveyed annually), sixteen mid-elevation transects (seven annually), and nine high-elevation transects (four annually) in Sequoia and Kings Canyon. In Yosemite, field crews aimed to survey nine low-elevation transects (four annually), nine mid-elevation transects (four annually), and nine high-elevation transects (four annually).

Conducting Avian Surveys

In general, two observers in each park (2 in Yosemite, 2 in Sequoia and Kings Canyon) conducted surveys during a single season. In all four years of data collection, however, limitations on season-length, weather conditions, and/or travel time between transects necessitated additional observer assistance from other trained, project-related staff. All observers participated in an eighteen-day mandatory training, during which they refined their bird identification skills by sight and sound, developed the ability to estimate distance to birds with a laser rangefinder, and became proficient with all of the field methods for the study. At the end of the training program, all observers were required to pass a bird identification certification exam in order to begin conducting surveys. The study had a relatively high level of continuity of observers from 2011 to 2014, with the same IBP project leads and NPS SIEN lead throughout, and one, two, and two (2012, 2013, and 2014 respectively) field crew members returning each year from the previous season.

Point counts began within ten minutes of local sunrise and continued until three-and-one-half hours after sunrise. A pair of observers completed a single transect of at least fourteen point counts each morning. For each transect, the pair of observers began at the same transect origin point on a trail, or occasionally a road, and walked in opposite directions from one another, perpendicular to the trail, for 125 m, so that the two observers were 250 m from one another for their respective first point counts. For each successive point count, each observer would continue in the same direction away from the transect origin for 250 m to each new point.

Observers counted at each point for seven minutes (contrary to Siegel *et al.*, 2010), with observations of individual birds divided into three occupancy periods (0-3 minutes, 3-5 minutes, and 5-7 minutes). Before each point count, observers recorded data describing the site, which included: point coordinates (UTM), point count start time, noise interference level, and weather conditions. During each point count, observers recorded the following (as applicable) for each bird sighting: species, distance from observer, “seen” or “heard only,” group size, previous detection of bird, and flyovers (to acknowledge that an individual was only observed flying above the count area, and was not actively using the count area). Only adult birds were included in point counts, though offspring, nests, and other evidence of breeding were recorded separately.

For the data recorded in the field and for this thesis, coordinates recorded describe the location of each point count, and not the exact coordinates of each bird. Consequently, some variation almost certainly exists in the pairings of individual birds with habitat types, as habitat designation was based on a fifty meter radius circle centered on each count point.

Determination of Habitat Type

Upon completion of a transect, each observer returned along the same path of travel to the transect origin. During the return trip, an observer recorded habitat-related data for each point where a count occurred based on the California Wildlife Habitat Relationships (CWHR) classification system (Mayer and Laudenslayer, Jr., 1988). (*See “SOP 5: Conducting Point Counts” on p. SOP 5.1 in Siegel et al., 2010 for complete description of the previously-mentioned methods*).

For this thesis, the CWHR types recorded for each point count were cross-referenced using geographic coordinates of the annual transect survey points and the National Vegetation Classification Standard (NVCS) GIS data layer of each national park for consistency. The NVCS is a product of the Federal Geographic Data Committee that provides a nationwide standard for describing detailed vegetation types (Federal Geographic Data Committee, 2008). Because the

NVCS vegetation types are classified to a finer level of detail than the CWHR habitat types, I grouped the applicable NVCS types in each park into categories similar to those of the CWHR habitat types. In some cases, the CWHR designations varied slightly among years at single count points. Except in cases where a natural phenomenon (e.g. fire) drastically changed the vegetation at a point between years, I assigned a single, primary CWHR habitat type to each count point across all four years, based on the NVCS and CHWR data, and personal familiarity with each transect. This allowed me to compare bird species among years and within a single habitat type.

Snow Depth Data

I collected information about snow depths in the national parks during the study period from the California Department of Water Resources' Data Exchange Center online (California Data Exchange Center, 2015). These snow sensors are part of the California Cooperative Snow Surveys Program, which has been coordinated by the California Department of Water Resources since the program's origin in 1929. The sensors record snow data remotely, and are operated by one of many participating agencies, including the NPS, United States Forest Service, US Army Corps of Engineers, Bureau of Reclamation, and University of California, Merced. I obtained snow depth data from sixteen active snow sensors: seven in Sequoia and King Canyon, five in Yosemite, two in Sierra National Forest, near Yosemite's southern boundary, and two in Sequoia National Forest, near Kings Canyon's western boundary and Sequoia National Park's southern boundary (Table 3.1, Figure 3.1, and Figure 3.2).

The snow sensors did not specifically collect data for use in this study, so the locations of the point counts do not correspond exactly to the locations of the snow sensors. Rather, the snow sensors range in distance to avian survey transect origins from 0.85 km to 17.74 km ($\bar{x} = 6.9 \pm 4.4$ km), though only the Green Mountain snow sensor exceeds 12.5 km. I selected the Green Mountain snow sensor for survey 6019 (distance = 17.74 km) for multiple reasons, even though it was not the nearest sensor. Most importantly, neither of the two closer sensors that are actively collecting

snowfall data, Merced Lake and Cora Lakes, offered sufficiently accurate snow depth data. The Cora Lakes snow depth was only recorded once monthly, while the Merced Lake sensor failed to collect April snow depth data in three of the four years, an apparent sensor malfunction. Additionally, the elevation of the Green Mountain sensor is 200 m more similar to the survey 6019 elevation than the Merced Lake sensor.

Table 3.1. Sierra Nevada active snow sensor stations. Sixteen snow sensor stations measured snow depths in this study. Management designation, elevation in meters, and the avian transect with which each sensor was paired are also included. Primary habitat, the type of which the highest number of count points were composed, is included for each transect. CWHR codes for habitats are: SMC (Sierran mixed conifer), RFR (red fir), LPN (lodgepole pine), and SCN (subalpine conifer).

Snow Sensor Name	Location	Sensor Elevation (m)	Corresponding Avian Transect(s)	Primary Habitat of Transect
Lower Kibbie Ridge (KIB)	Yosemite NP	2042	6027	SMC
Ostrander Lake (STR)	Yosemite NP	2499	6029, 6034	RFR, RFR
Slide Canyon (SLI)	Yosemite NP	2804	6032	SCN
Tenaya Lake (TNY)	Yosemite NP	2484	6014, 6026	LPN, SMC
Tuolumne Meadows (TUM)	Yosemite NP	2621	6016, 6023	LPN, LPN
Green Mountain (GRM)	Sierra NF	2408	6019	LPN
Big Sandy (BSN)	Sierra NF	1755	6025	SMC
Big Meadows (BIM)	Sequoia NF	2316	8001	SMC
Wet Meadows (WTM)	Sequoia NF	2728	8005	RFR
Charlotte Lake (CRL)	Kings Canyon NP	3170	8004	SCN
State Lakes (STL)	Kings Canyon NP	3139	8008, 8010	RFR, SMC
Chagoopa Plateau (CHP)	Sequoia NP	3139	8003	SCN
Giant Forest – USACE (GNF)	Sequoia NP	2027	8002, 8013, 8034	SMC, SMC, SMC
Crabtree Meadow (CBT)	Sequoia NP	3261	8007	SCN
Panther Meadow (PTM)	Sequoia NP	2621	8006	RFR
Upper Tyndall Creek (UTY)	Sequoia NP	3475	8011	LPN

Snow sensors differ in elevation from 4 m to 598 m ($\bar{x} = 259 \pm 180$ m) from their associated avian transect origin. Such differences in elevation are less considerable than they appear, however, because the survey points that emanate in two directions from each transect origin frequently differ greatly in elevation from the origin, and are often closer in elevation to the snow sensor than the transect origin. More specifically, for example, even though transect 6034 begins at only 2,047 m

elevation, both segments of the transect increase in elevation as distance from the origin increases, and the final count point of the southwestern segment differs in elevation from the Ostrander Lake snow sensor by a mere 45 m. This pattern frequently repeats, and in turn the snow sensor paired to a particular transect provides a sufficient general idea of snow conditions at that transect. Lastly, no snow sensor existed near enough in distance or elevation to transect 8010 to justify including this survey in the snow depth analysis.

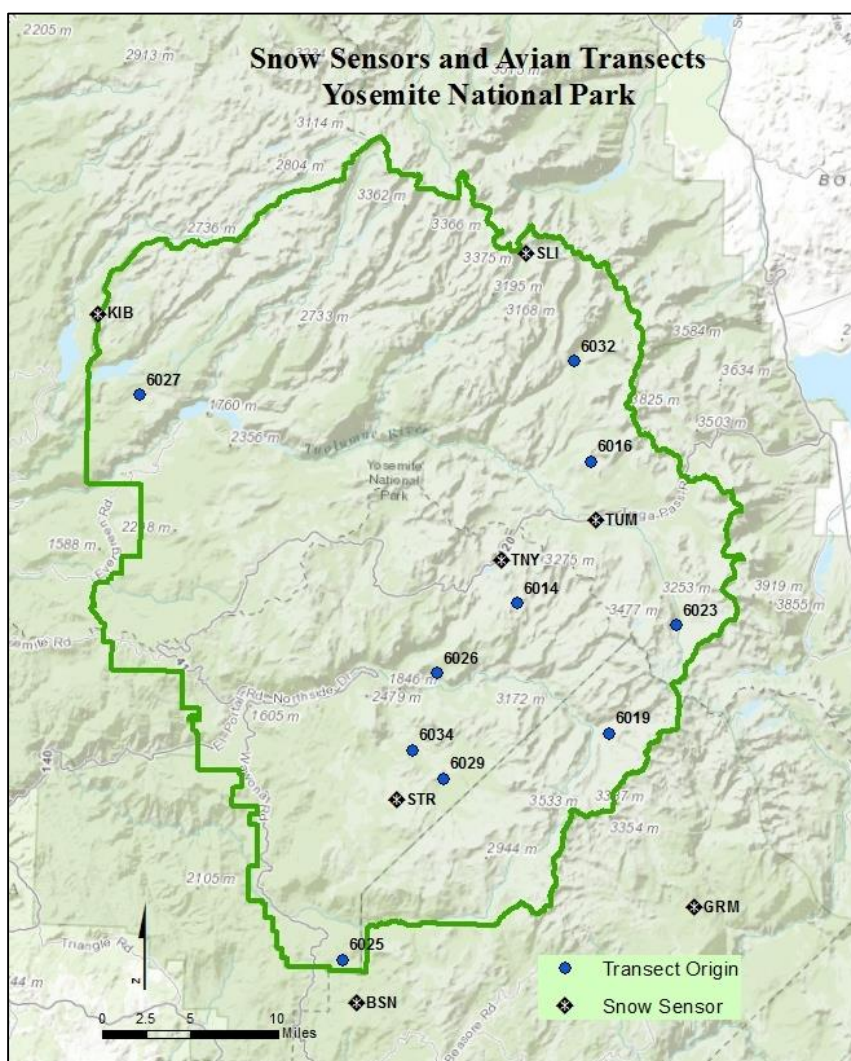


Figure 3.1. Map of Yosemite National Park. This map shows the locations of the annual transect origin points and their corresponding snow sensor locations for the northern portion of the study area. See Table 3.1 for sensor names and abbreviations.

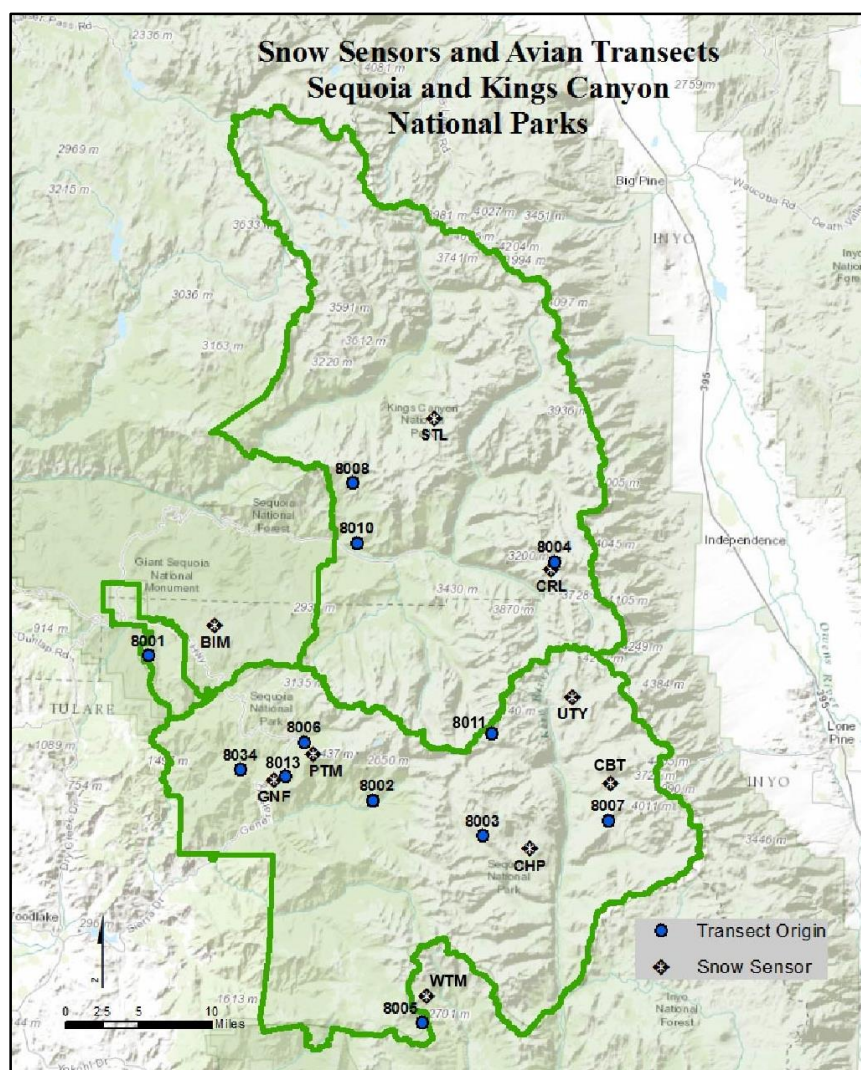


Figure 3.2. Map of Sequoia and Kings Canyon National Park. This map shows the locations of the annual transect origin points and their corresponding snow sensor locations for the southern portion of the study area. See Table 3.1 for sensor names and abbreviations.

As snow conditions vary even across short distances depending on elevation, aspect, slope, vegetation cover, and other factors, I have made every effort to pair the appropriate snow sensor to each avian transect, based on these factors. In several cases, the way in which specific snow sensors recorded data about the snow differed, and I took steps to make the data for analysis consistent. For example, at four sites in some of the years, the sensors recorded snow depth hourly rather than daily, so the April average is based on twenty-four daily sensor readings rather than just one. Additionally, some sensors only provided snow water content data during the study period. Snow

depth can be approximated using snow water content values and monthly conversion factors (California Data Exchange Center, 2015). The conversion factors for April 1 and May 1 are 2.25 and 2.0, respectively. In order to convert the daily snow water content values, I multiplied the values within spans of five days by their corresponding conversion factor (Table 3.2). This gave daily snow depths for the month of April each year, which I then averaged within each year to match the daily snow depth data records of other snow sensors.

Table 3.2. Conversions for snow water content and snow depth. The daily snow water content was converted to snow depth using the corresponding conversion factors for each period of five days in April. Note that the conversion factors assume a linear relationship throughout April.

Date Span	Conversion Factor
April 1 – April 5	2.25
April 6 – April 10	2.20
April 11 – April 15	2.15
April 16 – April 20	2.10
April 21 – April 25	2.05
April 26 – April 30	2.00

The Guyot Flat snow sensor, which would have paired well with avian transect 8007, also only recorded a single, monthly measurement, and I therefore elected to pair transect 8007 with the Crabtree Meadow sensor. This sensor recorded daily April snow depth during 2011 and 2013, but not 2012 or 2014. It did, however, record once-monthly snow depth for April 2012 and 2014. To fill in these blanks, I calculated the ratios of 2011 and 2013 snow depth between Crabtree Meadow and nearby Chagoopa Plateau for March, April, and May. Using three months allowed me to be more certain of consistently-occurring ratios between the sites. The snow sensor at Chagoopa Plateau is approximately 11.5 kilometers southwest of Crabtree Meadow, but both are in alpine meadows, and on generally west-facing slopes with a difference in elevation of 122 m. I compared these ratios to the ratios of once-monthly recordings between the two sites in order to approximate a daily depths ratio for 2012 and 2014. Using the daily depths ratio and the known daily measurement means from Chagoopa Plateau, I approximated mean April snow depth for 2012 and

2014 at Crabtree Meadow. I confirmed the validity of my Crabtree Meadow estimates by using the same process to compare ratios using the Upper Tyndall Creek sensor.

The sensor at Panther Meadow (PTM) also only recorded snow depths as April 1 measurements, so I used the same ratios method based on Giant Forest sensors (GNF and GFR) to estimate mean April depths at Panther Meadow. Finally, among the Big Sandy, Ostrander Lake, and Slide Canyon sensors, there were four mean April snow depths I could not directly calculate from the existing data. As with the earlier cases, I defined ratios using monthly, hourly and daily snow data from the Fresno Dome, Peregoy Meadows, and Horse Meadow sensors, respectively, to calculate best estimates for the missing values.

Determination of Bird Species for Analysis

For analysis in this study, I selected the most commonly-detected species at the annual transects across all years, and within the four vegetation types of focus. We detected fourteen species at least 300 times during the four years of study (Table 3.3). I chose to combine and analyze Hammond's and Dusky Flycatcher as a single species, due to their close taxonomic relationship and potential for identification confusion by sight and sound. This may have some confounding effects on the ecological significance of the results for these species as Dusky Flycatcher typically inhabits slightly higher elevations and habitats than Hammond's Flycatcher, but there is considerable overlap between the two (Siegel *et al.*, 2011).

Table 3.3. Common bird species selected for study. The number of transects at which each species occurred in each year and the combined total are listed. The combined total transects column explains the number of unique annual transects (out of 22) at which each species was detected over the course of four years. Also included are the raw total detections of each species across years within the four habitat types of the study. Maximum possible values for each year are 19 (2011), 22 (2012), 22 (2013), 21 (2014), and 22 (combined). Species are listed in descending order by the total transects where detected.

Species	Band Code	2011	2012	2013	2014	Total Transects	Total Detections
Hammond's / Dusky Flycatcher (<i>Empidonax hammondi</i> / <i>E. oberholseri</i>)	HAFL DUFL	17	21	18	16	22	463
Mountain Chickadee (<i>Poecile gambeli</i>)	MOCH	19	22	22	21	22	928
American Robin (<i>Turdus migratorius</i>)	AMRO	11	18	22	18	22	316
Yellow-rumped Warbler (<i>Setophaga coronata</i>)	YRWA	18	22	22	19	22	878
Dark-eyed Junco (<i>Junco hyemalis</i>)	DEJU	19	22	21	21	22	931
Cassin's Finch (<i>Haemorhous cassinii</i>)	CAFI	16	14	20	16	22	343
Steller's Jay (<i>Cyanocitta stelleri</i>)	STJA	11	14	16	18	19	690
Red-breasted Nuthatch (<i>Sitta canadensis</i>)	RBNU	10	13	16	16	17	733
Golden-crowned Kinglet (<i>Regulus satrapa</i>)	GCKI	9	12	13	14	17	432
Western Tanager (<i>Piranga ludoviciana</i>)	WETA	8	14	15	13	17	514
Fox Sparrow (<i>Passerella iliaca</i>)	FOSP	10	11	15	9	16	472
Clark's Nutcracker (<i>Nucifraga columbiana</i>)	CLNU	10	10	12	11	15	306
Western Wood-Pewee (<i>Contopus sordidulus</i>)	WEWP	11	13	13	14	15	319
Nashville Warbler (<i>Oreothlypis ruficapilla</i>)	NAWA	7	12	11	13	15	312

ANALYSIS

We entered the avian point transect data into a Microsoft Access database during and after each field season. Next, I queried the data in Microsoft Access to obtain records of the fourteen common species found within the focus habitat types on annual transect points, and transferred the data as tab-delimited text files into the program, DISTANCE v6.0 Release 2 (Thomas *et al.*, 2010), for calculation of abundances and densities.

DISTANCE Analysis – Calculating Avian Abundance

I analyzed avian point count data in the program, DISTANCE v6.0 Release 2, to predict the absolute density and abundance of each species within survey transects. DISTANCE is a software program that provides researchers the ability to “enter, import, and view data, design

surveys, and run analyses” (Thomas *et al.*, 2010) in order to estimate size or density of biological populations. Based on distance sampling principles (Buckland *et al.*, 2001; Thomas *et al.*, 2002, 544), observers need not record on a survey every individual of a species that is present in an area. As long as users maintain several basic assumptions (Buckland *et al.*, 2008, S94), DISTANCE can calculate the detection probability of a species, and therefore estimate abundance, based on available observations and the horizontal distances to observed individuals. This application of DISTANCE is called conventional distance sampling (“CDS”). Point transect sampling, as has been conducted for this data collection, is one of the most common methods for monitoring avian populations, and DISTANCE is the most widely-used software for distance sampling analysis (Marques *et al.*, 2011). The developers of DISTANCE have added the ability for the program to simultaneously analyze several variables (multiple-covariate distance sampling, “MCDS”), in addition to horizontal distance from observer, that affect the ability of a person to detect an animal. For example, not only does a bird become more difficult to see or hear with greater distance, but surrounding vegetation, weather conditions, and ability of observer might also influence the likelihood of a person to detect a specific bird (Bailey *et al.*, 2007).

Relatively small numbers of observations limited my ability to treat observer or habitat as covariates in the MCDS engine in DISTANCE. Therefore, I used the CDS engine for abundance and density analyses, and post-stratified, using count point vegetation. I also pooled all observers, treating the detection rates of each bird species by different observers as equal. Because all observers went through the same pre-survey training and examination (*see Conducting Avian Surveys*, p. 15), I have assumed all observers equal in their abilities to detect birds and estimate distances for these analyses. Although there probably are differences in observer abilities, the pooling robustness property of distance sampling allows data pooling across strata without generating bias, as long as there is not high detection variability among strata (Buckland *et al.*, 2004, 389-392). Pooling data across observers also reduces the effects of rounding observation distances to the nearest five- or ten- meter mark. However, during training the project manager

encouraged observers to record distances to the nearest meter, and thus, upon examination of data histograms in DISTANCE, the data did not show evidence of rounding.

Model selection. The model selection within DISTANCE for determination of detection function is based on a parametric key function paired with a series adjustment term (Thomas *et al.*, 2010). Within the CDS engine, DISTANCE offers four parametric key functions: half-normal, hazard-rate, negative exponential, and uniform. Three series adjustment terms: simple polynomial, Hermite polynomial, or cosine, can be paired with a key function, though the key function can also operate alone. The series adjustment terms serve to improve the fit of the model to the observation data. After considerable exploration of the data within several model definition combinations, I identified a half-normal parametric key function with a simple polynomial series adjustment term as the best model for these data.

As recommended in Buckland *et al.* (2001, 151), where appropriate, the data for each species were truncated to remove the largest 10% of distances in each year (also called “right truncation”), in order to eliminate a tail in the distribution of the data that could skew the model. In a few cases, especially for species that observers detected fewer times, 10% right truncation resulted in poor model fit, and five percent right truncation was used instead.

Initially, I analyzed each species to calculate a study-wide (“global”) density across all four years. I post-stratified this analysis by year to estimate density, detection function, and encounter rate for each species in each year. Also in this analysis, I set an individual right truncation distance (meters) for each species, rather than a percentage truncation, visually truncating the data at the distance where detection probability reached approximately 0.10.

In a second set of analyses, I estimated density for each species, each year, with no further stratification. In DISTANCE, this meant a global resolution for density, encounter rate, and detection function with each year as a separate analysis. As stated earlier, these analyses truncated the largest 10% of observations to better fit the model to the data.

Finally, I ran four sets of analyses (one for each year) that estimated each species' density, encounter rate, and detection function, post-stratified by habitat type (Sierran mixed conifer, red fir, lodgepole pine, and subalpine conifer).

DISTANCE 6.2 – Justification of Use

The raw data of this research represent a measure of per-site avian abundance, with the number of each species present near each count point available in the data record of a given site. Field observers were sufficiently trained in identification of all species that I have included in this study, and put forth their best efforts to detect as many species and individuals as they could during each survey. Therefore, I could have proceeded with analysis in this study using only the raw data. However, the raw data do not account for variation in detectability from species to species, such that data for species that are more easily detected, but not necessarily more abundant, would be analyzed with higher assumed abundance than data for species that are difficult to detect.

DISTANCE calculates the detectability and encounter rate for each species using field observations, and fits the data to models to more accurately estimate the density of each species in an area. For example, an observer might detect only one Hammond's Flycatcher during a point count, when in fact three are present. Hammond's Flycatchers frequently perch on low branches in dense forests, sometimes without moving or singing for long periods of time (Davis, 1954; Mannan, 1984), making them difficult to detect. This would likely result in a lower detection rate than that of a Steller's Jay, for example, which often moves to examine an intruder (human observer) to its territory (*personal observations*), and vocalizes loudly and often. A bird's rate of singing can greatly affect the probability of its detection. Detection function may also vary based on size of birds and relative abundance in an area. DISTANCE accounts for these differences in detection function among species to predict the absolute density and abundance of each. Although raw field data provide an estimate of the number of individuals present for each species, applying

DISTANCE models to the raw data balances the disparities in abundance due to detectability of species, and delivers more thorough estimates of density.

Species Richness Analysis

I calculated species richness across entire transects, the cumulative species composition of all points in a transect, for each year. The richness values include all species detected on counts, except raptors, waterfowl, aerial insectivores, and species occurring on less than five percent of total transects. A complete list of these species is included in Appendix B. In defining species richness as the number of species detected at a site, this analysis made the assumption that all species present at a given site were detected. Extrapolation of the existing data to determine absolute richness values would have involved learning and carrying out Canonical Correlation Analysis within the R Project for Statistical Computing, or a similar program that calculates absolute richness, and is outside the scope of this thesis. I included the most commonly-occurring habitat type for each transect alongside the annual richness values and four-year mean (Table 4.2), based on the habitat that was represented by the highest number of count points within each transect. Additionally, I determined the species richness for each habitat type during each year. In many cases, this separated single transects into different habitat classes, but it better allowed for the determination of geographical differences in species richness or elevational movements of species from year to year.

Statistical Analyses for Abundance and Species Richness

Several of the focus species have a tendency to occur primarily at relatively lower or higher elevations, respectively, and therefore at certain transects these species did not occur at all in any year. The resulting zero values in the abundances for these species at some transects skewed the data such that they were not normally distributed. Consequently, for several of the statistical tests, I used the recommended non-parametric tests. To examine the abundance patterns of each species among years and across habitat types, I used Kruskal-Wallis H tests. For the species that showed

significant difference in abundance among years, I conducted a post-hoc test to identify year pairings that differed statistically from one another. The Steel, Dwass, Critchlow-Fligner (SDCF) multiple comparison analysis is a post-hoc test that compares the medians of all pairs of independent variables within a group, and is recommended after a Kruskal-Wallis test to identify the pair(s) most relevant to the results of the statistical analysis. I followed this with Mann-Whitney U tests for all species to examine the relationships of abundance medians between 2011 and 2012, and between 2013 and 2014, the years in which abundances generally had the greatest changes across species.

I also used one-way analysis of variance (ANOVA) to compare the species richness means of transects among years. A slightly left-skewed distribution of richness led me to test for differences in species richness medians using the Kruskal-Wallis test as well. Next, I examined the species in terms of their distributions across habitat types, using the annual density measures for each species in each habitat type to calculate the Smith and Wilson evenness index (E_{var} ; Smith and Wilson, 1996). The Smith and Wilson evenness index is a statistic used to explain the balance of avian abundance values across vegetation types in a sample, independent of the richness of the sample (*see Smith and Wilson, 1996 for the E_{var} equation*). E_{var} values are restricted between 0 and 1, and higher values indicate less variation in the abundance values among habitat types.

In order to study yearly changes in species richness between elevationally-adjacent habitat types, I used two-sample *t*-tests, comparing each pair of adjacent habitats within each year. For examination of the abundance of each species, I calculated abundance change from one year to the next in each habitat type. This helped to identify potential occurrences of upslope or downslope shift in habitat in specific years. I did not conduct statistical tests for individual species abundance within each habitat type. Although I knew the variance and standard deviation for each annual abundance mean within each habitat, after calculating the difference between two years, I could no longer apply those variances to test for statistical differences of year to year change between habitats.

Lastly, I measured correlation between snow depth and avian abundance. Based on the avian transect-snow sensor pairings in Table 3.1, I created datasets containing abundance and snow depth pairs for each species, and examined the degree to which the two variables are related.

CHAPTER IV

RESULTS

Differences among Years

Avian abundance. I calculated abundances with a “per count point” unit, basing each count point’s area on a 200 m radius circle, centered on the point. Therefore, the abundance values are the per-hectare (ha) density of a species multiplied by the 12.57 ha area that composes a count point. Additionally, after evaluation of the DISTANCE-calculated abundances and associated variances, I removed avian transect 8011 from the Kruskal-Wallis and Mann-Whitney U tests due to its consistently-high variances and outlying abundances. With a transect origin at approximately 12,000 ft., transect 8011 consisted of only three count points within the habitat types of focus in this study, thereby producing very few birds from point counts.

Only four of the fourteen species showed statistically significant differences ($\alpha = 0.05$ for all tests in this study) in density and abundance among the four years of the study: Dark-eyed Junco (Kruskal-Wallis, $H_{3,80} = 9.951$, $p = 0.019$), Mountain Chickadee ($H_{3,80} = 15.593$, $p = 0.00137$), Red-breasted Nuthatch ($H_{3,80} = 11.712$, $p = 0.00844$), and American Robin ($H_{3,80} = 8.288$, $p = 0.04$). Upon further investigation within years for Dark-eyed Junco, Mountain Chickadee, and Red-breasted Nuthatch, the Steel, Dwass, Critchlow-Fligner analysis identified several pairs of years for each species in which the median densities differed statistically from one another (Table 4.1). Pairwise comparisons of density and abundance between years differed statistically for Dark-eyed Junco between 2013 and 2014 (SDCF, $p = 0.0483$) and nearly so between 2012 and 2014 ($p = 0.0771$; Table 4.1 and Figure 4.1). Mountain Chickadee had statistically different median densities between 2011 and 2014, 2012 and 2014, and 2013 and 2014 (SDCF, $p = 0.0199$; $p = 0.0029$; $p =$

0.0338, respectively; Table 4.1 and Figure 4.1), whereas median densities for Red-breasted Nuthatch differed between 2011 and 2014, and 2012 and 2014 (SDCF, $p = 0.0454$; $p = 0.0208$, respectively; Table 4.1 and Figure 4.1). The relatively high densities in 2014 for each of these species resulted in the frequent statistical difference between other years and 2014, whereas none of 2011, 2012, or 2013 differed statistically from one another for any species. Also notable, the median density for Dark-eyed Junco in 2012 was greater than in 2013 (Table 4.1), but the mean abundance (and by extension, mean density) for Dark-eyed Junco in 2013 exceeded that of 2012 (Figure 4.1). This occurred because of Dark-eyed Junco's comparatively lower variance for mean abundance in 2012 (4.25 versus 11.2).

Table 4.1. Post-hoc differentiation of median densities by year. Paired years with statistically different median densities in the post-hoc test, Steel, Dwass, Critchlow-Fligner multiple comparison analysis for Dark-eyed Junco, Mountain Chickadee, and Red-breasted Nuthatch.

Species	Paired Years	Median Densities	Standard Error ($\pm$)	p
Dark-eyed Junco	2012, 2014	1.59, 2.98	0.26, 0.44	0.0771
	2013, 2014	1.48, 2.98	0.24, 0.44	0.0483
Mountain Chickadee	2011, 2014	0.96, 2.13	0.12, 0.33	0.0199
	2012, 2014	0.78, 2.13	0.13, 0.33	0.0029
	2013, 2014	1.04, 2.13	0.15, 0.33	0.0338
Red-breasted Nuthatch	2011, 2014	0.33, 1.03	0.12, 0.22	0.0454
	2012, 2014	0.23, 1.03	0.06, 0.22	0.0208

In contrast, although the Kruskal-Wallis test for American Robin suggested a significant difference in median density among years, the SDCF analysis did not offer any pair(s) of years that statistically differed in density from one another. Such a result is not uncommon in this sort of two-stage comparison, however, and has been previously addressed (Gabriel, 1969). Essentially, the Kruskal-Wallis test identified that the data are statistically different, but the post-hoc test did not have enough information to clearly identify the responsible year pairing(s). I may be able to attribute the Kruskal-Wallis difference in medians to the pairing of 2011 and 2013 for American

Robin, with median densities of 0.43 ± 0.59 and 0.85 ± 0.77 individuals per point, respectively, but these years still did not statistically differ in median density ($p = 0.105$).

For the above abundance analyses, I used only the DISTANCE analyses based on global stratification of each species in each year, those to which I applied a truncation to remove the largest 10% of sightings (*see p. 25*). I did not perform further statistical analyses on the first set of DISTANCE data, those to which I applied specific meter-based truncations and post-stratification

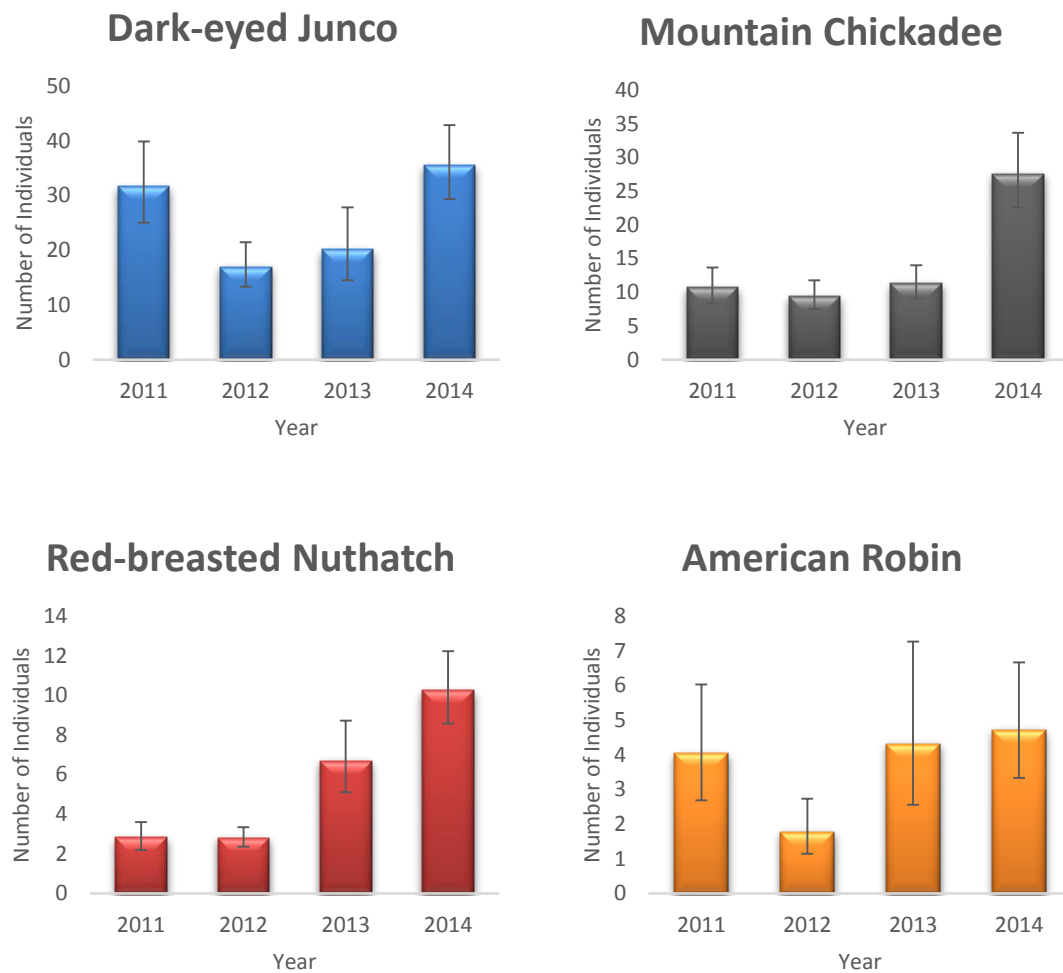


Figure 4.1. Annual mean abundances of Dark-eyed Junco, Mountain Chickadee, Red-breasted Nuthatch, and American Robin. These four species showed statistically significant differences in abundance among years for Kruskal-Wallis tests. Abundance values were calculated based on each 12.57 ha count point area. Associated vertical bars display the 95% confidence intervals for each mean abundance.

by year, as I determined the results with the 10% truncations provided all around better fits to their respective models.

Generally, all of the fourteen most common species except one decreased in density and abundance from 2011 to 2012, though none were statistically significant decreases. All of these species displayed an approximately similar pattern in density and abundance values over the course of the study, creating a “lazy U” shape from 2011 to 2014 (Figure 4.2). Abundances decreased from 2011 to 2012, remained similar or increased slightly in 2013, and then increased sharply in 2014. This pattern is most obvious in the five most abundant species, Golden-crowned Kinglet, Dark-eyed Junco, Mountain Chickadee, Yellow-rumped Warbler, and Hammond’s/Dusky Flycatcher, and becomes less noticeable with decreasing overall abundance. Only Clark’s Nutcracker increased

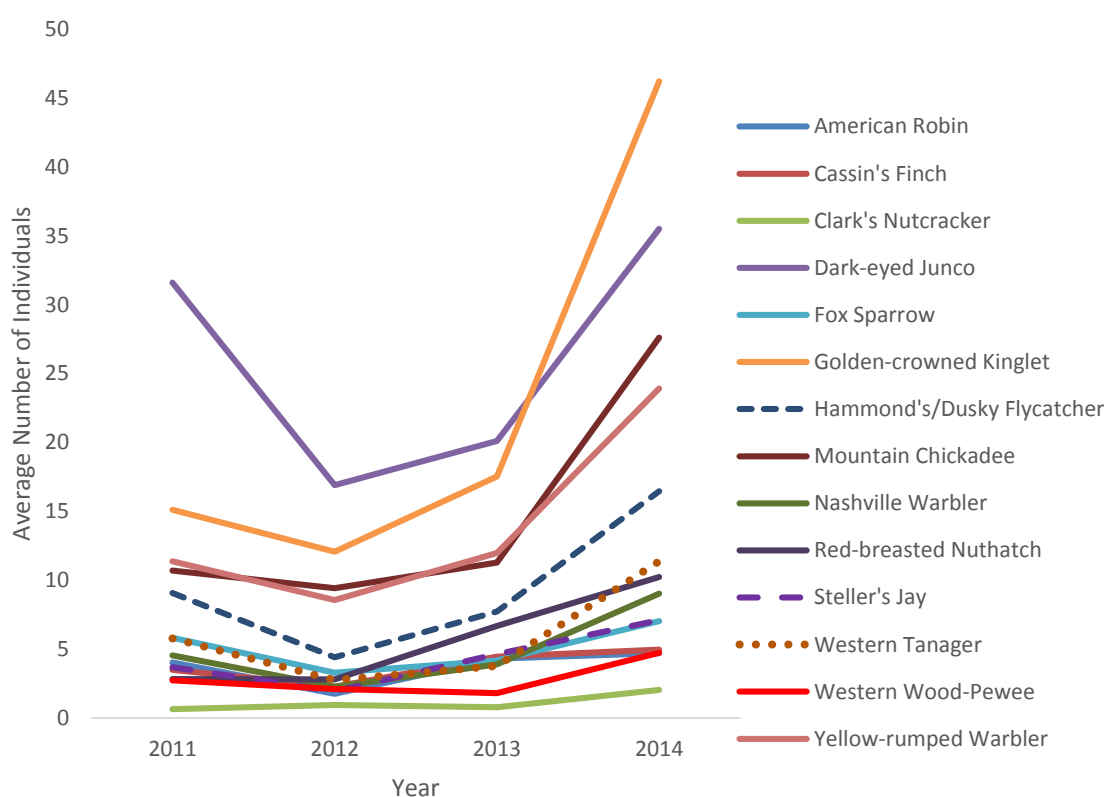


Figure 4.2. Annual mean abundances of all common species. Abundance values were calculated based on each 12.57 ha count point area in Yosemite, Sequoia, and Kings Canyon National Parks.

in density from 2011 to 2012, though not significantly, from 0.65 birds per ha to 0.95 birds per ha (Mann-Whitney, $U = 423.00$, $p = 0.8262$; Appendix C).

Due to the visually obvious increase in abundance for many species from 2013 to 2014, I tested the two years for difference of medians with Mann-Whitney U tests. Surprisingly, only two species showed significantly different median abundances between 2013 and 2014: Dark-eyed Junco ($m_{2013} = 1.45$, $m_{2014} = 2.95$; $U = 519.00$, $p = 0.0098$) and Mountain Chickadee ($m_{2013} = 1.02$, $m_{2014} = 1.93$; $U = 524.0$, $p = 0.0067$). Yellow-rumped Warbler also had a notable increase in mean abundance from 2013 to 2014 ($m_{2013} = 1.01$, $m_{2014} = 1.59$; $U = 494.0$, $p = 0.0536$). Golden-crowned Kinglet displayed tremendous increase in median abundance from 2013 to 2014 (from 0.64 to 2.8 individuals per count point), but its statistical significance may have been affected by the use of the relatively-weaker Mann-Whitney U test. The Mann-Whitney U test also considers the equality of the shape of the distributions being tested. Upon closer inspection, mean abundances of three transects in 2014 exceeded ten individual Golden-crowned Kinglets per count point, whereas in 2013, one transect's mean abundance of 8.4 individuals per count point was nearly double that of any other transect that year. The tail of the 2014 distribution for Golden-crowned Kinglet abundance created by the three outlying transects gave the 2014 distribution a different shape than 2013, perhaps impeding the ability of the test to recognize a difference in median abundances between the years.

Species richness. Analysis of variance of the species richness at survey transects did not reveal a significant difference among the four years (one-way ANOVA, $F_{3,80} = 2.01048$, $p = 0.1191$, Table 4.2). Because histograms showed the distributions of species richness in each year to be skewed slightly left rather than normally distributed, I also ran a Kruskal-Wallis test to compare median species richness among years. However, the Kruskal-Wallis test did not show statistically significant difference in median richness among years either ($H_{3,83} = 6.231$, $p = 0.101$).

The measure of species richness on per transect basis gave those transects composed of more count points a higher likelihood of having a greater species richness. As it was more time-consuming to establish a count point in its first year than to re-find that point the following year, observers were able to more efficiently survey in subsequent years and establish additional count points. Therefore, in general, each transect had more count points from which to analyze species richness from one year to the next. I attempted to mitigate this inequality by calculating species

Table 4.2. Annual richness values for each avian transect. Mean four-year richness and standard error are also included. Primary habitat, the type of which the highest number of count points were composed, is included for each transect. CWHR codes for habitats are: SMC (Sierran mixed conifer), RFR (red fir), LPN (lodgepole pine), and SCN (subalpine conifer). Transects with a richness value of “N/A” were not surveyed in that year. Transects are listed in descending order by mean richness.

Transect	2011	2012	2013	2014	Mean Richness $\pm$ S.E.	Primary Habitat
6027	N/A	33	37	35	35 $\pm$ 1.15	SMC
8001	32	31	38	33	33.5 $\pm$ 1.55	SMC
8013	31	32	34	36	33.3 $\pm$ 1.11	SMC
6025	N/A	27	37	35	33 $\pm$ 3.06	SMC
8034	31	35	31	29	31.5 $\pm$ 1.26	SMC
6029	22	30	35	33	30 $\pm$ 2.86	RFR
8010	25	32	32	32	29.7 $\pm$ 2.33	SMC
8005	20	33	30	33	29 $\pm$ 3.08	RFR
6026	24	27	32	33	29 $\pm$ 2.12	SMC
6034	24	25	32	27	27 $\pm$ 1.78	RFR
8002	26	26	24	28	26 $\pm$ 0.82	SMC
8008	24	24	23	N/A	25.8 $\pm$ 2.1	RFR
8006	19	22	22	28	22.8 $\pm$ 1.89	RFR
8003	15	22	21	25	20.8 $\pm$ 2.1	SCN
6014	13	13	25	23	18.5 $\pm$ 3.2	LPN
6032	14	17	21	20	18 $\pm$ 1.58	SCN
6023	16	13	16	23	17 $\pm$ 2.12	LPN
6016	11	16	14	21	15.5 $\pm$ 2.1	LPN
6019	15	12	13	19	14.8 $\pm$ 1.55	LPN
8004	10	12	17	14	13.3 $\pm$ 1.49	SCN
8007	15	11	10	14	12.5 $\pm$ 1.19	SCN
8011	N/A	8	11	8	9 $\pm$ 1.00	LPN

richness per count point for each transect to determine a species richness value balanced by effort. I then tested the species per count point richness values with one-way ANOVA and Kruskal-Wallis tests, but did not find species richness per point to differ among years (ANOVA, $F_{3,80} = 0.3704$, $p = 0.775$; K-W, $H_{3,83} = 1.722$, $p = 0.632$).

Differences among Habitat Types

Avian abundance. All fourteen of the common species were detected at least once in each habitat type in at least one year, except Nashville Warbler, which was never detected in subalpine conifer forest during the four-year study. Other species, including Steller's Jay and Fox Sparrow were only rarely detected in the higher elevation habitats, lodgepole pine and subalpine conifer. In contrast, in four years, Clark's Nutcracker was detected only twelve times in Sierran mixed conifer forest, nine of which were in 2013. Based on the distributions of individuals across habitats for each species, most species displayed a preference toward a subset of the total habitat types with low or variable evenness (E_{var}) measures. Only Dark-eyed Junco, Mountain Chickadee, Hammond's/Dusky Flycatcher, and Yellow-rumped Warbler had consistently high evenness across habitats, with E_{var} values of at least 0.80 for all four years (Table 4.3). The evenness of these four species, and less evenness of other species, is easily visualized in the figures of Figure 4.3 and Appendix D.

Common species only rarely showed a large abundance increase in one habitat type and large decrease in another habitat during the same year. From 2011 to 2012, Mountain Chickadee decreased in red fir forest by 15.3 individuals per point, and increased by 9.4 individuals at lodgepole pine points (Table 4.4 and Figure 4.3). Mountain Chickadee also decreased slightly in subalpine conifer and Sierran mixed conifer. The abundance for Mountain Chickadee reversed direction from 2012 to 2013, increasing by 6.6 individuals per point in subalpine conifer and 3.3 individuals in red fir, while decreasing by 7.2 individuals in lodgepole pine.

Table 4.3. Smith and Wilson evenness index (E_{var}) for common species. Values express the balance of a species' distribution across habitat types. E_{var} of 1.0 represents equal abundance in all habitats. Species are listed by four-year mean E_{var} (highest to lowest).

Species	2011 E_{var}	2012 E_{var}	2013 E_{var}	2014 E_{var}	Mean E_{var}
Mountain Chickadee	0.864	0.885	0.984	0.987	0.930
Dark-eyed Junco	0.907	0.871	0.898	0.977	0.913
Hammond's/Dusky Flycatcher	0.949	0.873	0.860	0.944	0.906
Yellow-rumped Warbler	0.879	0.894	0.981	0.839	0.898
American Robin	0.873	0.756	0.491	0.910	0.757
Western Wood-Pewee	0.883	0.881	0.401	0.648	0.703
Red-breasted Nuthatch	0.920	0.918	0.200	0.659	0.674
Cassin's Finch	0.681	0.457	0.703	0.800	0.660
Nashville Warbler	0.554	0.799	0.659	0.545	0.639
Western Tanager	0.632	0.865	0.359	0.676	0.633
Clark's Nutcracker	0.839	0.319	0.728	0.442	0.582
Steller's Jay	0.539	0.564	0.557	0.209	0.467
Fox Sparrow	0.517	0.629	0.229	0.315	0.423
Golden-crowned Kinglet	0.507	0.493	0.285	0.283	0.392

Table 4.4. Change in mean annual abundance by species. Abundance change values are the mean number of individuals at each count point within each habitat type for one year minus the mean value of the previous year in the same habitat. See Appendix E for 95% confidence intervals of each difference.

Species	2012 minus 2011				2013 minus 2012				2014 minus 2013			
	SMC	RFR	LPN	SCN	SMC	RFR	LPN	SCN	SMC	RFR	LPN	SCN
Western Wood-Pewee	0.6	-1.0	0	-1.4	-1.2	0.2	0.4	0.2	6.4	3.6	0.8	5.0
Hammond's/Dusky Flycatcher	-3.3	-8.2	-5.9	-3.6	3.1	1.5	2.9	0.8	10.9	4.5	7.4	11.4
Steller's Jay	-3.6	-4.0	0	-0.4	3.7	5.0	1.5	0.6	2.7	3.0	0.7	-0.8
Clark's Nutcracker	0.1	0.2	0.7	1.1	1.3	0.1	0	-1.8	-0.4	0.1	0.9	6.3
Mountain Chickadee	-2.4	-15.3	9.4	-2.2	4.2	3.3	-7.2	6.6	11.9	14.6	15.0	17.6
Red-breasted Nuthatch	-1.1	1.5	-3.8	0.6	6.0	7.6	0.3	-2.6	5.7	5.2	2.6	4.5
Golden-crowned Kinglet	-5.3	-2.4	2.5	1.3	2.3	26.6	4.9	-4.5	44.7	67.5	-2.5	4.4
American Robin	-3.8	-3.9	-2.0	0.5	2.4	1.6	5.8	0	-0.1	0.4	0.1	2.1
Nashville Warbler	-7.0	1.0	1.8	0	5.8	1.4	-1.8	0	3.0	20.4	2.6	0
Yellow-rumped Warbler	-4.8	-7.8	9.7	-0.8	4.9	-2.5	-0.1	0.5	9.9	39.1	11.8	2.1
Dark-eyed Junco	-15.6	-25.1	-8.5	-9.1	-5.3	17.7	9.4	0.5	21.6	4.5	19.9	17.1
Fox Sparrow	-9.2	-4.1	-0.3	-5.8	3.4	0.4	-0.6	0.4	7.4	-0.2	0.9	0.1
Western Tanager	-6.2	-5.3	0	3.8	2.2	1.3	0.3	-0.2	14.1	11.9	5.1	-3.6
Cassin's Finch	-1.4	-0.1	-0.4	1.0	3.0	0.6	3.5	0.8	-1.1	3.0	3.7	-1.2

Golden-crowned Kinglet increased tremendously in red fir from 2012 through 2014, and in Sierran mixed conifer from 2013 to 2014 (Appendix D). However, no considerable decreases occurred for Golden-crowned Kinglet in other habitats that would suggest a shift in habitat type. Rather, Sierran mixed conifer and especially red fir provide excellent resources for Golden-crowned Kinglet, making them prime habitat for this species.

Yellow-rumped Warbler had a similar pattern of abundance changes to those of Mountain Chickadee (Table 4.4 and Figure 4.3). Most notably, Yellow-rumped Warbler decreased in abundance in red fir forest by 7.8 individuals between 2011 and 2012, compared to an increase of

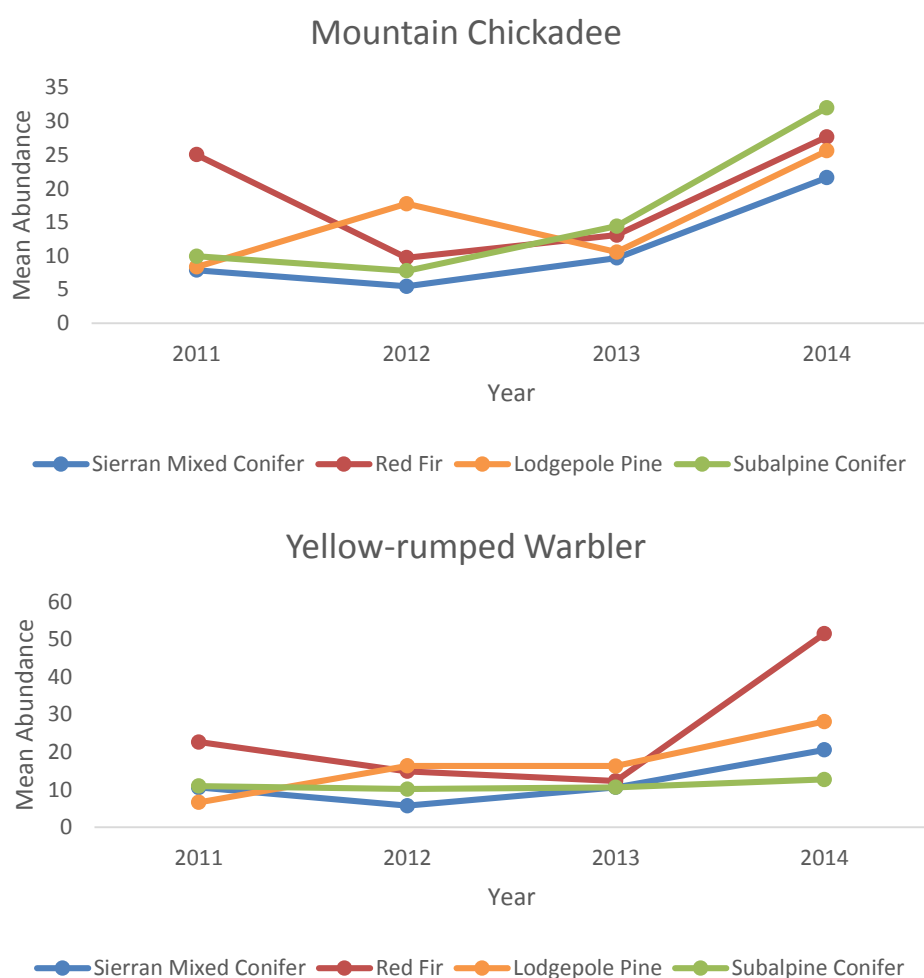


Figure 4.3. Annual mean abundance by habitat for Mountain Chickadee and Yellow-rumped Warbler. Abundance values are based on each 12.57 ha count point area. See Appendix D for abundance by habitat figures for the other twelve common species.

9.7 individuals in lodgepole pine forest. In Sierran mixed conifer, Yellow-rumped Warbler also decreased between 2011 and 2012.

Species richness. In each of the four years, the lowest elevation habitat, Sierran mixed conifer, maintained the highest mean species richness per count point, as expected (Figure 4.4). Also not surprisingly, mean count point species richness decreased with successively higher elevation habitats in each of the four years. For example, in 2011, Sierran mixed conifer forest had a mean species richness of 9.86 ± 3.26 species per count point, while red fir, lodgepole pine, and subalpine conifer hosted 7.18 ± 2.17 , 3.95 ± 2.12 , and 3.06 ± 1.71 species per point, respectively (Mean $\pm$ standard deviation; Table 4.4).

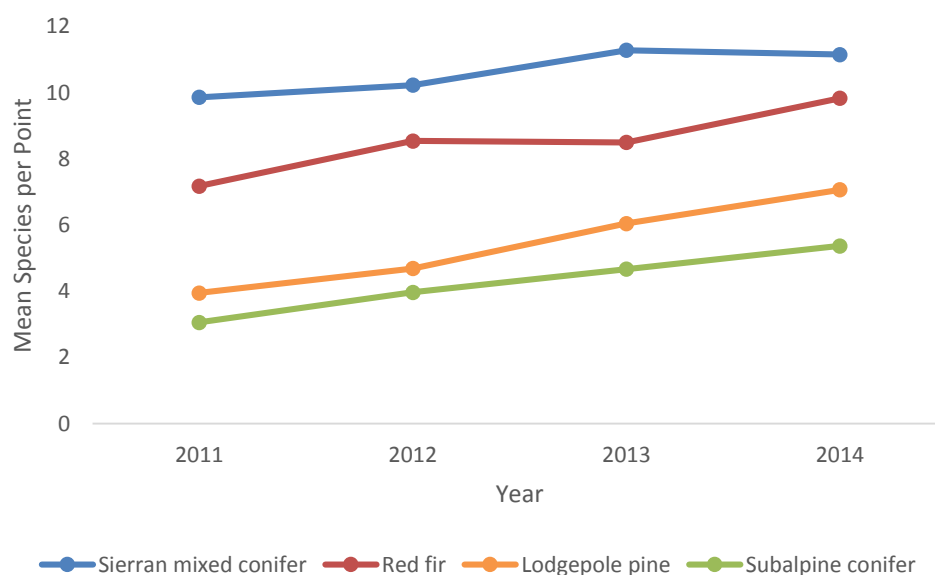


Figure 4.4. Annual mean species richness by habitat. Richness values are based on each 12.57 ha count point area within Sierran mixed conifer, red fir, lodgepole pine, and subalpine conifer forests in Yosemite, Sequoia, and Kings Canyon National Parks.

Interestingly, all four habitat types had a positive trend in species richness from 2011 to 2014 (Figure 4.4). This positive trend and its meaning are further discussed in the “Avian Richness by Habitat” section of Chapter V.

Table 4.5. 2011 richness *t*-tests for adjacent habitats. Table includes species richness mean and standard error for each habitat type in 2011, and two-sample *t*-test results between elevationally-adjacent habitat types. Richness values are based on the average number of species present at a count point within the specified habitat type. Habitat types are presented from lowest (top) to highest elevation. See Appendix B for complete list of species for richness analyses.

Habitat Pair	2011 Mean Species Richness $\pm$ S.E.	<i>t</i>	<i>p</i>	df
Sierran mixed conifer Red fir	9.86 $\pm$ 0.34 7.18 $\pm$ 0.38	4.365	< 0.0001	121
Red fir Lodgepole pine	3.95 $\pm$ 0.28	6.895	< 0.0001	87
Lodgepole pine Subalpine conifer	3.06 $\pm$ 0.24	2.403	0.009	107

Table 4.6. 2012 richness *t*-tests for adjacent habitats. Table includes species richness mean and standard error for each habitat type in 2012, and two-sample *t*-test results between elevationally-adjacent habitat types. Richness values are based on the average number of species present at a count point within the specified habitat type. Habitat types are presented from lowest (top) to highest elevation. See Appendix B for complete list of species for richness analyses.

Habitat Pair	2012 Mean Species Richness $\pm$ S.E.	<i>t</i>	<i>p</i>	df
Sierran mixed conifer Red fir	10.23 $\pm$ 0.27 8.54 $\pm$ 0.35	3.31	0.0006	160
Red fir Lodgepole pine	4.69 $\pm$ 0.28	8.62	< 0.0001	98
Lodgepole pine Subalpine conifer	3.97 $\pm$ 0.27	1.88	0.031	117

Table 4.7. 2013 richness *t*-tests for adjacent habitats. Table includes species richness mean and standard error for each habitat type in 2013, and two-sample *t*-test results between elevationally-adjacent habitat types. Richness values are based on the average number of species present at a count point within the specified habitat type. Habitat types are presented from lowest (top) to highest elevation. See Appendix B for complete list of species for richness analyses.

Habitat Pair	2013 Mean Species Richness $\pm$ S.E.	<i>t</i>	<i>p</i>	df
Sierran mixed conifer Red fir	11.28 $\pm$ 0.30 8.5 $\pm$ 0.42	4.73	< 0.0001	171
Red fir Lodgepole pine	6.05 $\pm$ 0.28	4.99	< 0.0001	100
Lodgepole pine Subalpine conifer	4.67 $\pm$ 0.26	3.58	0.0002	121

Table 4.8. 2014 richness *t*-tests for adjacent habitats. Table includes species richness mean and standard error for each habitat type in 2014, and two-sample *t*-test results between elevationally-adjacent habitat types. Richness values are based on the average number of species present at a count point within the specified habitat type. Habitat types are presented from by increasing elevation. See Appendix B for complete list of species for richness analyses.

Habitat Pair	2014 Mean Species Richness $\pm$ S.E.	<i>t</i>	<i>p</i>	df
Sierran mixed conifer Red fir	11.15 $\pm$ 0.31 9.83 $\pm$ 0.40	2.23	0.014	163
Red fir Lodgepole pine	7.07 $\pm$ 0.30	5.62	< 0.0001	99
Lodgepole pine Subalpine conifer	5.37 $\pm$ 0.32	3.87	< 0.0001	122

To further examine the relationships of species richness between habitats, I calculated the richness differences at each survey point between two consecutive years in a given habitat type. I then averaged the differences within each habitat type, and analyzed them using two-sample *t*-tests for elevationally-adjacent habitats. The *t*-tests of year-to-year richness changes between habitat types confirmed that one or fewer elevational shifts in richness occurred between habitats at the combined-species richness level. Only one of the nine *t*-tests for this analysis showed statistical significance of year-to-year richness difference between habitats. The mean species richness decrease between 2013 and 2014 for Sierran mixed conifer ($\bar{x}$ = -0.29) compared to the increase in richness for the same years in red fir ($\bar{x}$ = 0.88) may indicate an upslope shift in species from Sierran mixed conifer to red fir forest (p = 0.03; Table 4.11).

Table 4.9. Differences in species richness from 2011 to 2012 by habitat. Table expresses the change in mean species richness from 2011 to 2012 for each habitat type. Sample size (n) for differences between years is given in parentheses for each habitat. Two-sample *t*-test results for each pairing of elevationally-adjacent habitats are also shown, indicating no statistical differences in mean species richness change between habitats for these years.

Habitat Pair	Mean Difference (2012 minus 2011)	<i>t</i>	<i>p</i>	df
Sierran mixed conifer	0.61 (n = 93)	-0.75	0.23	128
Red fir	1.05 (n = 37)			
Red fir	0.73 (n = 56)	0.56	0.29	91
Lodgepole pine				
Lodgepole pine	0.45 (n = 56)	0.58	0.28	110
Subalpine conifer				

Table 4.10. Differences in species richness from 2012 to 2013 by habitat. Table expresses the change in mean species richness from 2012 to 2013 for each habitat type. Sample size (n) for differences between years is given in parentheses for each habitat. Two-sample *t*-test results for each pairing of elevationally-adjacent habitats are also shown, indicating no statistical differences in mean species richness change between habitats for these years.

Habitat Pair	Mean Difference (2013 minus 2012)	<i>t</i>	<i>p</i>	df
Sierran mixed conifer	0.75 (n = 122)	0.36	0.36	113
Red fir	0.56 (n = 43)			
Red fir	1.27 (n = 59)	-1.37	0.09	100
Lodgepole pine				
Lodgepole pine	0.72 (n = 61)	1.13	0.13	118
Subalpine conifer				

Table 4.11. Differences in species richness from 2013 to 2014 by habitat. Table expresses the change in mean species richness from 2013 to 2014 for each habitat type. Sample size (n) for differences between years is given in parentheses for each habitat. Two-sample *t*-test results for each pairing of elevationally-adjacent habitats are also shown, indicating a statistical difference in mean species richness change between Sierran mixed conifer and red fir for these years.

Habitat Pair	Mean Difference (2014 minus 2013)	<i>t</i>	<i>p</i>	df
Sierran mixed conifer	-0.29 (n = 125)	-1.87	0.03	77
Red fir	0.88 (n = 40)			
Red fir	1.02 (n = 60)	-0.24	0.41	98
Lodgepole pine				
Lodgepole pine	0.60 (n = 63)	0.78	0.22	121
Subalpine conifer				

Snow Sensor Station Records

The snow sensors recorded significantly deeper average April snow in 2011 ($\bar{x} = 220.1$ cm) than in any other year (Figure 4.5; one-way ANOVA, $F_{3,60} = 45.0$, $p < 0.0001$). In addition, based on two-sample t -tests, the snow depths in Yosemite National Park and Sierra National Forest, the northern half of the study area, were not statistically different from depths in the southern part, Sequoia and Kings Canyon National Parks and Sequoia National Forest, in any year (Table 4.12).

Table 4.12. T -test values for northern and southern snow sensors. Snow depth values were compared between northern and southern snow sensors using t -tests for each year. Statistical values are shown, explaining that the snow depths did not differ between the northern and southern sensors in any year.

Year	t	p	df
2011	0.73	0.24	11
2012	1.09	0.16	7
2013	0.94	0.19	9
2014	0.36	0.36	9

The calculated Panther Meadow snow depth for 2011 ($\bar{x} = 328.2$ cm), with a sensor elevation of 2,621 m, was greater than the 2011 mean depths at every higher elevation sensor, including Upper Tyndall Creek ($\bar{x} = 176.0$ cm, 3,475 m elev.) and Crabtree Meadow ($\bar{x} = 134.6$ cm, 3,261 m elev.). The cause and potential importance of this difference is discussed in Chapter V. In the northern snow sensors, the Tuolumne Meadows sensor recorded much lower snow depths than the sensors at similar elevations (Figure 4.5).

Snow Depths and Avian Populations

No species in any year showed a strong correlation between transect-level avian density and snow depth. In fact, only Clark's Nutcracker in 2011 and 2012 and Fox Sparrow in 2014 had a correlation coefficient of at least ± 0.6 (Table 4.13). Unexpectedly, the correlation coefficients for Clark's Nutcracker in 2011 and 2012 were both positive, implying that as snow depth increased,

the density of Clark's Nutcracker increased as well. For all correlation tests, I did not include transects with densities of zero. For some species, such as Nashville Warbler, this removed a

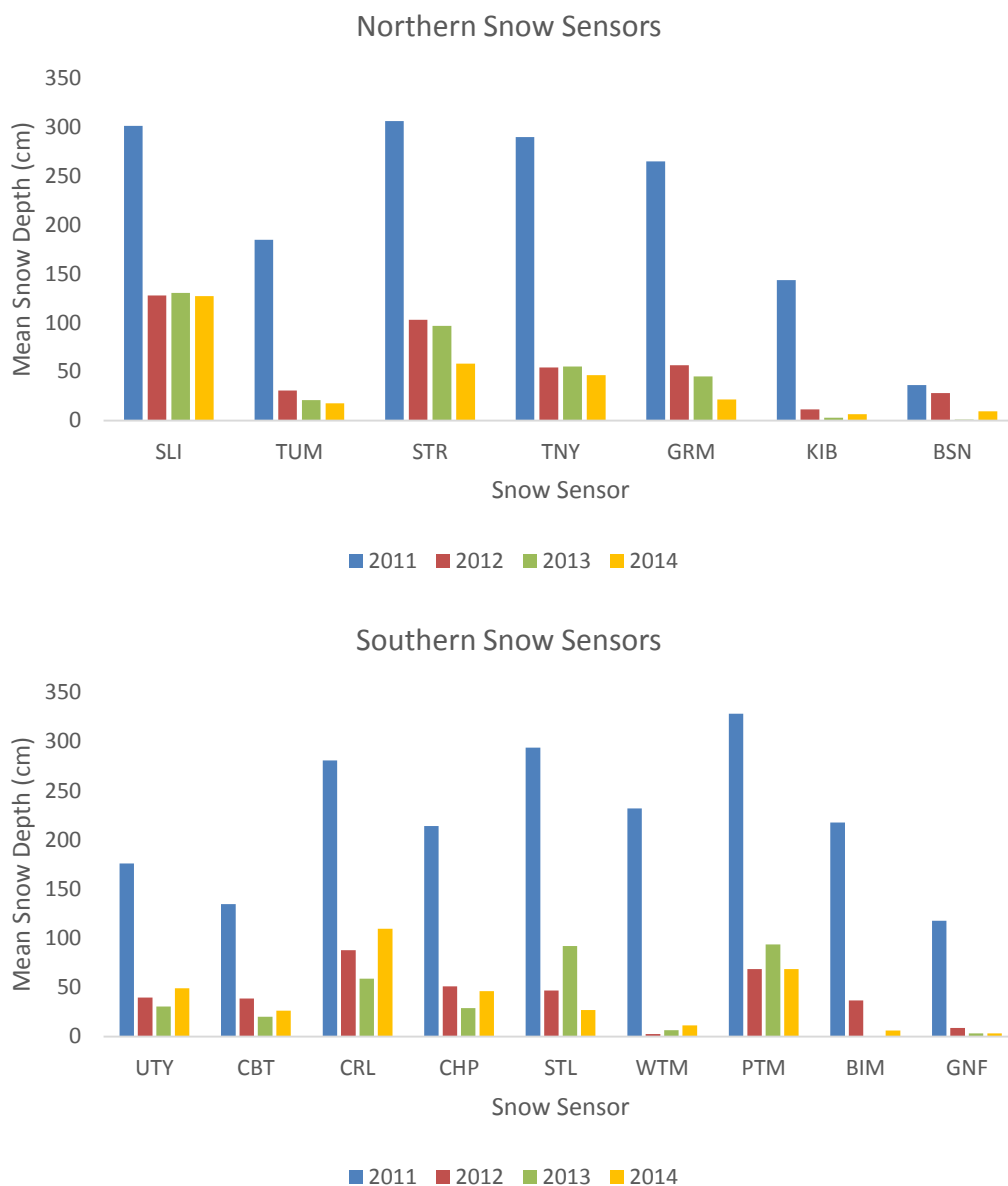


Figure 4.5. Mean April snow depths by sensor. Mean snow depth in April at each snow sensor in the northern portion (Yosemite National Park and Sierra National Forest) and in the southern portion (Sequoia and Kings Canyon National Parks, and Sequoia National Forest) of the study area are shown for 2011 through 2014. Snow sensors are listed in descending order by elevation. See Table 3.1, p. 18 for full snow sensor names.

considerable number of data points, but including the zeroes would have given inaccurate correlation coefficients for the species whose distributions do not extend into the highest or lowest habitats.

Table 4.13. Correlations between snow depth and species density. Correlation coefficients (r-values) for each species in each year, comparing the independent variable of snow depth to the dependent variable of species density (individuals per hectare) are shown. Species are listed in descending order by highest single-year correlation coefficient ($\pm$).

Species	2011	2012	2013	2014
Clark's Nutcracker	0.60	0.79	-0.32	0.14
Fox Sparrow	0.29	0.04	0.15	-0.68
American Robin	0.53	0.07	0.48	-0.01
Nashville Warbler	0.52	0.36	0.35	-0.15
Red-breasted Nuthatch	0.11	-0.46	-0.26	-0.22
Mountain Chickadee	0.45	0.08	0.20	0.08
Yellow-rumped Warbler	0.43	0.06	-0.10	-0.08
Western Tanager	-0.40	0.01	-0.10	-0.30
Cassin's Finch	0.30	-0.37	0.11	0.13
Golden-crowned Kinglet	0.23	-0.32	-0.12	-0.11
Hammond's/Dusky Flycatcher	0.30	-0.13	-0.17	-0.23
Western Wood-Pewee	0.24	-0.21	-0.27	0.23
Dark-eyed Junco	-0.05	-0.04	-0.17	-0.14
Steller's Jay	-0.10	0.13	-0.09	-0.06

CHAPTER V

DISCUSSION AND CONCLUSIONS

Avian Abundance among Years

Only Dark-eyed Junco, Mountain Chickadee, Red-breasted Nuthatch, and American Robin exhibited statistical differences in abundance among years. Abundances of Dark-eyed Junco and American Robin both decreased greatly from 2011 to 2012, as did abundances of Fox Sparrow, Western Tanager, and Hammond's and Dusky Flycatchers, though not significantly for the latter species. Of these species, Dark-eyed Junco, American Robin, and Fox Sparrow are primarily ground foragers, eating seeds or soil-based arthropods. The deep snow of 2011 that persisted well into the avian breeding season may have made access to food difficult, decreasing likelihood of nesting in 2011 for these species (Robinson *et al.*, 2007). A reduced number of nesting birds in 2011 could result in fewer overall birds of that species present the following breeding season, though neither DeSante (1985) nor Hejl *et al.* (1988) found adult fecundity responsible for decreases in abundance. Dark-eyed Juncos spend winters at much lower elevations before moving upslope to breed in the Sierra Nevada (Beedy and Pandolfino, 2013). A year of deep spring snowpack, such as 2011, may have resulted in lower overall breeding elevations, as many of these species can breed at a variety of elevations (Beedy and Pandolfino, 2013), though this occurrence would more likely show depressed abundance in 2011 and a rebound in 2012.

Additionally, the prolonged snow cover may have delayed the hatch times of aerial insects upon which Hammond's and Dusky Flycatchers feed, altering the time of year or degree to which these species bred in 2011. However, Western Wood-Pewee and Yellow-rumped Warbler are also largely dependent upon aerial insects for food, and none of these species experienced the same

dramatic decrease in abundance. The correlations of avian densities to snow depths in this study do not specifically support these conclusions, nor did this study measure availability of insects, but these conclusions should still not be ruled out as potential drivers of change in annual abundances.

Surprisingly, all of the common species except Western Wood-Pewee and Clark's Nutcracker had sequentially higher abundances from 2012 to 2014. This would seem to suggest the occurrence of a broad, environmentally-favorable change before or during those years. Decreases in mean April snow depth from 2011 through 2014 (Figure 4.5) hint at the possibility that a negative correlation may exist between snow depth and annual abundance, but this was not statistically supported by the data (Table 4.12). It could require several consecutive years of deep spring snow (or lack thereof) and a longer-term study to elucidate a correlation between these variables. An extended study over several cycles of drought and heavy snow could potentially identify the compounding effects heavy snow following several years of drought on avian abundance. Future research should also examine bird population patterns following consecutive heavy snow years.

Clark's Nutcracker's ability to cache seeds for later use, its documented fluctuations in population density depending on the season (Lorenz and Sullivan, 2010), and no evidence to show that Clark's Nutcracker does not breed because of deep snow (Schaming, 2015), may exclude it from the list of species potentially affected by deep snow. Also, because of deep snow in 2011, field crews did not survey several of the high elevation transects. More so than any of the other common species in this study, Clark's Nutcracker favors high elevations, thus the reported 2011 abundance may not exactly reflect the actual 2011 abundance for Clark's Nutcracker.

The increasing abundance trend from 2012 to 2014 for such a variety of species could potentially reflect a difference in observer effort or DISTANCE model estimations. I was one of the observers for all three years (2012-2014), and am familiar with the abilities of all of the observers to detect birds. I have no reason to believe any meaningful differences in observation abilities existed from one year to the next that would result in such widespread abundance increases in many bird species. The increase is not likely a result of differences in the number of surveys

completed each year either. In 2012, observers surveyed 284 points, compared to 290 in 2013 and 283 in 2014. The models I chose in DISTANCE to estimate density and abundance could have had some effect on the outcomes, but I tested many model and series adjustment combinations, and am confident that I chose appropriate combinations to fit the data. The models I selected did not differ from one year to the next either, so any effect the models had on the densities ought to have been equal for all years.

Species Richness among Years

Avian richness did not differ among years by any measure of analysis. ANOVA and Kruskal-Wallis tests for annual richness per transect and per count point did not identify differences among years in the number of species present. This is not unexpected considering relative stability of the richness measurement and the large number of transects and survey points at which I measured the richness. For a species to be included in the richness value, it needs only to appear once on a given transect or count point, depending on which one I am using to measure. For example, a count point with ten Mountain Chickadees, one Cassin's Finch, and one Western Tanager has the same richness value as a point with four Mountain Chickadees, four Cassin's Finches, and four Western Tanagers: three. Therefore, even if the abundance of every species on a point count declined sharply, as long as a single individual of each species was still present, the point would have the same richness as it did before the decline.

As mentioned in Chapter III, I did not include raptors, waterfowl, aerial insectivores, and species occurring on less than five percent of total transects. I removed raptors and aerial insectivores from consideration because of their foraging methods. With a typical foraging pattern of flying high above the ground and often covering a large area in a short time, their presence on a point count is more coincidence than a direct relation to any resources that the habitat around the point count provides. Similarly, the presence of waterfowl is dependent upon a nearby, suitable body of water, and as only a small percentage of points occurred near suitable water, it would have

biased the richness values to include waterfowl. Finally, removing species that occurred on less than five percent of transects helped to mitigate the effects of “accidental” sightings on the richness values. Several of the accidental sightings, including Hutton’s Vireo and Western Scrub-Jay, occur in habitats adjacent to those in this study, but are not representative species of any of my focus habitats and were therefore removed.

It is of no consequence in this case, but worth noting that calculating the richness at the count point level tended to favor the transects with fewer total points. For example, if two transects each had detections of thirty species in a year, but one transect had fifteen count points while the other had only ten, the per point species richness, 2 and 3 species per point, respectively, would appear quite different between them. Perhaps, adding another five points to the second transect would bring its total species to forty-five, thus maintaining the 3 species per point average. More likely, though, as appears to have occurred in this dataset, a certain threshold exists for number of count points after which the total species richness plateaus.

Avian Abundance by Habitat

Mountain Chickadee and Yellow-rumped Warbler showed year-to-year changes in abundance in elevationally-adjacent habitat types that may suggest an upslope or downslope shift by a species during a breeding season. The large decrease in Mountain Chickadee abundance in red fir forest compared to the increased abundance in lodgepole pine from 2011 to 2012 illustrates a potential upslope movement of Mountain Chickadees during the breeding season of 2012.

Although the results do not specifically support the influence of snow on this shift, the deep snow of 2011 may have forced more Mountain Chickadees to breed in lower elevation red fir forest. In turn, the following year’s low snowpack across habitats may have allowed Mountain Chickadees to return to the higher lodgepole pine forests, which are, in addition to red fir, important habitat for many cavity-nesting species including Mountain Chickadee (Purcell, 2002). However, chickadees then decreased in abundance in 2013 within lodgepole pine forests, while they increased in each of

the other three forest types. This distribution could have been a result of a boom in the population during the 2012 breeding season, after which offspring dispersed into all habitats in order to maximize foraging and nesting opportunities. Purcell (2002) found that abundance for cavity-nesters peaked in red fir and lodgepole pine forests, so it might be expected that there would be regular fluctuation in abundance between red fir and lodgepole pine habitats. I did not examine other environmental factors that could have been responsible for the fluctuations among habitat types, and therefore cannot conclude how any additional environmental factors affected abundance, though one or more may have played a role.

Yellow-rumped Warbler also notably increased within lodgepole pine forest from 2011 to 2012, while decreasing in the other three habitats, especially red fir. As insectivores, Yellow-rumped Warblers primarily use a sallying behavior to grab insects in mid-air before returning to a perch. As mentioned with other species, their reliance upon insects may have driven them to lower elevations in 2011 if insects were not plentiful in the high elevation forests. Also, the positive change in Sierran mixed conifer forest between 2012 and 2013 makes the 2011 to 2013 change essentially zero for this species. With a relatively small negative change followed by small positive change in Sierran mixed conifer, this probably implies that the abundance changes in Sierran mixed conifer are within the normal annual variation for Yellow-rumped Warbler.

Recent surveys within mixed-conifer habitats of Carson National Forest in New Mexico and Eldorado, Stanislaus, and Tahoe National Forests of the Sierra Nevada recorded ranges of Yellow-rumped Warbler densities that suggest those of my study fluctuated within expected bounds (Beason and Leukering, 2005; Fogg *et al.*, 2015). In 2014, Yellow-rumped Warbler abundances increased from the previous year by at least 9.9 individuals on average in three habitats, with an increase of 39.1 individuals per count point in red fir forest. This may have been a result of widespread successful recruitment of the previous year's offspring into the breeding population, which can be an important source of higher abundance in Sierra Nevada birds (DeSante, 1985; DeSante, 1990). The lack of deep snow during the 2013-2014 winter may have allowed for greater

survival of young birds (Hejl *et al.*, 1988), and the subsequent abundance surge of many of the common bird species in 2014. For Yellow-rumped Warbler, Mountain Chickadee, and other species that regularly inhabit both red fir and lodgepole pine forests, some ambiguity may exist in the results because of a significant elevational overlap between these species. In lodgepole pine forest, red fir often composes a small percentage of total individuals. Although red fir frequently grows in pure stands, lodgepole pine is occasionally present within this habitat type as well.

The effect of small numbers of detections in specific habitats contributing to overestimated abundances occurred in isolated cases for Cassin's Finch, Clark's Nutcracker, Fox Sparrow, Golden-crowned Kinglet, Nashville Warbler, Red-breasted Nuthatch, Western Tanager, and Western Wood-Pewee. All of these species have in common a less-even distribution among habitats (Table 4.3), and showed preference for one or two habitat types. The lower evenness for these species brings to the forefront an important concept that likely influenced the distributions among habitats for these species. Species that show a strong preference for one or few habitats may not have a choice but to breed in those habitats despite the unfavorable environmental conditions. On the other hand, species that have a more even distribution among habitats, such as Mountain Chickadee, Dark-eyed Junco, and Yellow-rumped Warbler, may have more flexibility to shift between habitats to avoid poor conditions. In other words, except in extreme cases, environmental conditions may not be as strong of a determining factor as is the need for specific resources in specific habitat types.

Species Richness by Habitat

As is generally expected across a range of elevations, and supported by Purcell (2002) and Hejl *et al.* (1988), the lowest elevation forest type, Sierran mixed conifer, of Yosemite, Sequoia, and Kings Canyon National Parks hosted the highest avian species richness of the four forest types. In ascending elevation and descending richness, red fir, lodgepole pine, and subalpine conifer were the second, third, and fourth-most species rich habitats.

As mentioned in Chapter IV, all four habitat types had a positive trend in species richness from 2011 to 2014. Even nearly every year-to-year change in species richness across all habitat types had positive growth. This growth in species richness for each subsequent year in all habitats suggested that species may have been fluctuating between elevationally-adjacent habitat types outside of their typical distributions. In other words, a species that regularly occurs in Sierran mixed conifer and red fir forests may have been detected in lodgepole pine and/or subalpine conifer forest in subsequent years, increasing the mean richness in higher elevation habitats. Conversely, observers may have detected species more normally found at high elevations in Sierran mixed conifer, in turn increasing each year's mean richness in SMC. A pattern of broader fluctuation of species outside of their typical habitats often implies either increased competition for resources or a general lack of resources within the normal habitat(s). The latter seems especially possible with ongoing drought potentially reducing available food or nesting habitat (Bolger *et al.*, 2004).

The changes within habitats from 2012 and 2013 present an interesting finding. While red fir forest showed a slight decrease in species richness from 2012 to 2013 (8.54 to 8.50 species, $p = 0.47$), both Sierran mixed conifer and lodgepole pine had their greatest increases in combined species richness between those years (Sierran mixed conifer: 10.2 to 11.3 species, $p = 0.006$; lodgepole pine: 4.7 to 6.1 species, $p = 0.0005$). This combination of stagnation in one forest type and growth in two others could imply species movements from red fir into each of the other forest types. The similarity in red fir richness between 2012 and 2013 does not necessarily mean that the same eight to nine species were present in each year. One species may have been regularly detected one year but not in the next; while a different species that was not found in red fir in 2012 could have been present in 2013. However, because the richness value for each habitat type is the mean of dozens of point counts, I cannot assume that the similar richness values are as simple as no change in the species composition, or one species replacing another.

Within Sierran mixed conifer and lodgepole pine on the other hand, an increase in richness by more than one species in each habitat occurred between 2012 and 2013. Observers created and

surveyed ten more Sierran mixed conifer count points in 2013 than were surveyed in 2012, an eight percent increase in the total points. It is possible that the additional seventy minutes of point counts at these new points provided the opportunity to consistently detect greater species variety throughout Sierran mixed conifer forest. Lodgepole pine forest counts increased by only one point (less than two percent), however, and thus the increase in lodgepole pine species richness is almost certainly not due to a difference in time spent observing.

The survey dates may also influence the species richness within these forest types. For example, I surveyed transect 8001 with another observer on May 24, 2013, approximately the time of year we intended to visit. The previous year, my field partner and I postponed our survey at transect 8001 until July 20 because of a prescribed burn that had occurred earlier in the season near this transect. In 2013, we recorded thirty-eight species along this transect compared to only thirty-one species in 2012. The differences in richness were probably a combination of post-breeding, upslope migration by some species (Siegel *et al.*, 2011) and the relatively less singing that was occurring during the late 2012 survey, reducing the detectability of many birds (Alldredge *et al.*, 2007). Nashville Warbler, a known upslope migrant, exhibited a particularly good example of the former cause. In 2012, we failed to detect any Nashville Warblers along transect 8001, whereas in 2013 they were abundant both during and between point counts (*pers. obs.*, and unpublished SIEN data, 2013).

A similar situation may have affected lodgepole pine species richness. The survey of transect 6014 on July 19, 2012 detected only thirteen species, while the same transect on June 30, 2013 had twenty-five different species. As a comparison, we detected twenty-three species along the same transect on July 2, 2014. The driving factor for species richness in this case was again probably the lack of singing birds later in the season. While we attempted to visit a given transect during the same few days in each year, occasionally weather or other logistical challenges made that impossible.

Notes about Wildfire in the Study Area

As I mentioned above, National Park Service fire crews conducted a prescribed burn at transect 8001 in Kings Canyon National Park during our 2012 avian field season. Additionally, between the 2013 and 2014 field seasons, the Rim Fire burned approximately 104,131 ha in Stanislaus National Forest and Yosemite National Park, including chaparral, oak woodland, and coniferous forest habitats around annual transect 6027 and five alternating transects. Birds and other wildlife species typically flee habitats with active wildfires, creating a temporary reduction in biodiversity in those areas. A reduction in biodiversity probably occurred at transect 6027 in August, September, and October 2013 while the Rim Fire burned, though I have not found data to confirm this.

A Point Blue Conservation Science study of two recent fires in the Sierra Nevada evaluated specific abundance in burned and unburned forests for thirteen of the fourteen common bird species that I studied (Fogg *et al.*, 2015). Seven of the thirteen species showed no difference in abundance between burned and unburned forests. Four species, Hammond's Flycatcher, Golden-crowned Kinglet, Yellow-rumped Warbler, and Red-breasted Nuthatch were more abundant in unburned forest, and two species, Fox Sparrow and Western Wood-Pewee had greater abundances in burned forest. Fox Sparrow and Western Wood-Pewee generally select more open habitat than many of the other species in my study, which explains their increased abundance after fires. Wildfire can result in slightly reduced abundance and species richness values (Fogg *et al.*, 2015), though the post-fire succession process favors some species over others at different times during regeneration (Raphael *et al.*, 1987), making the time-after-fire a crucial variable in the interpretation of post-fire avian population values. Initially, post-fire conditions favor ground and shrub foraging species, whereas "foliage-searching" species begin to increase in burned areas at a later successional stage (Raphael *et al.*, 1987).

I did a post-analysis investigation of raw abundance for several species at transect 6027, comparing 2013 to 2014. Western Wood-Pewee and Fox Sparrow abundances were essentially the

same, with eleven and nine Western Wood-Pewees, and one and one Fox Sparrows in 2013 and 2014, respectively. Fox Sparrows often favor montane chaparral habitat adjacent to or within open forest, and may have been more abundant overall in chaparral surrounding transect 6027 than in the mixed conifer forest included in my analyses. On the other hand, we detected nine Yellow-rumped Warblers at transect 6027 in 2013 and only one in 2014. Raw abundance for Red-breasted Nuthatch increased from five individuals in 2013 to eight individuals in 2014. Coincidentally, transect 6027 exhibited the highest species richness of all annual transects, with 37 species detected in 2013 and 35 in 2014, showing no clear effect of the Rim Fire on richness. Although I did not directly analyze potential effects of wildfire on avian populations at my study sites, it is important to consider that various factors besides or in addition to snow depth may have influenced the avian abundances and species richness of my study at least as much as the snow depth.

Consideration of Snow Depth Results

Generally, the readings of snow depth from the sensors successfully projected the known pattern across the four years of the study: well-above average snow depths in 2011, lasting into the avian breeding season, followed by three years of historically dry winters in the Sierra Nevada (Figure 4.5). Unfortunately, many of the sensors malfunctioned at various dates in April of the four study years, recording snow depths for days or weeks at a time that were obviously inaccurate. I detailed the methods I used to work around these errors in Chapter III, but was nonetheless left with educated guesses for several of the monthly mean depths, which in a few cases, were probably not accurate.

The Panther Meadow sensor provides the most obvious case for which the data may be incorrect. Despite the fact that the Panther Meadow sensor is only the seventh highest of nine sensors in elevation in the southern study area, the mean April 2011 snow depth was the highest of any site. The depths at Panther Meadow in 2012 through 2014 are also relatively high compared to the higher elevation sensors. This sensor may hold snow for a longer period of time than some of

the higher sensors because of its position on a northeast aspect, whereas the sensors at State Lakes, Upper Tyndall Creek, Chagoopa Plateau, and Wet Meadows are all on generally south-facing slopes. All of these sensors, including Panther Meadow are positioned in meadows, and should therefore have comparable sun exposure in that respect. The depth readings at the highest sensors, Upper Tyndall Creek and Crabtree Meadow, may be influenced by strong winds, though the exact effects are unknown as the wind could just as likely blow the snow away as it could create deep drifts. In short, I am aware that the potential inaccuracies of the snow depth calculations at several sites may have confounded the resulting relationships between snow depth and bird populations, though unlikely to a degree that would drastically affect my conclusions.

Discussion of Snow Depth related to Bird Density

None of the common bird species in this study displayed a strong correlation between their abundance and snow depths in any of the four years. Numerous factors may have driven this result. First, perhaps none of the snow depths were extreme enough, either high or low, to drastically change population numbers of these species. Also, the widespread range of avian survey dates during each year may lessen the importance of the snowpack in April. Although we began conducting point counts at low elevations by the third week of May each year, we did not typically survey at high elevations until late June at the earliest. It may be that measurements of snow, or near lack of snow, during April had no bearing on the activity or presence of birds at higher elevations when we surveyed in June or July. Especially in the years when snow had melted at all elevations by mid to late May, it is possible that it did not matter to breeding birds that a small amount of snow was present. As my study did not measure avian productivity, it is still possible that snow depths affected onset or success of nesting (Siegel *et al.*, 2014) but that other factors outside of the breeding season normalized abundances across years in such a way that snow depth did not correlate with abundance.

As mentioned in Chapter IV, Clark's Nutcracker and Fox Sparrow were the only species to show an abundance to snow depth correlation of at least 0.60. The Clark's Nutcracker correlation coefficients of 0.60 and 0.79 in 2011 and 2012, respectively, suggest that nutcracker abundance increased with deeper snow. This contradicts a previous study in the Sierra Nevada, which suggested that rather than tolerating high snowpack, Clark's Nutcrackers moved to lower elevations on the eastern slope to overwinter and breed, before returning to subalpine sites for seed caching and harvesting in late summer (Tombach, 1978). Assuming most individuals of this species still maintain such a pattern, this seems to identify a drawback in my study design for Clark's Nutcracker, similar to the date difference in snow measurement and survey completion mentioned above. In 2012, observers only detected six Clark's Nutcrackers (seven percent of annual total) on surveys before June 29, and in the high snow year of 2011, only two individuals (three percent) before June 29. This is perhaps largely a product of the types of habitats surveyed before and after this date, and their suitability for nutcrackers, but may also represent a dearth of nutcrackers at high elevation before July. If the latter is the case, the numbers of nutcrackers present at subalpine sites during the April snow sensor measurements may drastically differ from the abundances observed two to three months later during avian surveys.

On the other hand, Clark's Nutcracker has also been documented choosing low-elevation sites for seed caching (Lorenz *et al.*, 2011). This could imply that while Clark's Nutcracker relies on high elevations for the production and acquisition of its primary food sources, encountering this species at higher elevations in the summer is more coincidental; and that pairing the summer detection data with snow sensors from similar elevations does not adequately explain the relationship between this species and snow depths. Single species studies often have limitations for understanding broad ecosystem dynamics, but such studies may be particularly useful in the future to identify how and when single species use specific habitats. This would allow for more detailed protocols when studying bird species relationships to snow depth.

Fox Sparrow annual abundance showed an inverse correlation with snow depth ($r = -0.68$) in 2014, a relatively strong relationship compared to the previous three years for Fox Sparrow (2011: $r = 0.29$, 2012: $r = 0.04$, 2013: $r = 0.15$). The stronger inverse correlation of 2014 is most likely an anomaly, driven by low sample size ($n = 9$) and low snowpack. We only detected Fox Sparrow along nine of the twenty transects that were included in the correlation, which is similar to its distribution among transects in other years (Table 3.3). High annual density estimates at transects 8001 and 8013 also skewed the correlation. Mean snow depths at the corresponding snow sensors for Fox Sparrow detections varied from 3.0 to 68.6 cm, but depths were clumped into two groups. Four of the depths were less than 12 cm, while the other five were greater than 46 cm. Consequently, with no snow depth readings between these values, it is difficult to confirm the negative correlation between Fox Sparrow abundance and snow depth. Despite no definitive conclusion of snow depth correlating with avian abundance in this study, because models have predicted decreases in April snowpack of five percent by 2030 and greater than 33% by 2060 (Knowles and Cayan, 2002), it is imperative to confirm or refute this lack of relationship with further studies immediately.

Drawbacks of snow sensor data and avian transect pairing. Unfortunately, because snow depth data were not collected at the exact location and time of each avian transect, I had to rely on the assumed accuracy of the snow sensors. In general, most sensors provided at least a few good years of data during April. In years when data were clearly incorrect or missing entirely, though, I had to use other methods, as detailed in Chapter III, to obtain a complete picture of the snow depths over the four years. While I was careful to obtain and calculate the most accurate snow depths possible, the variety of methods I was forced to use resulted in a patchwork of readings. I cannot be certain that the calculations I made for missing or incorrect values were representative of the actual snow depths, and this may have influenced the accuracy of the statistical analyses.

Further, although I began this research under the impression that there were more than enough snow sensors present throughout my study areas to independently match one to each of the avian transects, the actual number that were active and could provide accurate data for the month of April were far fewer. Consequently, I had several cases where multiple avian transects had to pair with the same snow sensor (Table 3.1). This meant that I attributed the same amount of snow to multiple avian abundances. I cannot say that this result would not have occurred anyway with sensor-transect independence, but being able to match each transect with its own snow sensor and depths would have most likely better-distributed the data points in the correlation tests.

Climate Change Considerations

Although a changing climate does not likely have a major effect on four years of avian population data, the fluctuations in abundance among years in this study may infer a broader trend with climate change in forest habitats. The seasonality of high elevation forests, such as those in the Sierra Nevada, makes for a short time in which food, especially insects, is abundant. Changing climate has created greater variation in the peak time of food abundance, and populations of long-distance migrants have declined in temperate forests as a result of mismatched timing between food peak and migrant arrival (Both *et al.*, 2009). It would certainly be reasonable to expect peak time for insects to have varied between the high snowpack year of 2010-2011 and the extremely dry winters of 2012-2013 and 2013-2014. Perhaps the low snowpack of 2013 allowed for an early and extended foraging period for the insect-eating Neotropical migrants named earlier, leading to a prolific breeding season, and the considerable increases in abundance that all of these species showed in 2014.

The tendency for long-distance migrants to have smaller clutch sizes and lower annual fecundity than residents and short-distance migrants (Böhning-Gaese *et al.*, 2000) may also negatively affect long-distance migrants with increasing variability in snowpack and food availability. Many studies have documented a shift in arrival time at the breeding grounds for

migrating birds, with many species arriving days earlier now than historically. This changing phenology may particularly benefit short-distance migrants, such as American Robin and White-crowned Sparrow, as there has been a greater change in first arrival date for short-distance migrants than long-distance migrants (Végvári *et al.*, 2010). Ultimately, however, population fluctuations potentially due to climate change can be difficult to differentiate from density-dependent fluctuations, but continued, long-term studies of avian communities should aid in making the distinction.

Factors Outside of the Sierra Nevada

The fluctuations in abundance and richness that I observed among migratory songbirds in this study may not have occurred as a result of environmental or biological factors in the Sierra Nevada. Western Wood-Pewee, Hammond's and Dusky Flycatchers, Nashville Warbler, and Western Tanager are all Neotropical migrants, arriving in the Sierra Nevada for breeding in April or May each year. Some Yellow-rumped Warbler individuals may migrate from the tropics each spring also, but many spend winter at lower elevations in southern California. Long distance, migratory songbirds face some of their greatest challenges away from the breeding grounds. Small songbirds may experience up to 85% of their annual mortality during migration (Silleet and Holmes, 2002). Similarly, Sæther and Engen (2010) concluded that mortality during the non-breeding season has the greatest effect on population declines in songbirds and other altricial species. The search for food and adequate stopover sites compound the physiological struggle of long-distance migration (Lindström, 2003; Altshuler and Dudley, 2006), with habitat degradation and loss causing increased competition and stress for birds (Klaassen *et al.*, 2012).

The dynamics between urban landscapes and riparian corridors can also determine the quality of the corridors for use by migrating birds. Riparian areas with high tree cover area and less, close-proximity urban development had higher abundance, species richness, and more even distribution of native birds in a study of Cincinnati, Ohio, United States (Pennington *et al.*, 2008).

Pennington *et al.* (2008) found Neotropical migrants to be particularly sensitive to the spatial area of urban development, and more positively correlated with the presence of native vegetation than any of the other bird guilds.

The quality and conditions at the wintering grounds of migrants can also affect reproductive success at breeding sites. Male American Redstarts, rare migrants within eastern California, which spent winter in suboptimal habitats arrived later to breeding grounds than males in high quality habitats, while females produced earlier and more offspring when they had wintered in good habitat (Norris *et al.*, 2004). Further, bird species with specific habitat requirements are likely more susceptible to habitat degradation at their wintering grounds. Mangrove forests provide habitat for at least 140 bird species, with seventy-seven Neotropical migrants inhabiting these forests during winter or migration (Bayly and Gomez, 2008). The increasing removal and degradation of Central American mangrove forests as a result of pollution, timber harvest, and development threaten wintering migrant birds that rely on mangroves. Also, bird species with high site fidelity in winter habitat could be particularly imperiled if they cannot find suitable habitat near a removed or degraded habitat of former inhabitation (Warkentin and Hernández, 1996). Common Yellowthroat, a rare and declining breeding bird within the Sierra Nevada, prefers inhabiting black mangrove forest during the winter, with 51% of Common Yellowthroat individuals found in mangroves and the remaining 49% divided among other habitats in the Corozal District of Belize (Bayly and Gomez, 2008). Other migrant songbird species, including Northern Parula and Northern Waterthrush, although only rarely present in the Sierra Nevada, specifically rely on black mangrove to an even greater degree than Common Yellowthroat.

A study of avian-plant community relationships along the Salado River in Puebla and Oaxaca, Mexico identified a rich diversity of wintering Neotropical migrants, making up 53.6% (forty-three species) of the total bird species in riparian mesquite habitat (Arizmendi *et al.*, 2008). The aridity of this region makes riparian habitat a valuable resource for human inhabitants as well, and human disturbance has created a vegetation mosaic. Areas with the greatest plant densities

harbored the highest bird abundances and species richness, whereas habitats with patchy vegetation and more human visitation had the lowest population values. In fact, 59% of the wintering migrants exclusively used the undisturbed habitat, and only 3.7% of birds solely inhabited the most-disturbed habitat (Arizmendi *et al.*, 2008). Interestingly, Arizmendi *et al.* (2008) single out Nashville Warbler as the only species to use all four habitats, suggesting that it may be somewhat less sensitive than other species to fragmentation. Lastly, Chipping Sparrow, Warbling Vireo, and MacGillivray's Warbler, all of which breed in the Sierra Nevada, were three of four species in Arizmendi *et al.* (2008) that were only found in mesquite bushes, underscoring the importance of conserving specific plant species or habitats to maintain these wintering bird populations. Besides the condition of winter habitats and migration stopover sites, numerous additional factors can affect year-to-year survival of birds, and population fluctuations at a breeding location may be only partially or not at all a product of conditions at the breeding grounds.

Broad Geographic Picture: Advancing Forests and Why Birds Matter

Countless climate change models predict upslope shifts in the geography of forested habitats, which has the potential to dramatically change distributions and species compositions for plants, birds, and other animals (Lenihan *et al.*, 2008). An upslope shift would consequently decrease or eliminate high elevation habitats, forcing several of the species in my study to adapt, migrate to suitable habitat, or perish. One model estimated decreases of 12-32% in area of Sierran mixed conifer and true fir habitats by 2070, and a corresponding areal increase of at least 23% for low elevation blue oak and foothill pine habitat (PRBO Conservation Science, 2011). Others have estimated major decline in the spatial extent of subalpine forest and extreme vulnerability of this habitat to climate change (Lenihan *et al.*, 2008; Hauptfeld *et al.*, 2014). Additionally, a subalpine shrub zone could develop at the lower elevations of the alpine zone in response to warming temperatures (Lenihan *et al.*, 2008).

Based on climate prediction models and maps in Kueppers *et al.* (2005), Lenihan *et al.* (2008), Tingley *et al.* (2012), and Siegel *et al.* (2014), I have created a theoretical figure for the future of Sierra Nevada elevational distributions for the common bird species in my study (Figure 5.1). Since many of these bird species inhabit multiple habitat types, I have placed them within or between the habitat(s) in which they are most abundant. This figure does not account for potential mismatches of avian-habitat phenologies, or for changes in interspecific competition for food and other resources. It does, however, illustrate where such issues could develop for birds. The figure for 2070 shows a large percentage of the common species inhabiting Sierran mixed conifer forest, and this could create increased competition for resources. The expanded Sierran mixed conifer zone in 2070 of Figure 5.1 counters the predicted reduction of this habitat in some models (PRBO Conservation Science, 2011), but others suggest elevational expansion (Lenihan *et al.*, 2008). The combination of an upslope shift in lodgepole-fir and subalpine species and increased precipitation forecasts, the Sierran mixed conifer forest could maintain or expand its lower elevation limit while also moving upslope.

Tingley *et al.* (2012) detailed the complex, heterogeneous changes in elevation that bird species have exhibited over the past one hundred years in response to precipitation changes and temperature increases in the Sierra Nevada, in which more species have made downslope than upslope expansions. Some species, such as Red-breasted Nuthatch and Cassin's Finch, in fact, expanded their lower elevation limit downslope and maintained a similar upper elevation limit (Tingley *et al.*, 2012). In contrast, Dusky Flycatcher maintained the same lower limit and expanded into higher elevations, while Western Tanager's upper elevation limit shifted downslope,

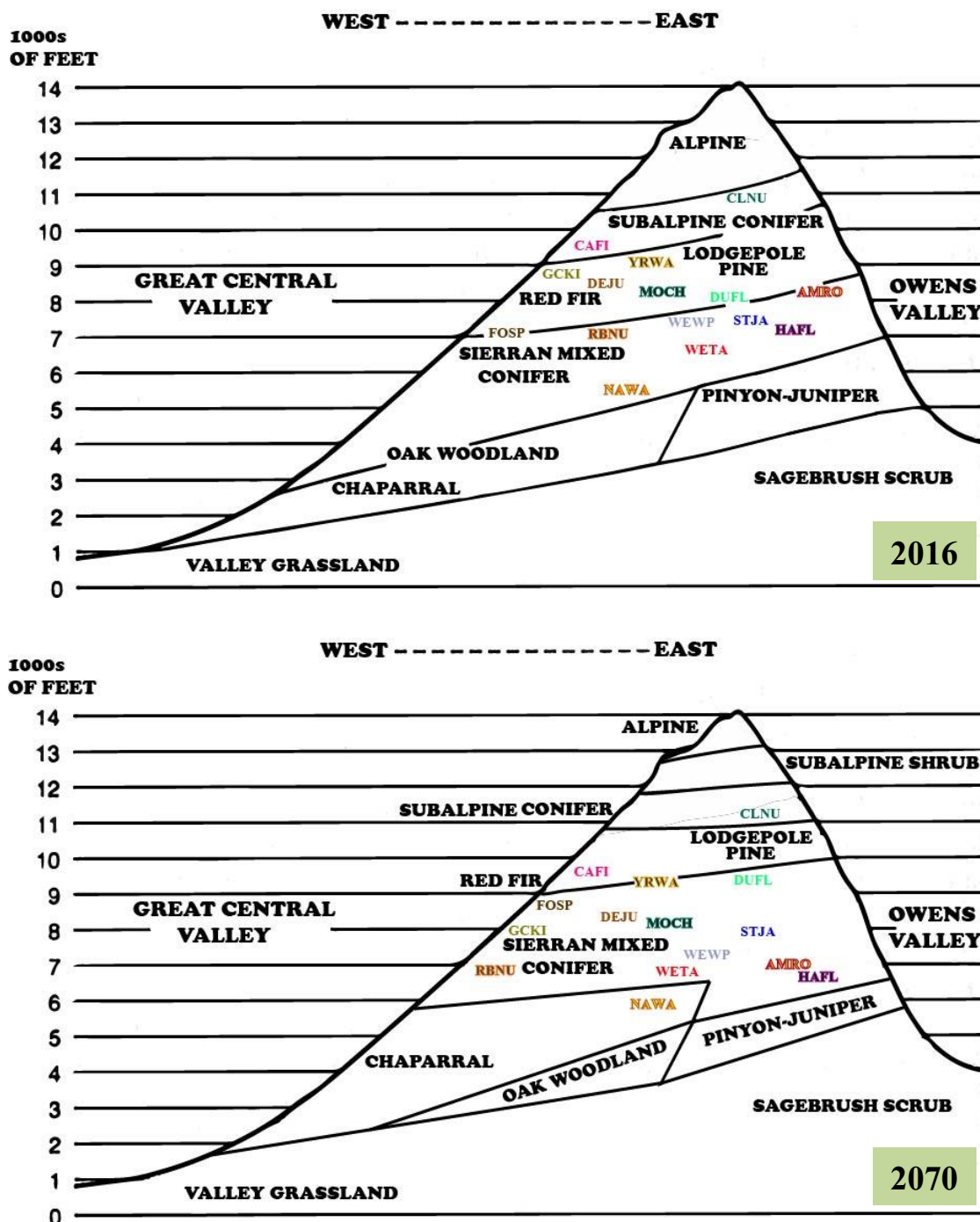


Figure 5.1. Theoretical representation of bird species relationships and habitat associations in the year, 2070. The two graphics compare present elevational distributions of common Sierra Nevada bird species and their breeding habitats to a theoretical depiction of shifts in distribution by 2070. Vegetation zonation is based on Michaelsen (2009) image. See Table 3.3 for species banding code abbreviations.

effectively shrinking its elevational distribution. I interpreted historical movements of the common Sierra Nevada bird species as detailed in Tingley *et al.* (2012) and the predictions of Siegel *et al.* (2014) to plot each species at a theoretical median elevation for its 2070 range in Figure 5.1.

Birds and the human connection. Changes in the distributions, abundance, and richness of birds and other wildlife inevitably influence humans' connections with the environment and the places we share. Countless journal articles, books, films, and other media have documented the declines, hypothetical and real, of birds, and examined why we care and why we should care (Carson, 1962; *The Messenger*, 2015; and others). Birds have played important roles for humans and their ecosystems for perhaps our entire existence. Indeed, "humans share an ageless bond with birds, their song, and their persistent presence in our lives" (*The Messenger*, 2015), with ancient humans interpreting bird song and flight to predict the future, according to the director of the film, *The Messenger*, Su Rynard. For humans, they have functioned as religious symbols and centerpieces of ceremony (Weidensaul, 2007), as well as icons of patriotism (e.g. Bald Eagle and Wild Turkey). Birds also provide joy in our lives, flaunting bright and complex plumages, and serenading us with beautiful, raucous, and peculiar songs and calls.

Birdwatching is an extremely popular activity around the world, with approximately 47 million birdwatchers in the United States alone (Carver, 2013). A study by the U.S. Fish and Wildlife Service estimated that 88% of people who watch birds do so around their homes, largely in the form of feeding birds (Carver, 2013). The colors and behaviors of birds can keep humans transfixed for hours at a time. When asked why they feed birds, people surveyed explained that, in addition to the pleasure they felt and the opportunity to observe the birds closer, they had an attachment, a duty to protect and care for "their" birds (Jones, 2011). This concept of ownership, or perhaps "parenthood," is widespread throughout various aspects of human lives, and it seems logical that it would also be present in our connections to wild animals. Paradoxically, backyard feeding of wild birds may even be an important conservation practice, supported by such projects

as Cornell Lab of Ornithology's Project FeederWatch, though this idea has been challenged, especially in Australia. Those in opposition to feeding birds assert that it teaches birds unnatural behaviors and creates a dependency upon feeders for birds.

As a related note, birdwatchers offer a boost to the economy as well, having spent four billion dollars on bird food, over \$900 million on binoculars and spotting scopes, and about \$970 million on bird feeders, baths, and nest boxes in 2011 (Carver, 2013). Birdwatching has an economic impact farther afield as well, with birdwatching as a major component of ecotourism, and birdwatchers in the United States spending nearly fifteen billion dollars on food, lodging, and transportation for birding in a single year.

As briefly mentioned in Chapter II, birds also provide a tremendous variety of ecosystem services for us, and maintain important balances within our ecosystems. Declines in avian populations would support more frequent outbreaks of disease and rodent and insect infestations, less effective pollination and seed dispersal of plants, and increased numbers of intact carcasses; not to mention potential food shortages for predators and unchecked prey populations. Birds also provide sustenance for cultures that rely upon them as a source of food.

Over time, lacking or minimal seed dispersal could result in loss of entire sections of forest. Studies on the island of Guam, where the brown tree snake has effectively eliminated the presence of birds on the island, have shown that the dearth of seed dispersing birds caused cessation of forest regeneration in areas where forest had been degraded (Caves *et al.*, 2013). Forest dynamics undoubtedly differ between the Sierra Nevada and Guam, but Clark's Nutcracker, for example, fills a similar niche in subalpine forests to the seed-dispersing birds of tropical forests, carrying seeds many kilometers and caching them. These seeds are often forgotten and germinate to become new trees. The combination of upslope forest advancement and declines of seed-dispersing birds would certainly result in the extirpation of some tree species. We cannot wait for the consequences of reduced ecosystem services that birds provide; we must acknowledge our appreciation for these services and take steps to maintain them.

Final Remarks and Future Studies

In conclusion, I asked whether bird abundance and richness fluctuate with spring snowpack in the Sierra Nevada. Early spring snowpack may influence avian abundance and species richness in the Sierra Nevada, but this study does not clarify the dynamics of this possible relationship. None of the common species in this study had abundance to snow depth correlations of greater than ± 0.80 in any year, and only two species, Clark's Nutcracker and Fox Sparrow, had correlation coefficients exceeding ± 0.60 . This study did not record April snow depths at the locations of avian transects, instead relying upon historical data of active snow sensors. Additional studies could improve upon the design of my study by implementing a snow measurement methodology at transects during the year of the avian surveys.

Notable decreases in abundance occurred between 2011 and 2012 for many of the common species, including Dark-eyed Junco and American Robin. The deep snow of 2011 may have indirectly contributed to this decline, reducing the available food for birds, and thus forcing many individuals to forego nesting. Ground-foraging species would have had difficulty accessing seeds and earthbound insects, while the snow may have delayed hatches of aerial insects as well. I also recommend that future studies incorporate a component of monitoring nesting attempts for at least some of the common species, especially ground foragers. Monitoring avian productivity through nesting requires significant time dedication, however, and may come at the cost of fewer transects monitored. Most of the common species also greatly increased in total abundance from 2013 to 2014, two years that are quite similar in terms of early spring snow depths. It would be useful to examine other environmental conditions in the Sierra Nevada during those years to try to explain the nearly study-wide increase in species abundances.

Total avian species richness did not differ among years by any statistical measure that I employed, making it unnecessary to compare richness to snow depths in this study. It is perhaps not surprising that avian richness did not significantly differ among years, as species richness can

easily maintain a fairly stable measurement across years. Only if a considerable influx of species from habitats not included in this study or an emigration by several species out of the study habitats occurred might the overall annual species richness statistically differ.

On the other hand, consideration of the species richness by habitat offered a more specific inspection of annual fluctuations in richness. Predictably, species richness decreased by habitat with increasing elevation. However, all four habitat types showed increasing richness trends over the four years of the study. This may represent the need for species to extend outside of typical habitats to acquire resources, a behavior that can be attributed to drought conditions. Although the Sierra Nevada is currently experiencing approximately average snowfall (winter 2015-2016), many climate change prediction models warn that years of extremely low precipitation will become more common for the Sierra Nevada, and so studies should begin now to better understand how less (or lack of) snow affects various avian behaviors as well.

Finally, examination of species abundances by habitat type revealed a few examples where species may have shifted upslope or downslope to avoid poor conditions or exploit resources. Mountain Chickadee shifted largely from red fir to lodgepole pine and back to red fir between 2011 and 2013, while Yellow-rumped Warbler moved similarly from red fir to lodgepole, and in and out of Sierran mixed conifer. This may simply highlight Mountain Chickadee's ability to succeed in many habitats, especially middle-to-high elevation montane forests.

I suggest that future studies aiming to identify movement between habitat types and potential breeding in different habitats should specifically focus on bird species that are evenly distributed across a range of habitats. Species such as Dark-eyed Junco, Mountain Chickadee, and Yellow-rumped Warbler have sufficient abundances in several habitats that they ought to provide good data about when and hopefully why these habitat shifts occur. On the other hand, the adaptability of these species to many habitats may mask major shifts occurring in more specifically adapted species. Reduced food availability, increased competition, limited nesting opportunities, and other ecological responses to climate change's less frequent heavy snow; increasingly severe

droughts; and warmer temperatures may all factor into the consequences of birds being forced into suboptimal habitats. For some species, especially those with smaller overall distributions, such shifts could have broad, long-lasting repercussions, and we must act now to mitigate the effects, maintain high quality ecosystems, and continue to understand how our actions affect birds and other living organisms around us.

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APPENDICES

Appendix A. SIEN Data Sharing and Confidentiality Agreement

DATA SHARING AGREEMENT

Sharing of Sierra Nevada Network Inventory and Monitoring Data National Park Service, Pacific West Region

This data sharing agreement is entered into by and between the U.S. Department of the Interior, National Park Service (NPS) and TYLER STUART

I. Background and Purpose

The purpose of this agreement is to establish standards for the sharing of information collected by the Sierra Nevada Network (SIEN) Inventory and Monitoring Program as described in Section II.

II. Description of the Data to be Shared

The NPS Inventory and Monitoring Program collects, maintains, and disseminates information about cultural and natural resources. Generally, the NPS makes information available to park employees, state and federal agencies, the scientific community and public. However, dissemination of certain information concerning cultural and/or natural resources could put them at risk. This information is subject to laws with confidentiality requirements, in addition to FOIA requirements, intended to protect park resources from harm, destruction, and theft.

The portion of the NPS Inventory and Monitoring Program data covered by this agreement includes information held by the SIEN. This includes national databases (NPSpecies, NatureBib, NPSTORET) as well as Network and project-specific databases. These data are often dependent on the research and observations of many scientists and institutions, and reflect our current state of knowledge. Many areas have never been thoroughly surveyed, however, and the absence of data in any particular geographic area does not necessarily mean that species or ecological communities of concern are not present.

The data shared under this agreement have undergone varying degrees of quality assurance procedures to ensure their objectivity, utility, and integrity; therefore some portions of the data may be considered preliminary.

III. Confidentiality Agreement

I agree to keep confidential any protected information that I may develop or otherwise acquire as part of my work with the National Park Service. I understand that with regard to protected information, I am an agent of the National Park Service and must not release that information. I also understand that by law I may not share protected information with anyone through any means except as specifically authorized by the National Park Service. I understand that protected information concerns the nature and specific location of endangered, threatened, rare, commercially valuable, mineral, paleontological, or cultural patrimony resources such as threatened or endangered species, rare features, archeological sites, museum collections, caves, fossil sites, geosites, and sacred ceremonial sites. My acquisition or development of any such information does not constitute a release to the public under the Freedom of Information Act (FOIA), and the NPS has not waived any FOIA exemption or other legal protection for this information. Lastly, I understand that protected information must not be inadvertently disclosed through any means including websites, maps, scientific articles, presentation and speeches.

I understand that some information shared under this agreement may not have undergone appropriate quality assurance procedures and may be considered preliminary.

SIEN Data Sharing Agreement, 2009

Page 1 of 2

IV. General Provisions

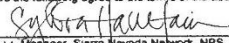
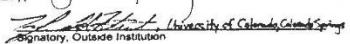
Acknowledgement Policy. Users are requested to acknowledge their use of these data in publications, research proposals, web sites, and other outlets with the following statement: "Data were provided by the National Park Service, Sierra Nevada Network Inventory and Monitoring Program, Bird Monitoring database. Funding for the collection of these data was provided by the National Park Service Inventory and Monitoring Program." Users are also requested to send copies of any publication using these data to:

Term of Agreement. This agreement will remain in effect throughout the duration of the National Park Service, Sierra Nevada Network Inventory and Monitoring Program. Any signatory may withdraw from the agreement upon notification of all other signatories. No further amendment to the agreement would be required for this action.

Amendments. Any signatories may propose changes to this agreement. Any change will be in the form of an amendment and will not take effect until all signatories have agreed to and signed the amendment.

V. Approval Signatures

We the following agree to the terms of the above-described agreement.

PRD  January 24, 2016
Manager, Sierra Nevada Network, NPS Date
 03/02/2015
Signatory, Outside Institution Date

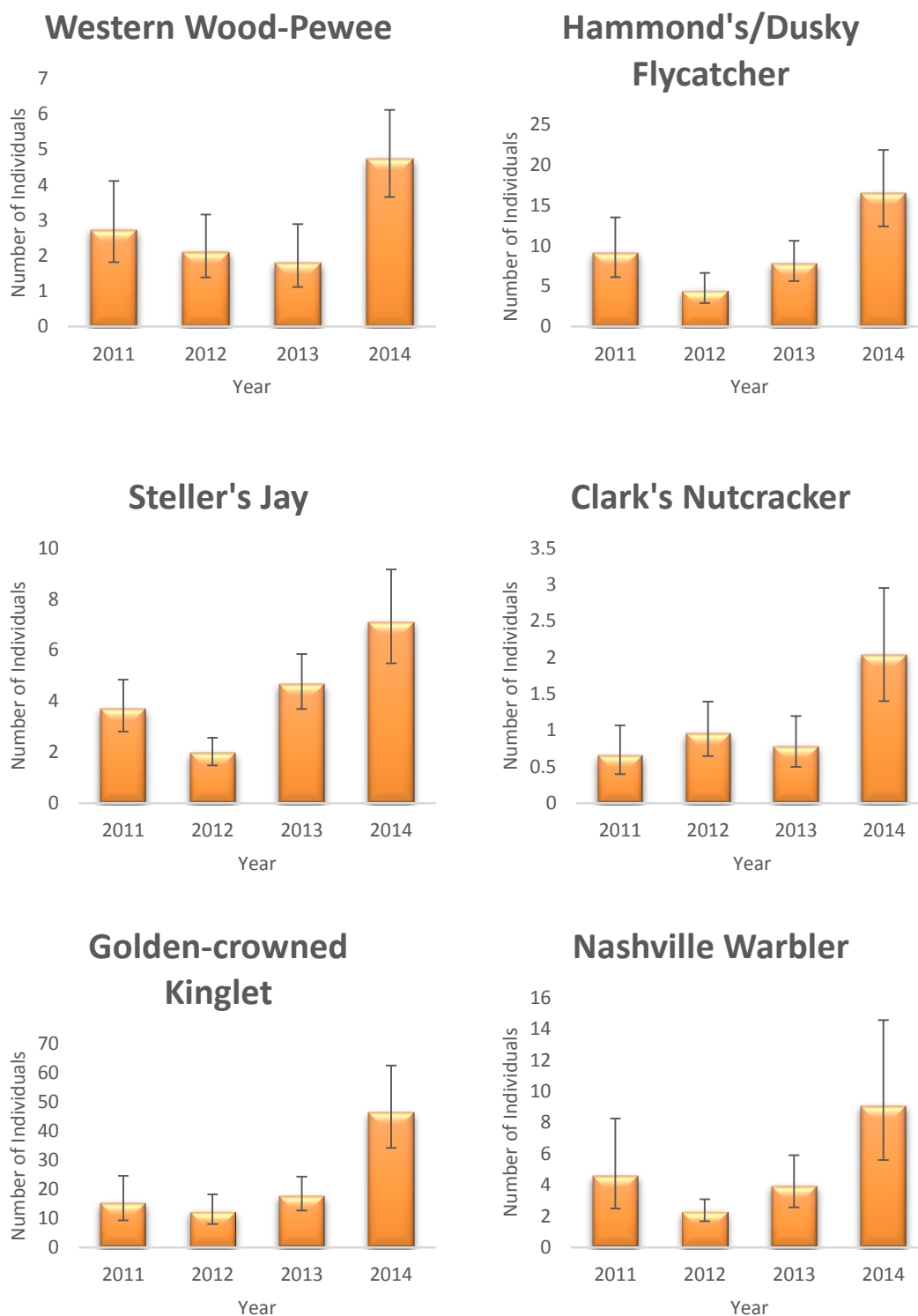
SIEN Data Sharing Agreement, 2009

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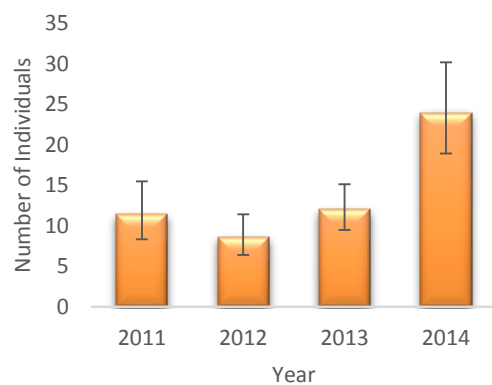
Appendix B. Complete list of species detected during point counts at Yosemite and Sequoia and Kings Canyon, and included in the species richness analysis.

Acorn Woodpecker (<i>Melanerpes formicivorus</i>)	Mourning Dove (<i>Zenaida macroura</i>)
American Robin (<i>Turdus migratorius</i>)	Nashville Warbler (<i>Oreothlypis ruficapilla</i>)
Anna's Hummingbird (<i>Calypte anna</i>)	Northern Flicker (<i>Colaptes auratus</i>)
Band-tailed Pigeon (<i>Patagioenas fasciata</i>)	Olive-sided Flycatcher (<i>Contopus cooperi</i>)
Bewick's Wren (<i>Thryomanes bewickii</i>)	Orange-crowned Warbler (<i>Oreothlypis celata</i>)
Black-headed Grosbeak (<i>Pheucticus melanocephalus</i>)	Pacific Wren (<i>Troglodytes pacificus</i>)
Black-throated Gray Warbler (<i>Setophaga nigrescens</i>)	Pacific-slope Flycatcher (<i>Empidonax difficilis</i>)
Brown Creeper (<i>Certhia americana</i>)	Pileated Woodpecker (<i>Dryocopus pileatus</i>)
Calliope Hummingbird (<i>Selasphorus calliope</i>)	Pine Grosbeak (<i>Pinicola enucleator</i>)
Cassin's Finch (<i>Haemorhous cassinii</i>)	Pine Siskin (<i>Spinus pinus</i>)
Cassin's Vireo (<i>Vireo cassinii</i>)	Purple Finch (<i>Haemorhous purpureus</i>)
Chipping Sparrow (<i>Spizella passerina</i>)	Pygmy Nuthatch (<i>Sitta pygmaea</i>)
Clark's Nutcracker (<i>Nucifraga columbiana</i>)	Red-breasted Nuthatch (<i>Sitta canadensis</i>)
Common Raven (<i>Corvus corax</i>)	Red-breasted Sapsucker (<i>Sphyrapicus ruber</i>)
Dark-eyed Junco (<i>Junco hyemalis</i>)	Red Crossbill (<i>Loxia curvirostra</i>)
Dusky Flycatcher (<i>Empidonax oberholseri</i>)	Ruby-crowned Kinglet (<i>Regulus calendula</i>)
Fox Sparrow (<i>Passerella iliaca</i>)	Song Sparrow (<i>Melospiza melodia</i>)
Golden-crowned Kinglet (<i>Regulus satrapa</i>)	Sooty Grouse (<i>Dendragapus fuliginosus</i>)
Gray-crowned Rosy-Finch (<i>Leucosticte tephrocotis</i>)	Spotted Towhee (<i>Pipilo maculatus</i>)
Green-tailed Towhee (<i>Pipilo chlorurus</i>)	Steller's Jay (<i>Cyanocitta stelleri</i>)
Hairy Woodpecker (<i>Picoides villosus</i>)	Townsend's Solitaire (<i>Myadestes townsendi</i>)
Hammond's Flycatcher (<i>Empidonax hammondii</i>)	Warbling Vireo (<i>Vireo gilvus</i>)
Hermit Thrush (<i>Catharus guttatus</i>)	Western Tanager (<i>Piranga ludoviciana</i>)
Hermit Warbler (<i>Setophaga occidentalis</i>)	Western Wood-Pewee (<i>Contopus sordidulus</i>)
House Wren (<i>Troglodytes aedon</i>)	White-breasted Nuthatch (<i>Sitta carolinensis</i>)
Lazuli Bunting (<i>Passerina amoena</i>)	White-crowned Sparrow (<i>Zonotrichia leucophrys</i>)
Lincoln's Sparrow (<i>Melospiza lincolnii</i>)	White-headed Woodpecker (<i>Picoides albolarvatus</i>)
MacGillivray's Warbler (<i>Geothlypis tolmiei</i>)	Williamson's Sapsucker (<i>Sphyrapicus thyroideus</i>)
Mountain Bluebird (<i>Sialia currucoides</i>)	Wilson's Warbler (<i>Cardellina pusilla</i>)
Mountain Chickadee (<i>Poecile gambeli</i>)	Wrentit (<i>Chamaea fasciata</i>)
Mountain Quail (<i>Oreortyx pictus</i>)	Yellow-rumped Warbler (<i>Setophaga coronata</i>)

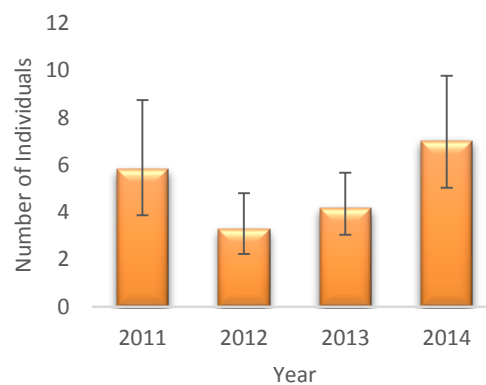
Appendix C. Annual mean abundances (individuals per 12.57 ha count point) of the ten bird species that did not show statistically significant differences in abundance among years in Yosemite, Sequoia, and Kings Canyon National Parks. Associated vertical bars display the 95% confidence intervals for each mean abundance.



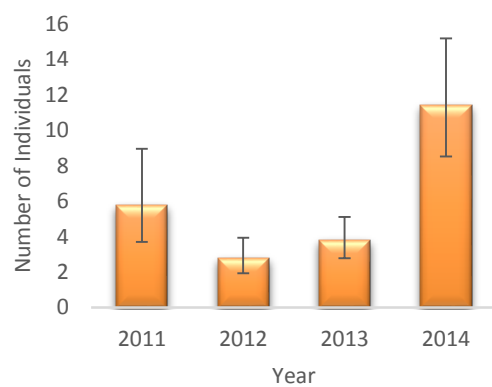
Yellow-rumped Warbler



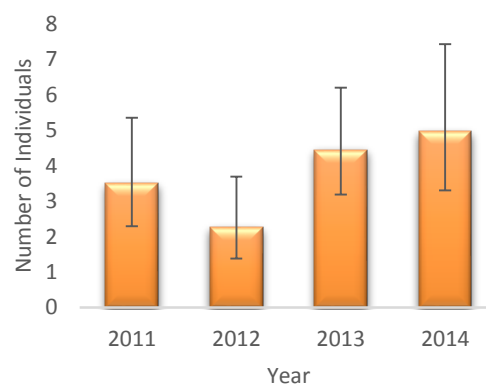
Fox Sparrow



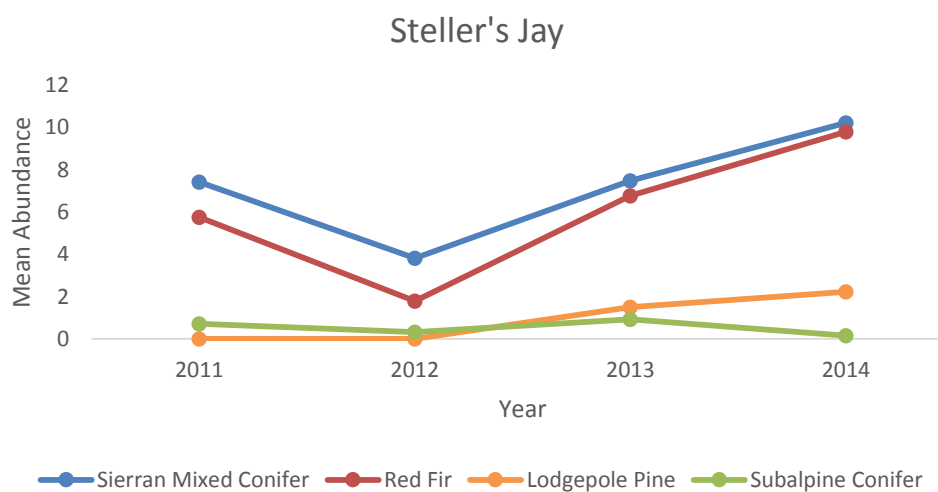
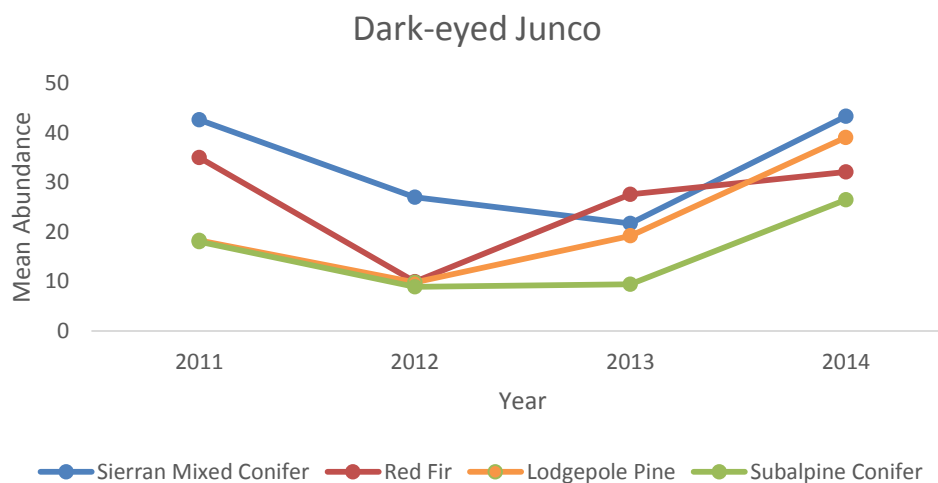
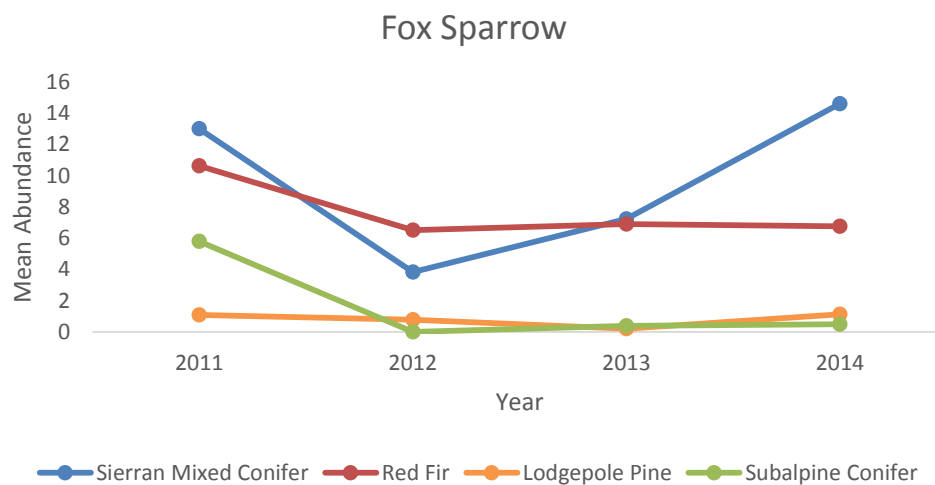
Western Tanager



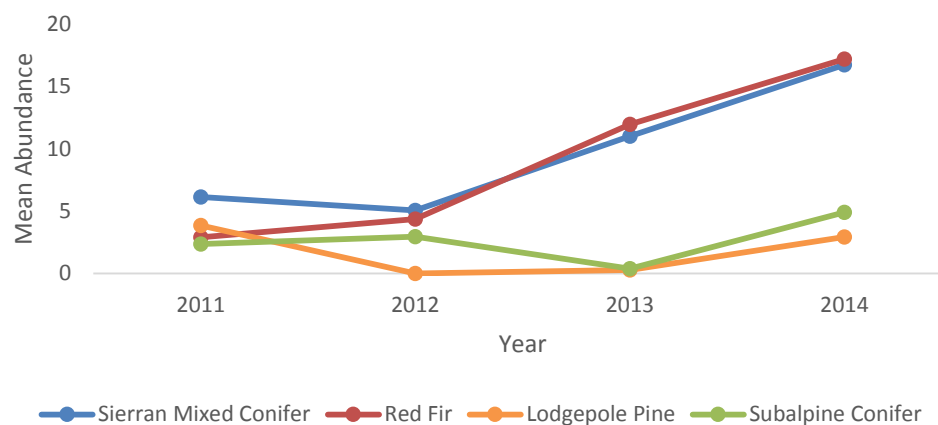
Cassin's Finch



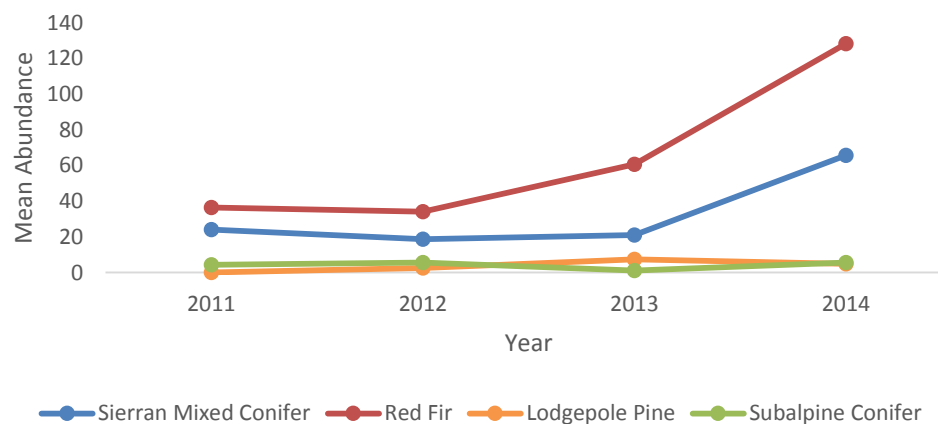
Appendix D. Annual mean abundances (individuals per 12.57 ha count point) in each habitat type of the eleven common bird species not included in Figure 4.3.



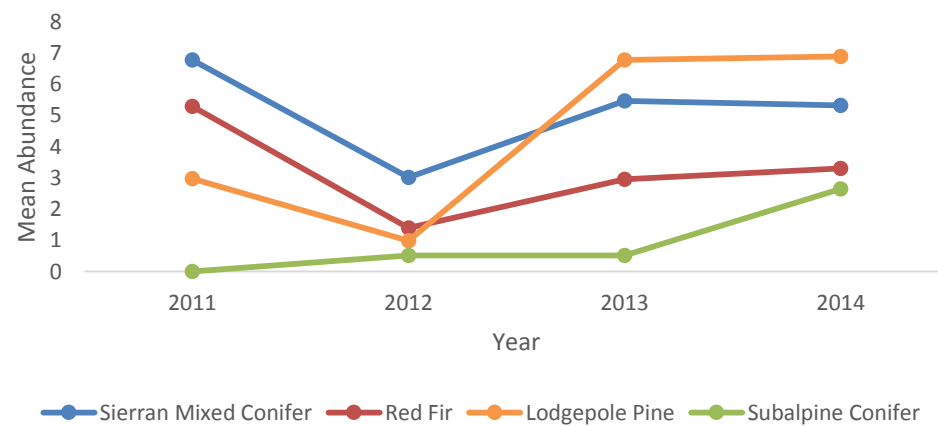
Red-breasted Nuthatch



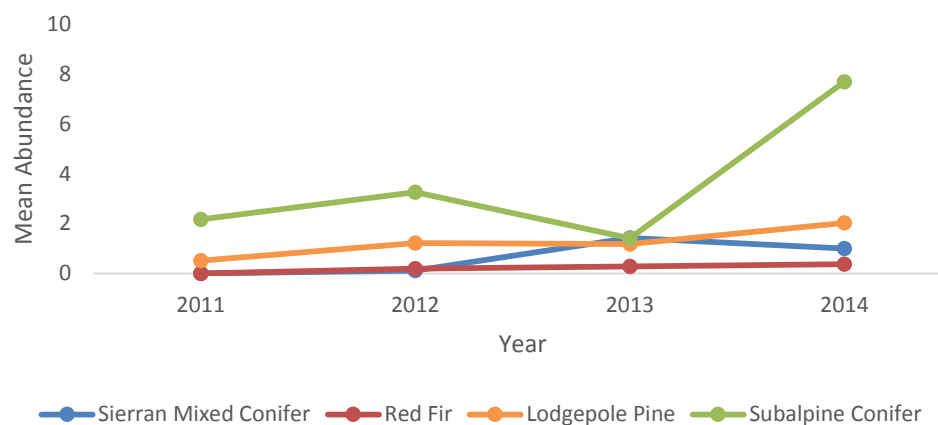
Golden-crowned Kinglet



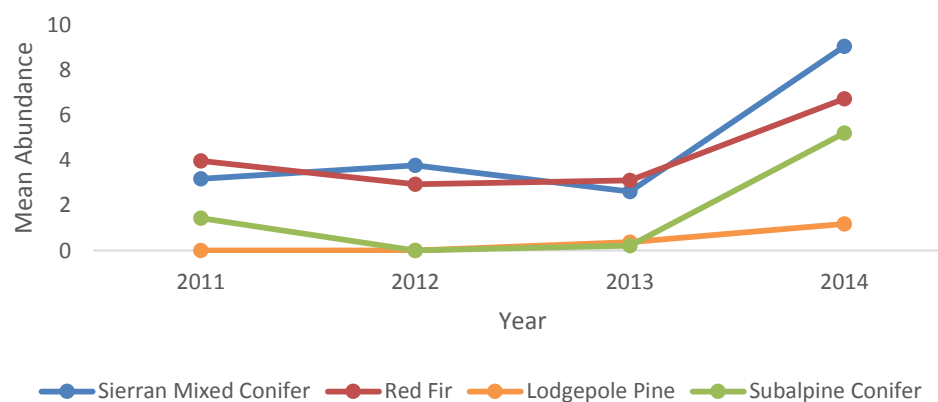
American Robin



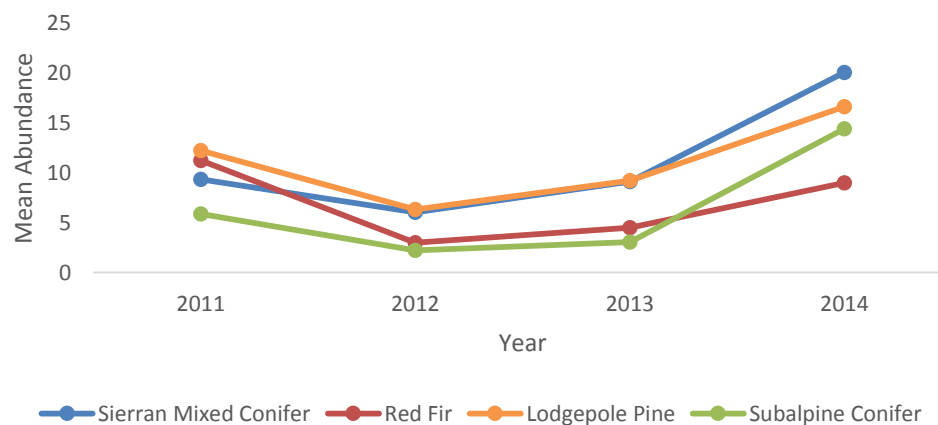
Clark's Nutcracker



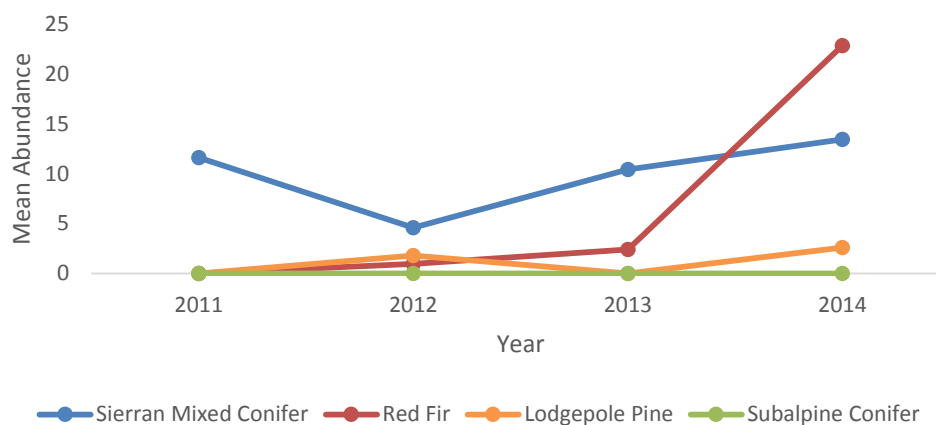
Western Wood-Pewee



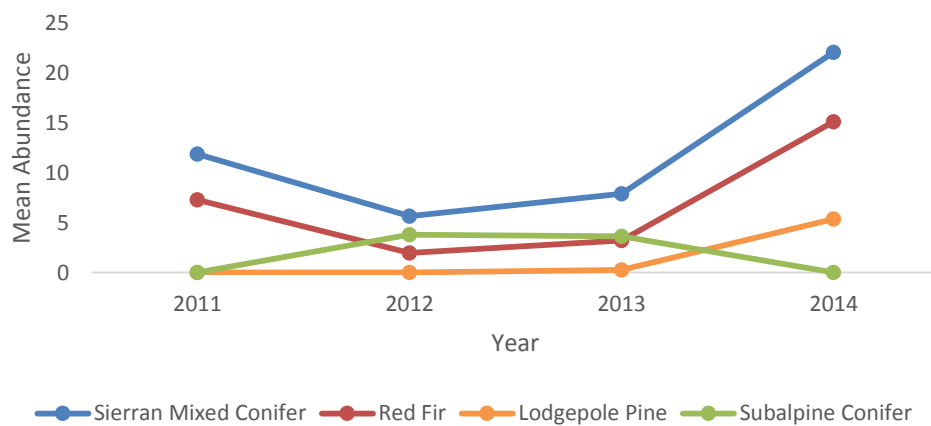
Hammond's and Dusky Flycatchers



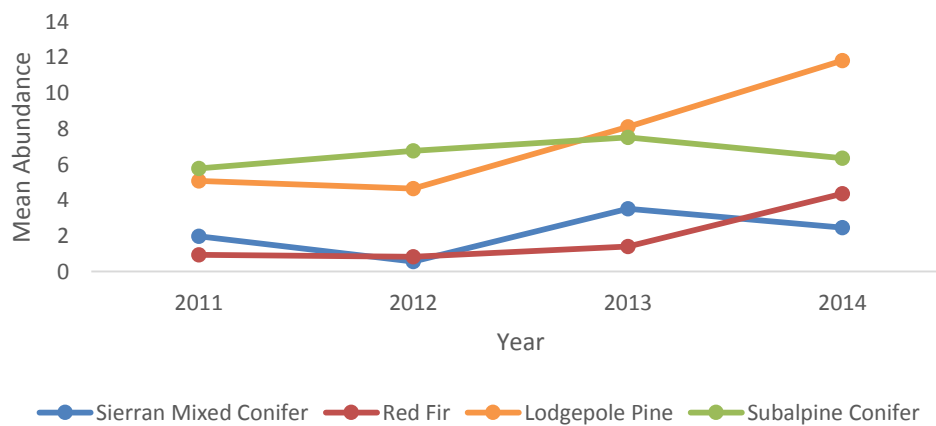
Nashville Warbler



Western Tanager



Cassin's Finch



Appendix E. 95% confidence intervals for change in mean annual abundance by species. The difference between consecutive years in the same habitat type falls within each range below with 95% confidence. “N/A” indicates a sample size of zero in one or both of the compared years, and no confidence interval could be calculated. See Table 4.4 for the mean differences between consecutive years in each habitat type.

Species	2012 minus 2011				2013 minus 2012				2014 minus 2013			
	SMC	RFR	LPN	SCN	SMC	RFR	LPN	SCN	SMC	RFR	LPN	SCN
Western Wood-Pewee	0.3 to 0.8	-2.6 to 0.6	N/A	N/A	-1.4 to -0.9	-0.9 to 1.3	N/A	N/A	6.1 to 6.8	1.8 to 5.5	-1.0 to 2.6	-5.2 to 15.2
Hammond’s/Dusky Flycatcher	-4.4 to -2.2	-11.1 to -5.3	-7.5 to -4.4	-5.0 to -2.3	2.3 to 3.9	-0.3 to 3.3	1.6 to 4.2	0.0 to 1.7	9.8 to 12.1	1.2 to 7.8	5.6 to 9.2	9.6 to 13.2
Steller’s Jay	-3.9 to -3.3	-4.6 to -3.3	N/A	-1.6 to 0.8	-4.2 to -3.1	-5.5 to -4.4	N/A	-1.2 to 0.0	-3.4 to -2.1	-4.1 to -1.9	-1.6 to 0.1	-1.3 to -0.3
Clark’s Nutcracker	N/A	N/A	0.5 to 0.9	0.8 to 1.4	0.2 to 2.5	-0.6 to 0.7	-0.2 to 0.3	-2.1 to -1.6	-2.6 to 1.7	-0.4 to 0.6	0.6 to 1.2	5.8 to 6.7
Mountain Chickadee	-2.8 to -2.0	-17.2 to -13.4	-10.8 to -8.0	1.1 to 3.2	-4.6 to -3.8	1.8 to 4.9	-8.3 to -6.0	-5.6 to -7.6	11.1 to 12.7	11.9 to 17.3	13.2 to 16.9	15.9 to 19.3
Red-breasted Nuthatch	-0.9 to -1.3	0.9 to 2.1	N/A	-5.5 to 6.6	5.6 to 6.3	6.9 to 8.3	N/A	-8.4 to 3.3	5.3 to 6.1	4.0 to 6.4	1.1 to 4.1	3.4 to 5.6
Golden-crowned Kinglet	-7.4 to -3.1	-7.7 to 2.9	N/A	-19.3 to -21.8	0.7 to 3.9	20.9 to 32.4	-10.2 to 19.9	0.5 to 8.4	41.8 to 47.5	56.5 to 78.6	-18.6 to 13.7	-2.4 to 11.2
American Robin	-4.4 to -3.1	-5.4 to -2.4	-2.9 to -1.1	N/A	2.0 to 2.9	-2.5 to -0.6	4.8 to 6.8	-0.6 to 0.6	-0.8 to 0.5	-1.1 to 1.8	-1.1 to 1.4	1.4 to 2.8
Nashville Warbler	-7.9 to -6.1	N/A	N/A	N/A	5.3 to 6.4	0.3 to 2.6	N/A	N/A	2.0 to 4.0	18.3 to 22.5	N/A	N/A
Yellow-rumped Warbler	-5.8 to -3.8	-10.3 to -5.4	8.4 to 11.0	-2.4 to 0.8	4.4 to 5.4	-4.0 to -1.0	-1.3 to 1.1	-0.8 to 1.7	8.9 to 11.0	34.8 to 43.4	10.2 to 13.5	0.6 to 3.5
Dark-eyed Junco	-17.4 to -13.9	-28.2 to -21.9	-10.4 to -6.5	-11.3 to -6.9	-6.5 to -4.2	15.3 to 20.1	8.0 to 10.8	-0.6 to 1.6	20.1 to 23.1	1.1 to 7.9	17.8 to 21.9	15.2 to 19.0
Fox Sparrow	-10.0 to -8.4	-6.4 to -1.9	-1.2 to 0.6	N/A	-3.7 to -3.1	-0.8 to 1.6	-1.0 to -0.1	N/A	6.7 to 8.0	-1.6 to 1.3	-0.1 to 1.9	-1.0 to 0.8
Western Tanager	-7.0 to -5.5	-6.4 to -4.3	N/A	N/A	1.9 to 2.5	0.5 to 2.0	N/A	-10.4 to 10.1	13.5 to 14.8	10.3 to 13.4	-5.4 to 15.6	N/A
Cassin’s Finch	-2.1 to -0.7	-2.8 to 2.6	-1.5 to 0.7	-0.6 to 2.6	2.6 to 3.3	-0.9 to 2.0	2.3 to 4.6	-0.6 to 2.1	-1.6 to 0.5	1.4 to 4.5	2.7 to 4.7	-2.2 to -0.2

