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Habitat use by wintering passerines in fire-managed forests along the U.S. Gulf Coast

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Habitat use by wintering passerines in fire-managed forests along the U.S. Gulf
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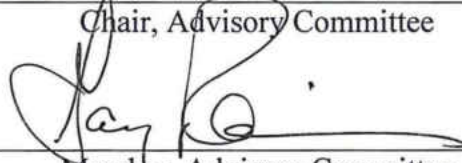
By

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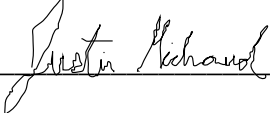
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Habitat use by wintering passerines in fire-managed forests along the U.S. Gulf Coast

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for the degree of
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DEDICATION

I would like to dedicate this to Marylyn and my son Nathaniel who have endured and been by my side throughout this process in one form or another. Also, there has to be some mention of my mom and dad who supported me through out.

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Abstract

Coastal habitats are being impacted by land development, fragmentation, and disturbance related to climate change. The remaining natural areas need to use planned management that may, in some cases, include the use of prescribed fire to maintain habitat quality. Numerous species of passerines, including some with declining populations, use the Gulf Coast as a wintering area, and some depend on habitats managed by fire. To provide information for land managers, I studied the winter bird community at Naval Live Oaks in Gulf Islands National Seashore with two primary objectives: (1) to describe the distribution, abundance, and diversity of the non-breeding winter bird community among the available habitats, and (2) to describe the distribution, abundance, home range size, foraging behavior, and spring-migration departure times of Ruby-crowned Kinglets (*Regulus calendula*) found in the dominant available habitats. These habitats include fire-managed areas such as longleaf pine savanna, oak scrub, and sand pine scrub. During the winters of 2013 and 2014, the bird community was surveyed with repeated fixed-area searches paired with vegetation surveys. Kinglets were banded and regularly re-sighted to record foraging behavior, map home ranges, and monitor spring departure times throughout the winter and the spring migratory period. Bird community surveys revealed differences in the abundance, distributions, and diversity between the two years and within habitats. Species richness was minimally higher in 2014 than in 2013, but overall abundance increased in all habitats. Different fire-management regimes provided varying structures of habitat that provide both high- and low-quality habitat. Habitat segregation among age/sex classes of Ruby-crowned Kinglets was apparent in some habitats. Male kinglets were on average larger than other

kinglets based on structural body size. Foraging attack type varied in regards to the burn treatment, but total attack rate did not. Body size was a strong indicator of the timing of spring departure, which may explain some of the overlap between the departure times of the age/sex classes. My results provide a description of the landbird habitat use in the different available habitats of the Naval Live Oaks area of the Gulf Islands National Seashore, and suggest that fire-based management will help to maintain the current bird community structure. The current mosaic of habitats at Naval Live Oaks provides both successional gradients and ecotonal gradients that provide a wide range of habitats. The diversity of habitats provides areas for a diversity of ecological niches. Species that use predominantly pine trees (Brown-headed Nuthatches, Pine Warblers, and Eastern Bluebirds) and open habitats have the strongest responses to fire management. Kinglets also responded to changes in habitat by demonstrating differing abundances and evidence of segregation among age/sex classes, but these differences were not completely dependent on fire-based management. Suitable high quality habitats were found in both burned and unburned areas, and depended largely on available foraging substrate and the corresponding horizontal structure.

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Chapter I

INTRODUCTION

Fragmentation and reduction of available habitat due to development have made coastal habitats one of the most imperiled of earth's landscapes (McGranahan et al. 2007). Coastal lowlands contain 10% of the world's human population, yet occupy only 2% of the land cover (McGranahan et al. 2007). The threat of climate change and related environmental stressors further endanger these ecosystems (Zhang and Leatherman 2011; Woodrey et al. 2012). Stressors include frequent extreme climatic events, such as irregular weather patterns and strong storm events that can damage and negatively impact plant and animal communities in coastal ecosystems (Diez et al. 2012).

Rising sea-levels and strong storms will continue to degrade coastal habitats through erosion, saltwater inundation, and high winds. Saltwater inundation has been shown to cause more tree mortality than high winds (Bianchette et al. 2009). However, strong winds also damage or kill trees, opening the canopy and changing the vegetation and soil conditions of the forest floor (Aber et al. 2001; Beckage et al. 2006). Trees damaged by storms are more vulnerable to insect infestations and diseases, which can further degrade the health and quality of forest stands (Ayres and Lomberdero 2000; Hanula et al. 2002).

Strong storms such as hurricanes have been shown to affect landbird populations during all parts of the avian annual cycle (Tossas 2006; Dobbs et al. 2009; Brown et al. 2011). Changes include both direct and indirect effects and previous studies have shown that indirect effects may be more important in influencing the community structure (Wiley and Wundererle 1993). Indirect effects of hurricanes on birds include loss of food

sources, foraging niches, roost sites, and canopy cover as well as increased exposure to predators (Wiley and Wundererle 1993). These effects coincide with the alteration of habitat structure that can affect avian communities by increasing the abundance of generalist species that prefer early-successional habitats (Michener et al. 1997; Rittenhouse et al. 2010; Brown et al. 2011).

The Gulf Coast of the United States provides habitat for many species of wintering birds, including both year-round residents and migrants (Duncan and Duncan 2000). Gulf Islands National Seashore (GUIS) was established to protect much of the remaining coastal habitat along the coasts of Florida and Mississippi, and provides refugia for wintering birds. Recent hurricanes, including Ivan (2004) and Dennis (2005), have altered and changed the habitats at GUIS. Hurricane Ivan made landfall on the coast of Alabama and northwestern Florida as a category three hurricane with sustained winds of 190 km/h. Hurricane Ivan's record storm surge eroded coastlines, and altered or destroyed dune systems (Wang et al. 2006). The alteration and loss of these dunes subsequently led to severe damage to all habitats and further degradation of secondary dune systems during Hurricane Dennis (Pries et al. 2008) and Hurricane Katrina in 2005 (Fritz et al. 2007). The storm surges combined with high winds caused extensive damage of the forest canopy, increasing the amount of woody debris (ground fuels) and contributed to a pine beetle (*Dendroctonus frontalis*) outbreak in the interior tracts of forest (M. Nicholas, pers. comm).

In response to hurricane damage from Opal (1995), GUIS implemented a controlled burning regimen following 50 years of fire suppression (Ruth et al. 2007). The main goals of the prescribed fire management at GUIS are to facilitate recovery of forest

structure, maintain a natural ecosystem by promoting longleaf pine (*Pinus palustris*) with a natural fire cycle (Engstrom et al 1984; Monsen et al. 2004; Platt et al. 2006), control invasive plant species, limit the presence of potentially destructive fuels, and control pathogen and insect invasions (Goheen and Hansen 1993; Monsen et al. 2004). The spatial arrangement and frequency of burns among forest patches at GUIs were designed to maintain habitats with a variety of structures and stages of succession, thus providing a diversity of resources for wildlife (Provencher et al. 2002; Monsen et al. 2004; Cox and Jones 2009; Freeman and Kobziar 2011; Johnson et al. 2011).

Many species of landbirds depend on fire to maintain high-quality habitat (Engstrom and Brownlie 2002; Tucker et al. 2004; Cox and Jones 2009). Habitat quality is affected by variation in resource and environmental conditions that in turn can limit the diversity, abundance, and survival of songbirds throughout the year, including during the winter (Johnson 2007). Habitat quality has been recognized as an important factor in the annual survival of songbirds (Silllett and Holmes 2002; Newton 2004). The non-breeding season is an important part of the annual cycle of birds (Studds and Marra 2005; Johnson et al. 2006; Keller and Yahner 2006), and some migratory bird species spend over half the year on the wintering grounds (Marra et al. 1998).

Currently, little is known about the response of winter bird communities and different species of birds to controlled burning in habitats along the Gulf Coast. The winter songbird community of GUIs includes year-round residents and short-distant migrants that may differ in their responses to changes in habitat conditions that follow extreme climatic events and management activities. My objective, therefore, was to identify patterns of habitat use by describing variation in the distribution of the winter

landbird community at GUIIS. Therefore, my general objective, detailed below, was to identify patterns of habitat use by describing variation in the distribution of the winter landbird community at GUIIS.

Study Area

GUIIS protects approximately 8,000 ha of coastal upland habitat consisting of barrier islands, inland forest tracts, and historical sites distributed across two states (Florida and Mississippi) in 12 separate management areas ¹(Figure 1). My focal study area was Naval Live Oaks, a large management unit of GUIIS located in the Florida portion of the park (N 30°21', W 87°08'). Naval Live Oaks encompasses 606 ha and has a history of clear-cuts, cattle grazing, and tree farming. The site has been subdivided into management parcels where prescribed fire has been used as a forest management tool on approximately 663 ha since 1999. Some parcels have been burned multiple times, most recently in 2011. Habitats were classified as mixed canopy (unburned), longleaf pine savanna (burned), oak scrub (burned), unburned sand pine scrub, and burned sand pine scrub. Each of these habitats fits distinctly within one of the two primary forest management strategies used at Naval Live Oaks, both burning and no burning.

The mixed canopy is a maritime forest consisting of large Virginia live oaks (*Quercus virginiana*), a variety of other hardwood species, and red cedars (*Juniperus virginiana*). Hurricanes and other strong storms have created canopy openings that have promoted an understory of dense shrubs and mixed hardwood regeneration. The longleaf pine savanna has an open, fire-dependent understory dominated by grasses, forbs, and palmetto. During the last decade, all longleaf pine habitats of Naval Live Oaks have been burned. The oak scrub habitat is dominated by various species of young scrub oak trees

¹ All figures and tables are presented in appendices as end of thesis.

with small gaps of bare ground and little other vegetation. Unburned sand pine scrub is a community of dense scrub oaks such as sand live oak (*Quercus geminate*), with a canopy of scattered sand pines (*Pinus clausa*) and some larger oaks. Burned sand pine scrub is a mix of oak and pines with an understory that varies based on the intensity of past fires and location relative to the ecotone. Burned sand pine scrub areas with low-intensity fires are generally regenerating forest with a more open understory, whereas high-intensity burned areas have a denser understory and are similar to the oak scrub.

OBJECTIVES AND ORGANIZATION

My study of the winter habitat use and behavior of birds in a fire-managed system had two primary objectives: (1) to determine the distribution, abundance, and diversity of the non-breeding winter bird community at Naval Live Oaks in Gulf Islands National Seashore, and to compare the communities in habitats that differ in how fire has been used as a management tool, and (2) to determine the non-breeding season distribution, home range size, foraging behavior, and spring migration departure timing of Ruby-crowned Kinglets (*Regulus calendula*) and compare these characteristics among habitats of Naval Live Oaks. These two aspects of my study are treated as separate chapters in my thesis.

My study provides baseline data about the current landbird population in the different habitats and management areas of GUIIS. Such information can assist land managers in planning the use of fire-based management or assessing the damage to the winter-bird community in the event of future hurricanes. In addition, my study of Ruby-crowned Kinglets increases our knowledge of a widely distributed wintering migrant whose populations are declining in parts of their breeding range.

Chapter II

INTRODUCTION

Ecosystems of the southeastern United States have evolved in the presence of frequent fire (Van Lear and Harlow 2000, Nowacki and Abrams 2008). As a result, the plant and animal communities of many habitats, including many species of birds, depend on fire to maintain suitable structure (Van Lear and Harlow 2000; Engstrom and Brownlie 2002; Cox and Jones 2009). Decades of fire suppression have changed the vegetative composition of these habitats by increasing the density of shrubs, trees, and woody debris. In addition, coastal ecosystems have been increasingly fragmented due to anthropocentric influences that limit natural ecosystem processes (Mitchell et al 2014). Habitat management has become increasingly important in these areas to facilitate habitat recovery and maintain a natural landscape (Mitchell et al 2014). Management practices that reduce the density of live and dead wood can involve mechanical removal, herbicide application, and prescribed fire (Provencher et al. 2002). Fire-based management is usually the preferred approach because of the ecosystem benefits of post-fire nutrient pulses, natural habitat preservation, and reduction in the amount of ground fuel that otherwise tends to accumulate over time. The frequency, intensity, and size of fires, as well as the composition of the existing community all interact to influence habitat structure during post-fire succession (Freeman and Kobziar 2011).

Landbirds wintering in North America use a variety of habitats depending on their ecological requirements. These requirements include food, habitat patches of appropriate size, and foraging substrates that are dependent on habitat structure. The density of some species varies based on winter habitat structure (Hardy et al. 2013). Non-migratory

resident birds have been found to use habitats differently during the breeding and non-breeding periods (McClure et al. 2013). Additionally, some partial migrants shift to different habitats and have different occupancy patterns (McClure et al. 2013).

Fire management has been shown to influence bird densities, abundance, and community composition. However, the time of year when burning occurred did not affect the winter bird community in the Georgia piedmont region (King et al. 1998). Some species of birds that associate with scrub habitat prefer areas that have not been burned for long periods of time (Breininger and Smith 1992). Breininger and Smith (1992) also observed that no species of birds found in scrub habitat preferred areas that were recently burned, whereas other species in longleaf pine systems, such as the Henslow's Sparrow (*Ammodramus henslowii*), were more abundant in areas burned the previous year (Bechtoldt and Stouffer 2005). The density and abundance of some species of birds fluctuate in the years after a fire. For example, Bachman's Sparrows (*Peucaea aestivalis*) increased in abundance for three years following fire in a longleaf pine savanna, but, after four years, densities and abundance decreased (Tucker et al. 2004).

Variation in availability of resources causes changes in bird communities and how birds use habitats. Provencher et al. (2002) compared hardwood-reduction techniques, including fire, in a longleaf pine ecosystem and found that bird abundance increased with hardwood reduction, but there was no difference in the richness of mixed-species flocks. Winter flocking is believed to be an adaptation to reduce the likelihood of predation and increase the likelihood of locating food (Berner and Grubb 1985). In some cases, food availability has been demonstrated to affect abundance (Johnson and Sherry 2001). For example, abundance of Yellow-rumped Warblers (*Dendroica coronata*) has been found

to be positively associated with the abundance of wax myrtle (*Myrica cerifera*), a plant that requires four to six years of growth to optimize fruit production (Borgmann et al. 2004). Burned and unburned areas have been found to have different community compositions, and this difference can be attributed to the different foraging techniques of different species of birds and differences in resource availability (Kreisel and Stein 1999). Food has been recognized as an important factor affecting over-winter survival of Swamp Sparrows (*Melospiza georgiana*) (Danner et al. 2013).

Among fire-dependent vegetation communities, the optimal fire regime varies, including the frequency and intensity of fire. For example, longleaf pine savannas are best maintained with frequent low-intensity burns (Glitzenstein et al. 1995), whereas some oak-scrub habitat depends on stand-replacing fires that occur at low frequency and high intensity (Van Lear and Harlow 2000). Sand pine-scrub contains both sand pine and various scrub oaks and typically requires a fire frequency of 20–40 years (Breininger and Smith 1992; Van Lear and Harlow 2000). Pine forests are important habitats for some winter migrants and tend to have higher rates of habitat use and occupancy by birds than other habitat types (Woodrey 1998; McClure et al. 2012), although bottomland hardwoods also appear to be important habitat for wintering birds along the Gulf Coast (Dickson 1978).

The abundance, density, and habitat preferences of landbirds are linked to many environmental factors, but habitat structure has been shown to have a strong influence on bird communities. Previous studies have revealed that fire, although important for maintaining structure, it is only one of many such variables, and the effects of fire are dependent on the natural habitat (Breininger and Smith 1992; Lloyd and Slater 2012).

Taken together, these studies provide evidence that habitat management that includes prescribed fire can have important positive effects on bird communities. However, these responses are site- and community-specific and often warrant focused studies to understand each situation.

The habitats at GUIS provide a variety of resources (food and habitat) for passerines throughout the winter. However, management regimes influence their distribution, abundance, and diversity among the major habitat types is not known. This study provides baseline data for land managers to use in management decisions, and contributes to our knowledge of the wintering birds in the southeastern United States.

In this chapter, I describe the effect of habitat and vegetation characteristics on the distribution, abundance, and diversity of the non-breeding winter-bird community at Naval Live Oaks in GUIS. To accomplish this, I conducted repeated surveys of the bird community in the dominant habitats across the winter season. I predicted that each of the habitats in Naval Live Oaks would provide suitable habitat for different specialist species. For example, Brown-headed Nuthatches (*Sitta pusilla*) should prefer the burned longleaf pine habitat and other pine-dominated areas. I also predicted that generalists, like Yellow-rumped Warblers, would be found in a wider range of habitats.

METHODS

Study area

My study took place from late December to mid-February over two winters (2012–2013 and 2013–2014), which included the primary overwintering period for short-distance migrants along the Gulf Coast. My study area was Naval Live Oaks which is part of GUIS (N 30°21', W 87°08'; Figure 1).

Bird Community Surveys

I followed an established protocol developed for the U. S. Park Service as part of the Gulf Coast Networks Monitoring Plan (Twedt 2010). Bird surveys were conducted as fixed-area searches, which increase detection of cryptic species by flushing birds from the vegetation. I surveyed 19 randomly located sites established as part of the monitoring plan (Twedt 2010). I added an additional four points in unburned mixed canopy habitat for a total of 23 sampling sites. Among habitats, there were four sites each in mixed-canopy and longleaf pine savanna, three sites in the burned oak scrub, five sites in unburned sand pine scrub, and seven sites in burned sand pine scrub. Each point was surveyed three times between 30 December and 16 February in both years. Surveys at individual sites were separated by a minimum of one week, and survey order was randomized in each period with some modifications to ensure all surveys at a single site and habitat were not conducted at the same time of day. The fixed-area searches were not conducted during periods of rain or high wind (> 15 km/h). Because there was a noticeable drop in bird activity in the afternoon, all searches were completed before 12:00 to maximize detections.

Fixed-area searches were conducted within a 50-m radius of a randomly located site center. Searches lasted 20 min and began as soon as the site center was located. I recorded all individuals detected by sight and sound to species, including fly-overs. I remained stationary for the first 2 min to detect conspicuous individuals, and then searched the area by walking in a zig-zag circle on a path of least resistance varying the distances from the site center.

Vegetation Surveys

I used a modified relevé method to quantify vegetation structure at each fixed-area site (Wood et al. 2010). A 25-m radius circle was established around the center of each fixed-area search site (the minimum size plot recommended to quantify habitat around search sites; Ralph et al. 1993). Vegetation cover was quantified at four height classes: trees (> 3 m), shrubs (50 cm–3 m), herbaceous (10– < 50 cm), and ground cover (< 10 cm). For each height class, I visually estimated cover based on a five-point class system: 1 = $< 5\%$, 2 = 5–24%, 3 = 25–49%, 4 = 50–75%, and 5 = $> 75\%$. In addition to estimating total ground cover, I used the same five-point system to estimate the extent to which the ground was covered by leaf litter, pine needles, moss/lichens, woody debris (including downed trees), vegetation (grass and herbaceous), and bare ground. Fruit and acorn presence were estimated based on number of fruiting plants/trees or acorn producing trees observed at each site using a four-point scale: 0 = none present, 1 = 1–3 plants, 2 = 4–10 plants, and 3 = ≥ 11 plants.

I measured horizontal structure with a 2-m robel pole in four randomly selected compass bearings at 4 m and 12 m from the site center. At each of the eight locations per site, I calculated total percent obstruction using the 0.5-m increments on the robel pole. At each of the same locations, I also quantified canopy closure with a concave spherical densiometer. Tree species composition was determined by selecting five trees from random compass bearings and a random distance from the site center (1–25 m). I identified each tree to species and measured height with a clinometer, and diameter at breast height (DBH) with a DBH tape. If no tree was within 10 m of the random point, no

data were recorded. I also recorded the species, height, and DBH of the tallest tree in the vegetation sampling area.

Statistical Analysis

For each bird survey site, I calculated the mean abundance of each species by year. I used those values to calculate the following metrics for each habitat and year combination: average abundance (\pm SE), mean species richness, total species richness, Simpson's dominance index, Shannon's diversity index, and the effective number of species (ENS). ENS is the exponential function of Shannon's diversity index, which allows for easier comparisons among different habitats. I classified each species based on migratory behavior (resident or migrant) and report the number and percentage of migratory species and the total abundance of migrants in each habitat. I classified each species into a foraging guild following De Graff et al. (1985), and report the frequency of these classes by habitat. The foraging guilds I used were aerial insectivore, foliage insectivore, granivore, omnivore, piscivore, carnivore, insectivore, and combinations of multiple guilds that included, carnivore/insectivore, granivore/insectivore, insectivore/omnivore, foliage insectivore/frugivore, and frugivore/insectivore.

For each vegetation variable, I calculated the mean and standard error by habitat. I used a Principal Component Analysis (PCA) to reduce multicollinearity and construct a set of composite variables to describe the vegetation among habitats. From the 28 total vegetation variables, I selected the 12 that showed the most variation in mean values among habitats to include in the PCA. From the PCA results, I created a linear combination of factor scores for each site. To associate birds and vegetation, I conducted Spearman Rank correlations using the mean abundance of individuals for each site with

factor scores of the first four axes the vegetation PCA. Species with only a single detection were not used in the correlation analysis. Because vegetation structure changed little between years, I compared the vegetation PCA scores (all vegetation measured in 2013) to two separate years of bird community data. All analyses were conducted using SPSS (SPSS Inc., Armonk, NY).

RESULTS

Bird Community

Overall, I detected 49 species of birds during fixed area searches, including 38 in 2013 (Table 1) and 44 in 2014 (Table 2). American Robins (*Turdus migratorius*) were the most abundant species in both years followed by Yellow-rumped Warblers and Cedar Waxwings (*Bombycilla cedrorum*). Forty-two of 49 species (85.7%) were detected inside 50 m, including 33 in 2013 (Table 1) and 36 in 2014 (Table 2). Blue Jays (*Cyanocitta cristata*) and Yellow-rumped Warblers were the most abundant species detected within 50 m in both years. Species found in all habitats (mixed canopy, longleaf pine savanna, burned oak scrub, unburned sand pine scrub, and burned sand pine scrub) in both years included Mourning Doves (*Zenaida macroura*), Red-bellied Woodpeckers (*Melanerpes carolinus*), Carolina Chickadees (*Poecile carolinensis*), Yellow-rumped Warblers, and Northern Cardinals (*Cardinalis cardinalis*). Most species were more abundant in 2014 than in 2013 (Tables 1 and 2).

In both years, bird communities were dominated by omnivores (n = 22 species) (Tables 3 and 4), with only one granivorous species (Mourning Doves) and one aerial insectivore (Eastern Phoebe [*Sayornis phoebe*]) detected. Foliage insectivores included House Wrens (*Troglodytes aedon*), Carolina Wrens (*Thryothorus ludovicianus*), Blue-

gray Gnatcatchers (*Polioptila caerulea*), and Ruby-crowned Kinglets. I also detected three frugivorous/insectivorous species, including Cedar Waxwings, Orange-crowned Warblers (*Vermivora celata*), and Palm Warblers (*Setophaga palmarum*).

Some species found in multiple habitats had their highest mean abundances shift between years among habitats, including Yellow-rumped Warblers that were most abundant in burned-oak-scrub in 2013 and longleaf pine in 2014, and Gray Catbirds that which were most abundant in mixed canopy in 2013 and in burned-oak-scrub in 2014 (Tables 1 and 2). Total abundance of migrants counted increased between years in all habitats sampled except burned-oak-scrub where abundance decreased from 2013 to 2014 (Tables 3 and 4). The total number of species of migrants increased or remained the same across years in all habitats, with the largest increase in longleaf pine savanna.

In 2013, numbers of species and total abundance were similar across habitats (Table 3). The greatest number of species was detected in burned sand pine scrub habitat (most survey points); longleaf pine savanna had the fewest species. Mean richness and Simpson's diversity were similar among habitats, whereas mixed canopy habitat had the highest Shannon diversity. The proportion of migrant species detected relative to the number of resident species detected was similar among habitats, but burned sand pine scrub had the most species of migrants and longleaf pine had the highest abundance of migrants. In 2014, the habitats again had similar number of species and total abundance, but overall species richness and abundance was higher than in 2013 (Table 4). Average richness per survey site increased only slightly between years, but species were more evenly distributed across multiple habitats in 2014. In 2014 as in 2013, mixed canopy habitat had one of the highest Shannon diversity values, but unburned sand pine scrub

had the highest average richness. As in 2013, the highest number of migrant species was detected in burned sand pine scrub, whereas longleaf pine had the highest percentage of migrant species and the highest abundance.

Vegetation

Vegetation structure differed among habitats and, in some cases I found considerable variation among sites within habitats. In mixed canopy habitat, the estimated percent canopy cover (4.25 ± 0.25) was two times greater than for any other habitat (Table 5). Height of the tallest tree in the mixed canopy habitat was similar to the longleaf pine savanna, but tree diversity was greater and canopy closure was higher. The most abundant trees were Virginia live oak, laurel oak (*Quercus hemisphaerica*), southern magnolia, red cedar, and pignut hickory (*Carya glabra*). Mean canopy height was skewed because of dense patches of regenerating forest from hurricane damage in the mixed canopy. These patches contained high densities of regenerating pignut hickory and laurel oak. The patches of regenerating forest also increased the estimates of horizontal structure due to the increased amount of shrubs. Dominant shrub species were yaupon (*Ilex vomitoria*), myrtle/chapman oak, laurel oak, and saw palmetto (*Serenoa repens*). The ground cover in the mixed canopy consisted mainly of leaf litter, with greater coverage of herbaceous vegetation where the canopy was open.

Longleaf pine savanna was dominated by longleaf pine, with a few small hardwood species such as sand live oak and southern magnolia (Table 5). The average tree height was higher than in any other habitat. Shrub cover in longleaf pine savanna habitat varied among sites, but tended to be low. Shrub species included sand live oak, turkey oak, saw palmetto, and three winter-fruiting species: wax myrtle, gallberry (*Ilex*

coricea), and yaupon. Fruit-producing vines were also present, but not identified.

Longleaf pine savanna had more fruit than any other habitat. The ground cover consisted mainly of pine needles, grasses, and herbaceous vegetation, with some woody debris and bare ground.

Burned-oak-scrub habitat was dominated by scrub oaks, including sand live oak, Myrtle/Chapman oaks, saw palmetto, and laurel oak (Table 5). Myrtle/Chapman oaks were difficult to distinguish in the field so they were combined. Horizontal cover was among the highest of all the habitats, but no fruiting plants were counted. Herbaceous vegetation and grass cover averaged low percentages, and most often were absent altogether.

Unburned sand pine scrub had a high density of shrubs, relatively few trees, and high abundance of woody debris (Table 5). Tree-cover estimates were comparable to the longleaf pine areas and the burned sand pine scrub, but understory was different. High canopy closure estimates were driven by an abundance of taller shrubs. Average tree height in this habitat was relatively low compared to other habitats. Dominant tree species included sand pine and sand live oak. Horizontal cover was higher than in any other habitat. Dominant shrubs included various scrub oaks, sparkleberry (*Vaccinium arboretum*), Florida rosemary (*Ceratiola ericoides*), and few regenerating sand pine. This habitat had little bare ground and high levels of ground fuels, including woody debris, downed trees, pine needles, and dried leaves.

Burned sand pine scrub habitat had intermediate levels for most of the vegetation measurements, in part because some of the sampling locations were dominated by taller oaks and others by mature pine, but also due to variation in fire history (Table 5). This

habitat had the highest diversity of trees and shrubs, shortest trees, the most snags, and the second highest fruit abundance. Sites varied in cover dominance of sand pine, turkey oak, sand live oak, and southern magnolia. At the ground layer, grasses and forbs co-dominated coverage throughout the sites.

Vegetation and Bird Correlations

The PCA included 12 habitat variables that were reduced to four principal components (Table 6). These principal components (PC) explained 83.3% of the total variation in the dataset. PC1 explained 34.2% of the total variation and had a high loading on the tree variables (tree cover, canopy closure, height of tallest tree and its corresponding DBH). PC2 explained 26.9% of the total variation and had high loadings on shrub variables (shrub cover, and total 4-m and 12-m robel pole percentages). PC3 explained 12.9% of the total variation and had high loadings on percent pine needles and number of snags. PC4 explained 8.5% of the total variation and had high loadings on percent bare ground, total ground vegetation, and low shrub coverage, and thus appeared to be associated with the degree of habitat openness.

I found a significant correlation with at least one principle component for nine of 17 bird species in 2013 (Table 7). Presence of Red-bellied Woodpeckers and American Robins was positively correlated with PC1, whereas the abundance of Eastern Towhees (*Pipilo erythrophthalmus*), White-throated Sparrows (*Zonotrichia albicollis*), and Yellow-rumped Warblers was negatively correlated with PC1. PC2 had a positive correlation with Eastern Towhee abundance. The abundance of Cedar Waxwings was negatively correlated with PC2 and positively correlated with PC3. Abundance of Gray Catbirds (*Dumetella carolinensis*) was negatively correlated with PC3 and PC4, whereas

the abundance of Brown-headed Nuthatches and Eastern Bluebirds (*Sialia sialis*) was positively correlated with PC4.

In 2014, the abundance of only four of 26 species of birds showed significant correlations with a principal component (Table 8). As in 2013, Eastern Towhees and White-throated Sparrows showed negative correlations to PC1. Brown-headed Nuthatches were positively correlated to PC3 in 2014, but positively correlated to PC4 in 2013. Yellow-rumped Warblers showed negative correlations to PC2 in 2014 and to PC1 in 2013.

DISCUSSION

GUIS Bird Community Composition

The winter bird community at GUIS differed in abundance, richness, and diversity between years and among habitats. However, in general, there was a high degree of overlap among habitats in species composition, and the richness values were relatively similar. For example, the maximum difference among habitats was 12 species for total richness and three species for average richness, but most pairwise differences between habitats were much lower. There were also similar numbers of species among habitats in most of the foraging guilds. This may be due, in part, to the fact that most species in Naval Live Oaks were classified as omnivorous, including many short-distance migrants and resident songbirds (De Graff et al. 1985). Although a number of species occupied a single habitat (8 in 2013, 9 in 2014), just one or two individuals were counted for most of these species.

I found higher abundance, richness, and diversity in 2014 than in 2013 in almost all habitats. Interestingly, migrants made up a larger proportion of the population in 2013

in all but one habitat. This suggests that the population increase between years was primarily due to increases in numbers of resident species, especially in terms of abundance, although there were also more migratory species in 2014. Such variation suggests the need to sample across multiple years to better account for population changes and annual habitat variation due to weather and successional changes within habitats.

Most species of birds in my study (in terms of richness and abundance) were omnivores. Generalist species typically use multiple habitat types, whereas specialist species tend to associate with one habitat type or a habitat characteristic such as openness (Julliard et al 2006). Omnivory is a typical characteristic of generalist species, and multiple species fitting this criterion were found throughout the study site in varying degrees of abundance.

Generalist species can respond to change and, in some cases, switch diets and alter habitat use within and between years. In my study, abundance of Yellow-rumped Warblers was negatively correlated with tree variables in 2013 and with shrub variables in 2014. Yellow-rumped Warblers were the most abundant species in both years and found in all habitats. The high abundance of this species in longleaf pine habitat in 2014 can be partially attributed to finding large flocks in a patch of wax myrtle at one survey site, a fruit that Yellow-rumped Warblers depend on in winter (Borgmann et al. 2004). This is clearly an example of a generalist species responding to resource conditions.

By definition, specialists are relatively sensitive to environmental change, and show population changes when habitat is altered (Rittenhouse et al. 2010; Brown et al. 2011). I found just a few species in each habitat with some degree of habitat

specialization, and all of them were observed in low abundance. For instance, Eastern Bluebirds were only found in open longleaf pine habitats and Gray Catbirds appeared to favor dense scrub.

Two other important limitations of my study may have influenced the results. First, sampling sites (i.e., fixed-area search locations for birds) were spatially nested within habitat patches, and there was a limited number of patches at Naval Live Oaks. The patch level replication in my study was limited, and so I was unable to statistically account for this pseudoreplication. Spatial factors such as proximity to the ocean and the highway or the arrangement of habitat patches may affect the bird community at Naval Live Oaks, although these spatial factors also likely affect the vegetation communities and even wildfire burn patterns, so it would be challenging to unravel such effects even if additional patches were available for sampling. Another limitation is that I did not record participation in flocks or analytically account for flocking tendencies of birds in terms of habitat use or guild placement. Foraging behavior is known to be correlated with sociality, specifically with frugivory being associated with gregariousness (Greenberg and Salweski 2005). Species such as Cedar Waxwings fall fit this association which were found in flocks and seen on few occasions in high numbers. Although I made some observations of frugivory and flocking, such patterns were not systematically integrated with data collection and analysis.

Habitat Relations

Even though my study design was descriptive, there were still patterns among habitats, in particular among burned and unburned habitats, in terms of species composition, abundance, and guilds. Burned-oak-scrub and unburned-sand pine-scrub

were the two densest scrub habitats in this study, yet the habitats have been maintained using different management strategies (i.e., high intensity burned vs. unburned). I found a high degree of overlap in bird species composition between these habitats, but Eastern Towhees, White-throated Sparrows, and Gray Catbirds were more abundant in the burned-oak-scrub, as would be expected given their similar habitat use (Greenberg et al. 1995, Falls and Kopachena 2010, Smith et al. 2011). Whereas unburned-sand pine-scrub had areas of broken canopy and scattered cover provided by sand pine and sand live oak that provided an additional forest layer facilitating additional forage opportunities. This habitat was dominated by species typically associated with the canopy such as Carolina Chickadees, Tufted Titmice (*Baeolophus bicolor*), Blue Jays, and most woodpecker species (Dickson and Noble 1978).

Unburned-sand pine-scrub had higher richness and diversity overall as well as a greater abundance and richness of winter migrants than the burned oak scrub, likely due to increased vertical structure. The composition of the vertical structure appears to be important depending on the density and composition on each forest layer. Burned-sand-pine-scrub was relatively heterogeneous in structure compared to the other habitats and had the highest overall bird species richness and the greatest variety in foraging guilds, likely because of the diverse vertical structure. However, some sites had very low richness and abundance of birds which is an indication of variation in burn regime and corresponding microhabitat.

Carolina Wrens were the only species to show higher abundance in unburned compared to burned scrub areas in both years. These wrens are permanent residents and maintain winter home ranges in and around their breeding habitat (Haggerty and Morton

2014). My results are similar to those of Berringer and Smith (1992), with more Carolina Wrens detected in habitats that were infrequently or not burned. Lynch (1991) suggested that, in the absence of fire, wrens recover from hurricane disturbance in just two years. The area-wide increase of Carolina Wrens between years in my study may indicate a successful breeding season and the subsequent dispersal of juveniles into previously unused habitat or the available habitats were becoming more suitable for this species given the lack of current fire management and natural habitat successional changes.

Longleaf pine was the most open landscape and is maintained by frequent fire regime, which is critical to management of this habitat (Mitchell et al. 2006). A frequent fire regime inhibits scrub growth and promotes grasses and palmettos. Unlike other species, Eastern Bluebirds were found exclusively in fire-managed longleaf pine in both years. In 2013, their abundance was positively associated with greater ground cover, as is found in the longleaf pine habitat. In an oak-woodland habitat, Hardy et al. (2013) found fewer Western Bluebirds (*Sialia mexicana*) in areas with dense vegetation than in open landscapes. Other species favored longleaf pine areas as well, such as White-crowned Sparrows (*Zonotrichia leucophrys*) and Swamp Sparrows (*Melospiza georgiana*). Swamp Sparrows have previously been described as associating with longleaf pine during winter (Johnson et al. 2009), although they probably tend to associate with wet locations within this habitat (Mowbray 1997). This habitat may be preferred by some species of sparrows, e.g., Henslow's Sparrows in Louisiana (Palasz et al. 2010), because of the open understory and greater seed production (Buckner and Landers 1979). Fire and mechanical manipulations can reduce shrub cover and lead to use by bird species that prefer prairie or savanna-like conditions (Breininger and Schmalzer 1980, Fitzgerald et al. 1992).

Although some species of wintering sparrows such as Henslow's Sparrows have been found to respond positively to the higher seed densities in recently burned longleaf pine savanna, fires that are too frequent can have a negative impact so Bechtoldt and Stouffer (2005) recommended biannual burning in the growing season. However, Thatcher et al. (2006) suggested that managers should burn a large percentage of savannas each year. My results suggest that it is not as simple as one management, and a dynamic mosaic of varying successful managements is ideal for high avian diversity and the subsequent benefits for individual species.

Fire in the southeastern United States promotes species of pine, such as sand pine and longleaf pine, by controlling hardwood encroachment (Van Lear and Harlow 2000; Parker et al 2001; Provencher 2002). Pine habitats are important for winter bird communities in ecosystems in the southeastern United States, in part because of their dominance across much of the landscape, but also because these habitats are critical for some specialists (Jackson 1994, Thatcher et al. 2006, McClure et al. 2012), and provide high-quality habitat for many generalists (Pearson 1993, King et al. 1998, Conner et al. 2002). My results also support this trend because the highest richness of migrants and greatest total abundance were in pine-dominated burned areas. Several resident species, as described in this and previous studies, were also associated with pine habitat, including Pine Warblers (*Dendroica pinus*) and Brown-headed Nuthatches (Morse 1967). Conifers tend to have higher mortality due to storm damage than hardwood species (Duryea et al 2007). As a result, management that promotes the recruitment of replacement pine stands, such as with prescribed fire, may positively impact winter bird populations in the southeastern United States.

Burned and unburned areas in coniferous forest have been shown to differ in community composition depending on the time since fire (Kreisel and Stein 1999). Kreisel and Stein (1999) found that trunk and branch-foraging species (i.e., Hairy Woodpecker, Downy Woodpecker, and Mountain Chickadee) were more likely to be found in areas associated with stand replacement fires than in unburned areas and suggested that changes in habitat structure and food resources were responsible for this pattern. Wintering Henslow's and Bachman's sparrows in the Gulf coastal plain also depend on habitats created by specific burn regimes (Palasz et al. 2010, Brooks and Stouffer 2011).

The unburned mixed canopy habitat in my study had the largest trees, and the highest canopy cover of all the habitats. However, the forest floor was highly variable because of small patch openings caused by recent hurricanes. Strong storms tend to alter coastal habitats to increase scrub density and thus favor bird species that predominantly use dense scrub, edge or open field habitats (Lynch 1991). The hurricane-caused patch openings in this habitat have developed into thick scrub, creating a structurally heterogeneous habitat with a variety of foraging substrates for multiple species of birds. During both years of my study, the winter bird community of unburned mixed canopy habitat consistently had the highest average richness and diversity. A study of the breeding birds in Florida's coastal scrub showed that bird richness and diversity is directly related to increase vertical and horizontal structure (Greenberg et al. 1995). As with the other habitats in my study, most species in the unburned-mixed-canopy habitat were generalists also found in the other habitats. Live oak habitats are important across the Gulf Coast for their role in storm abatement, but also as stopover habitat for migrating

birds (Somershoe 2004, Moore 2005). My results and those reported in previous studies (e.g., Breininger and Schmalzer 1990) suggest that these habitats are also important for winter bird communities.

Tree cover, canopy closure, and tree size had the greatest influence on the winter bird communities in my study because the presence of six species of birds was significantly correlated with PC1 (representing tree variables), more species than with other components. Dickson and Noble (1978) described bird habitat use in relation to the vertical structure in a hardwood forest in the Gulf Coastal Plain, and showed many species used specific height ranges. The science of ecology has a long history of describing such associations (MacArthur 1958), and this study follows by showing such patterns in habitats and seasons where they haven't been carefully studied before. Correlations with tree structure varied depending on the species. For example, Red-bellied Woodpeckers showed a positive correlated with tree structure, whereas the correlation for White-throated Sparrows and Eastern Towhees was negative. Canopy closure influences the overall horizontal structure and appears to affect the bird community as well. The habitats at my site varied in the horizontal structure, thus affecting the openness between habitats. For example, the burned longleaf pine habitats were open with practically no shrub layer, whereas the unburned habitats were composed of dense scrub. The tree structure influences the understory vegetation at my site from the direct effects of storm damage (i.e., wind throw gaps and mortality from salt water pulses) (Durya et al. 2007, Bianchette et al. 2009), and is further manipulated by controlled burning. Floristic components of habitats have been shown to influence

songbird density both negatively and positively depending on the composition of the understory and openness of the habitat (Hardy et al. 2013).

As expected, fruit availability appeared to be associated with the abundance of frugivorous and fruit-eating omnivore species. For instance, in 2014, Yellow-rumped Warblers were most abundant in longleaf pine habitat, which also had the highest density of fruiting shrubs, such as yaupon. The only habitat which did not have an increase in the number of migrants from 2013 to 2014 was burned-oak-scrub where no fruit production was observed. Fruit is important for many species of wintering birds in the southeastern United States, including in pine and oak habitats (Skeate 1987, Strong et al. 2005), and some have argued that the importance of fruit for wintering wildlife in the region has been underappreciated (McCarty et al. 2002). The relationships observed in my study suggest fruit is also important at GUIs. An ice storm that occurred at the end of January, in the middle of the second round of fixed area searches which likely limited arthropod availability for several days, and led to birds moving to other habitats where fruit was more abundant. Fruit resources have been shown to be a particularly important resource during freezing events (Kwit et al. 2004, Weinkam et al. 2016).

Conclusions

Compared to prescribed fire, wildfires often create greater heterogeneity, and managers are increasingly using techniques that enhance landscape heterogeneity of prescribed fire to better mimic natural burn patterns, including strip head, tree spot, and point source firing (Nowacki and Abrams 2008, Ryan et al. 2013). The heterogeneity in vegetation structure in my study area may have been related to burning practices or landscape features (e.g., topography). The patterns of bird species associations with

individual habitats observed in my study provide valuable insight into the role of fire for management along the Gulf Coast. This study identifies aspects of habitat suitability for winter birds, some of which prefer areas maintained by fire-based management. Fire management at GUIIS maintains several types of habitat (i.e., scrub, sand pine, and longleaf pine), which have a variety of vertical and horizontal structural differences, along with variation in the plant community composition. My results suggest that the composition of the wintering bird community is affected by burn treatment, but these effects are variable and complex, and likely related to successional status in burned and unburned areas. Leaving some areas unburned, or at least burned on a longer time interval (>3 years) adds to overall habitat heterogeneity, and the bird community. However, with long intervals between fires, such habitats are at risk of being more susceptible to wildfire, or become difficult to manage with controlled fire or by mechanical means.

The overall habitat mosaic at Naval Live Oaks influences the winter avian community, in particular, since there is some degree of specialization. Fire-based management appears to be an effective method for maintaining the habitat heterogeneity. A carefully constructed fire regime that creates a mosaic of patches with multiple stages of succession appears to promote increased diversity of the winter bird community at GUIIS, as it does elsewhere in the southeastern U.S. (Engstrom and Brownlie 2002, Harper et al. 2016) This approach could serve as a model for management of multiple habitats in one area. Fire-based management needs to continue for these areas to maintain suitable habitat for a wide variety of species.

Chapter III

INTRODUCTION

Events during the non-breeding period can have direct effects on the survival of birds as well as indirect effects that carry over into the migratory and breeding periods (Norris et al. 2004; Boone et al. 2010). Much of the research on how events during the non-breeding period might affect birds has focused on the quality of non-breeding habitats. Habitat quality is generally based on the quality and abundance of available resources (Johnson 2007). Recent studies have shown that the quality of habitats used by over-wintering songbirds can influence survival, body condition, spring migration departure times, and subsequent breeding success (Saino 2004; Studds and Marra 2005; Johnson et al. 2006; Keller and Yahner 2006; Studds and Marra 2011).

Most studies of the possible effects of variation in quality of non-breeding habitat on birds have focused on Neotropical and Nearctic migrants (Saino 2004; Studds and Marra 2005; Johnson et al. 2006; Keller and Yahner 2006; Studds and Marra 2011). Only recently have investigators begun to examine the effects of habitat quality on short-distance migrants wintering in temperate zones (Kreisel and Stein 1999; Brown et al. 2002; Diggs et al. 2011; Danner 2013). In many areas, particularly coastal areas, habitat quality can be influenced by weather (e.g., severe storms) and management practices. For example, fire-based management can change the structure and composition of natural habitats. These changes depend largely on the type of habitat and the fire regime, which together influence post-fire succession (Freeman and Kobziar 2011).

Post-fire succession can influence the composition of bird communities. For example, controlled burning can change habitat structure at different strata, including the

canopy, which can affect food availability and the foraging behavior of some insectivorous songbirds (Hartung and Brawn 2005). Specifically, in closed-canopy habitat (i.e., unburned), songbirds showed more selectivity for foraging on particular tree species than in open-canopy (i.e., burned) areas, and aerial insectivores were more likely to use different attack behaviors between open- and closed-canopy habitats than foliage-gleaning species (Hartung and Brawn 2005). For over-wintering birds, food availability can strongly influence body condition (Brown and Sherry 2006) and the likelihood of survival (Danner et al. 2013). Food availability has been linked to differences in occupancy, with larger-bodied conspecifics defending territories with more abundant food sources (Diggs et al 2011). In other cases, the abundance of a species can be linked to food availability (Johnson and Sherry 2001; Borgmann et al. 2004). For example, Yellow-rumped Warblers were found to be more abundant in areas with higher concentrations of wax myrtle fruit (Borgmann et al. 2004).

Food availability can also influence space-use patterns and social behavior (Brown and Long 2007; Diggs et al 2011). Winter birds have an array of social systems, including being solitary, territorial, floating in loose-boundary home ranges, joining mixed-species flocks, or some combination of these behaviors. Trade-offs in access to food and predation risk exists for in each of these strategies. In situations of patchy food availability, flocking or floating among territories may be advantageous (Greenberg and Salewski 2005; Brown and Sherry 2008). In contrast, predictable or at least evenly distributed resource conditions are correlated with ideal-despotic territorial systems in which territory-holding birds benefit (Sherry and Holmes 1996). For example, in American Redstarts, dominant territory holders have improved body condition and earlier

spring migration departure times than subordinate conspecifics (Marra and Holmes 2001; Studds and Marra 2005).

Given the importance of habitat quality to wintering songbirds, individuals might be expected to compete for higher-quality habitats if the species exhibits territorial behaviors. Sherry and Holmes (1996) suggested that multiple lines of evidence can be used to infer intraspecific competition for preferred habitats in the non-breeding season, including variation among habitats in bird densities, territorial defense, non-random distributions, changes in body condition, and differences in the length of time of residency. Density is sometimes a false indicator of habitat quality (Van Horne 1983), and so multiple indirect measures of habitat quality provide stronger evidence that birds are responding to variation in habitat quality. In this study, I used indirect measures of habitat quality to measure the response of Ruby-crowned Kinglets to habitat variation created by alternative habitat management approaches.

Ruby-crowned Kinglets (hereafter kinglets) are widely distributed temperate zone migrants found throughout the southern United States during the winter (Swanson et al. 2008). In Florida, kinglets arrive on wintering grounds in October and depart for breeding areas in March and April (Duncan and Duncan 2000). Populations of Ruby-crowned Kinglets are declining in the eastern part of the United States for unknown reasons (Swanson et al. 2008). Laurenzi et al. (1982) suggested that kinglet populations may be winter-limited in the Colorado River valley, yet little is known about population-related processes for this species in the winter, such as relative habitat quality, age/sex-based dominance, survival, and migration timing (Swanson et al. 2008, Somershoe et al. 2009).

Morse (1970) reported that kinglets are territorial during the winter. In contrast, Gram (1998) found that kinglets in dry pine oak forest were predominantly solitary, but also joined mixed species flocks and behaved as a nuclear flock species. In California, Humple et al. (2001) documented sex-biased habitat use, with females more abundant in scrub habitats than mixed evergreen habitats, possibly due to male displacement of females from higher quality habitats. In a comparison of kinglet body condition in different burn treatments during the non-breeding season, mass was lower, but not significantly lower, in riparian areas that had been burned (Samuels et al. 2005). The results of a previous study also suggest that male kinglets migrate earlier than females in the spring (Swanson et al. 1999). Given the few studies of kinglets during the non-breeding period plus the conflicting results of previous studies of their social behavior, additional studies of their winter behavior and habitat use are needed to better understand how habitat type and quality might affect their social behavior, foraging behavior, and time spent in wintering areas.

In this chapter, I report on the non-breeding season home range sizes, foraging behavior, and spring migration departure times of kinglets found in different habitats at Naval Live Oaks. The responses of kinglets to variation in food availability and habitat structure during the non-breeding season have not been well studied. Specifically, I tested for patterns of sex- and age-based patterns of site occupancy. I also compared habitats for differences in body size, site fidelity, and foraging behaviors. I expected that differences in these behaviors among habitats would, in turn, affect home range size and shape, migration departure timing, and apparent survival. Consistent and predictable patterns of these behavior and population characteristics would suggest differences in quality among

habitats, and suggest the importance of winter events in the life cycle of kinglets.

Although I did not directly measure resource availability or changes in body condition, differences in the behaviors and habitat use of kinglets provide an indirect measure of habitat suitability that can be used to inform land-management practices.

METHODS

Study Area

This study took place from late December to April over two winters (2012-2013 and 2013-2014), which included the primary overwintering period for short-distance migrants along the Gulf Coast and part of the spring Neotropical migration period. My focal study area was Naval Live Oaks (N.30°21', W.87°08') located in Gulf Islands National Seashore (Figure 2).

Kinglet Surveys

Kinglets were surveyed with fixed-area searches at 23 sites that included all of the focal habitats. See Chapter 2 for details, but briefly kinglets were surveyed three times throughout the winter period.

Bird Banding

During January, I target-netted kinglets using mist-nets (6 m x 2.5 m x 30 mm mesh) with a recorded playback of kinglet songs and calls. Each captured bird was marked with a unique combination of two color bands and an USGS aluminum band. Standard morphometric data were recorded, including tarsus length (± 0.01 mm) with digital calipers, unflattened wing chord, and tail length (± 0.5 mm) with a wing ruler, and body mass (± 0.1 g) on a digital scale. I assigned a fat score based on visual inspection of the furcular region following the methods of Holmes et al. (1989), with 0 no fat, 1 (trace),

2 (fat filling bottom of furculum), 3 (fat filling furculum), and 4 (fat mounded and spreading over breast muscle). A breast muscle score was also assigned based on the following scale: 1 (concave), 2 (flat), and 3 (convex). Half scores of fat and muscle were given to birds that fell between these categories. Each bird was aged and sexed according to Pyle (1987). All banding occurred in January and February, so age classes reflect the change in age class naming that occurs on January 1. Second-year (SY) birds were not sexed because males lack a red crest and cannot be accurately sexed. After-second-year (ASY) birds with a red crest were called males and ASY birds that did not have a red crest were considered females. After release, banded birds were re-sighted at least twice per week until 1 March when a regimented schedule was implemented to record foraging behaviors, map home ranges, and monitor for spring departure time.

Foraging Behavior

During re-sighting visits (see more detail concerning visits below), I recorded information about foraging behavior on a portable digital hand-held voice recorder. A narrative of movements and foraging behaviors was recorded for as long as a banded bird was in sight. In many cases, multiple short narratives were recorded on each re-sight visit. The recording was stopped when excessive chatter was believed to be directed at me. The foraging observations were based on the protocol used by Lovette and Holmes (1995). A bird was deemed foraging when it was actively moving and/or searching for prey and not foraging when it was vocalizing, exhibiting aggressive behavior, preening, or otherwise stationary and obviously not searching for prey. Foraging movements were classified as hops, short flights (≤ 1 m), and long flights (> 1 m). Foraging attacks were recorded using five common foraging maneuvers described by Remsen and Robinson

(1990): 1) gleans, 2) sally hovers, 3) sally strikes, 4) sallies, and 5) flutter chases.

Preliminary field observations suggested that these were the most frequent maneuvers used by kinglets in the non-breeding season in my study area.

Home Range Mapping

Following the initial banding of each bird, I returned to the area where each bird was banded and attempted to re-sight the individual and identify it based on its unique color band combination at least once per week during January and February. During return visits, the area around the original banding site was searched for a maximum of one hour and either out to a maximum of 200 m radius from where the bird had been captured or until the bird was re-sighted. After re-sighting and identifying a bird, I followed it for up to one hour to delineate its home range and record behaviors, and if I believed my presence was affecting the behavior of the bird, I left the area. During these observations, I recorded GPS locations (± 5 m accuracy) of focal birds at 5-min intervals beginning with the location where the bird was first re-sighted and continuing for up to an hour. I continued to locate each bird and map their locations until they left the area in the spring. In addition to the regular home range visits, opportunistic re-sightings were also recorded when observing neighboring individuals and moving through areas where banded kinglets were present.

Departure Time

Color-banded birds were monitored to estimate their dates of departure from the wintering grounds following protocols described by Studds and Marra (2011). After initially mapping the home ranges in January or February, I returned to the home range of each bird every three days from 1 March to 20 April, at which time the migratory period

for this species along the Gulf Coast is almost over (Duncan and Duncan 2000). For each bird, I searched the areas of known occupancy for a maximum of one hour. If a bird was not re-sighted during that time, I returned at least twice before the following regularly scheduled visit. If the bird was not re-sighted in the next scheduled visit, I returned once more before the next scheduled visit and used playback of kinglet songs and calls to verify its presence or absence. Evidence that a bird had departed included observation of new, previously undetected birds in the home range, the absence of singing by a male, which generally increased in frequency through March, and no response to playback in the core home range area.

Statistical Analysis

The structural complexity of the unburned sand pine scrub made re-sighting birds difficult, so I was able to map few birds ($n = 5$) in this habitat. Thus, I combined habitats into burned and unburned treatments. For birds followed in both winters, I randomly selected one winter of data for each bird for foraging, home range, and departure analyses.

All statistical analyses were conducted in R (R Core Team 2014). I used separate ANOVAs to test for differences in morphological measurements between age/sex classes. The response variable was age/sex classes and the explanatory variables included mass, muscle score, fat score, wing chord, tarsus length, and tail length. Significant ANOVA results were followed with post-hoc Tukey's tests. In addition, a structural body size index was calculated as the first axis of a PCA using the wing chord, tarsus length, and tail length measurements. The difference between the age/sex classes and body size was

tested with a one-way ANOVA and the significant results were followed up with post-hoc Tukey's tests.

Apparent survival and detection probability were estimated with program MARK using a Cormack-Jolly-Seber model (White and Burnham 1999). There were a total of five time intervals over the two years, including two one-month intervals during the winter periods of each year (January-February and February-March), corresponding to mid-winter and late-winter periods when all kinglets were present at the site, and a 10-month interval for the pre-migratory, migratory, breeding, and early winter (March-following January) periods.

For analysis of kinglet foraging behavior, I transcribed recordings and, for each observation period, determined the proportion of time spent foraging, foraging rate, and frequency of each attack type. Foraging observations of less than 10 sec or otherwise abnormal observations (e.g., pecking at bands) were omitted from analysis. Proportion of time spent foraging was calculated by dividing the time spent foraging by the total time a kinglet was observed. The total numbers of movements were divided by the time spent foraging to estimate foraging rate. Attack rates were estimated for each type of individual attack and for total combined attacks by taking the number of attacks and dividing by the total time spent foraging. For each kinglet, data for all observations were averaged to generate one value for each variable. Possible differences between years in proportion of time spent foraging, foraging rate, attack rate, glean rate, and sally hover rate were examined with a t-test using Welch's correction for unequal variance. I compared the proportion of time spent foraging and foraging rate among age/sex classes and burn treatments using MANOVA with a Pillai trace statistic. The two most common attack

type rates (glean rate and sally hover rate) and the total attack rate were compared among age/sex classes and burn treatments using MANOVA with a Pillai trace statistic, with each rate included as a separate response variable.

Estimates of home range size were calculated for all birds with more than 12 re-sight locations combined across at least five different days. For each bird, I used ArcMap to construct a kernel density home range estimate, which describes the probability distribution of an individual occurring in different areas. I calculated the grid cell size and search radius criteria of the kernel analysis separately for each bird based on the size, shape, separation of points, and habitat. Because the 95% distribution area included unused areas such as roads and ocean for several kinglets, I used a more conservative 90% distribution area as the best estimate of full home range areas. The 50% distribution area was used to describe the core area. I calculated total area (ha) and perimeter (m) of the full and core home ranges in ArcMap. To quantify home range shape, I calculated an edge-to-interior ratio by dividing the perimeter by the area of the total and core home ranges for each bird. A lower ratio indicates a simpler shape with more core area and less edge. I used a MANOVA with a Pillai trace statistic to test for differences in the area of the core and full home ranges among age/sex classes and burn treatments.

I estimated a re-sighting probability using a Cormack-Jolly-Seber model run in program MARK using birds that were banded and mapped across nine sampling occasions that corresponded to the 3-day interval visits through the late-winter and early-spring of each year (White and Burnham 1999). The re-sighting probability was used as a measure of sampling bias for the estimates of departure timing.

To identify factors that might influence spring migration departure time, I developed a generalized linear model (GLMER) using Program R, package lme4 (Bates et al. 2014). Departure date, measured as the number of days since 1 March, was used as the response variable, and the explanatory variables included habitat type, burn treatment, age/sex class, structural body size, 90% home range area, and total attack rate. Year was included as a random effect. I used a Poisson distribution because departure time was based on counts of days (Bolker et al. 2009).

I constructed a global model which included structural body size, 90% home range area, total attack rate, habitat, and burn treatment, along with the interaction of body size with each variable, and year as a random effect. Program R, Package MuMIn was used to test all combinations possible combinations of models and select the best ones based on AIC_c (Barton 2014). The best models were then averaged based on models with delta AIC_c value > 3 . For each age/sex class, I estimated predicted departure dates of three body size classes based on the interquartile range values of PC1: Q1 = small body size class, Q2 = medium body size class, and Q3 = large body size class.

RESULTS

Demographics

Across both years, kinglets were present in all habitats, and detected in 21 of the 23 fixed area search sites. I counted 41 kinglets in 2013 and 48 in 2014. Among habitats, unburned sand pine scrub had the highest mean abundance of kinglets, and the burned sand pine scrub habitats had the lowest mean abundance in 2013 (Table 1). In 2014, burned sand pine scrub had the highest and burned oak scrub had the lowest abundance of kinglets (Table 2).

I banded 54 kinglets over the two winters (2013: $n = 23$, 2014: $n = 31$) (Figure 2). On average, ASY males were larger than both ASY females and SY kinglets, but some differences were not significant (Table 9, Figures 3 and 4). ASY males had significantly greater mass than SY kinglets (Tukey's HSD, $P = 0.002$), but did not differ from ASY females (Tukey's HSD, $P = 0.064$). Differences among ASY males, ASY females, and SY birds in muscle and fat scores were not significant. ASY males had longer tails (Tukey's HSD, $P < 0.001$), longer wing chords (Tukey's HSD, $P < 0.001$), and longer tarsi (Tukey's HSD, $P < 0.001$) than SY kinglets. ASY males had significantly longer tails (Tukey's HSD, $P = 0.003$) and wing chords (Tukey's HSD, $P < 0.001$) than ASY females.

The PCA for structural body size produced one component (PC1) which explained 61% of the variation. Among the three variables included in the PCA, tarsus measurement had a moderate correlation with PC1 (0.51), whereas wing chord (0.91) and tail length (0.86) were highly correlated with PC1. Based on the combined index of structural body size (i.e. scores from PC1), ASY males were larger than both ASY females (Tukey's HSD, $P < 0.001$) and SY kinglets (Tukey's HSD, $P < 0.001$), but ASY females and SY kinglets did not differ (Tukey's HSD, $P = 0.21$) (Figure 4).

Overall, apparent survival and detection probability varied over time. The mid-winter periods (January–February) of both years had the lowest apparent survival, but, during the 2013 mid-winter period, the error overlapped considerably with the non-winter period (i.e., pre-migratory/migratory/breeding; Table 10). Five birds banded in the winter of 2013 returned to the same home ranges in the winter of 2014, including four in burned habitats.

Of 54 birds banded, 35 were followed during at least one winter and three during both winters, and an accurate home range could not be defined. In burned habitats, I monitored 11 ASY male, nine ASY female, and four SY kinglets. In unburned habitats, I monitored seven ASY male, four ASY female, and three SY kinglets. I did not relocate kinglets in all habitats. No kinglets were monitored throughout the winter in burned oak scrub habitat because the dense vegetation made it difficult to re-sight banded birds.

Foraging Behavior

I collected foraging behavior for 15 birds for a total of 192 observations greater than 10 seconds in 2013, and for 20 birds for 272 observations greater than 10 seconds in 2014. The number of observations per bird ranged from 1 to 29 (mean = 12.9 ± 1.3). Hops were the most frequent movement observed, followed by short flights; long flights were observed infrequently ($X^2 = 3,750.2$, $df = 2$, $P < 0.001$; Figure 5). Gleans were the most frequent foraging attack observed, and aerial sallies the least frequent ($X^2 = 1,066.2$, $df = 4$, $P < 0.001$; Figure 6).

Observations of 32 birds were used in my analysis of foraging behavior. I found no difference between years in the proportion of time spent foraging ($T_{21.85} = 1.2$, $P = 0.24$), gleans per minute ($T_{27.97} = 1.2$, $P = 0.62$), sally hovers per minute ($T_{26.86} = 0.70$, $P = 0.49$), or attacks per minute ($T_{23.53} = -0.2$, $P = 0.83$; Table 11). However, total foraging rate was higher in 2014 than in 2013 ($T_{26.79} = -3.7$, $P = 0.001$). I found no difference among age/sex classes and burn treatments in either the proportion of time foraging (MANOVA: Pillai = 0.11, $F_{2,26} = 1.6$, $P = 0.22$) or foraging rate (MANOVA: Pillai = 0.05, $F_{2,26} = 0.6$, $P = 0.53$). For sally hover rates, I found a significant interaction between age/sex classes and burn treatment (MANOVA: Pillai = 0.37, $F_{2,25} = 7.6$, $P = 0.003$). All

the age/sex classes showed the same trend by exhibiting more sally hovers in burned habitats, but ASY females had a much lower sally hover rate in unburned habitats than the other age/sex classes. I found no differences among age/sex classes and burn treatments in either glean rate (MANOVA: Pillai = 0.01, $F_{2,25} = 0.1$, $P = 0.95$) or total attack rate (MANOVA: Pillai = 0.06, $F_{2,25} = 0.9$, $P = 0.41$).

Home range

I mapped 35 home ranges over the two winters of my study ($n = 13$ in 2013, and $n = 22$ in 2014). Many kinglets appeared to occupy the same home range areas throughout the winter period. Inside of these home ranges there was considerable overlap between unbanded birds and banded birds (Figure 7), and, in some cases, the overlap was nearly complete (Figure 8). I captured 77% of the kinglets in their core (50% kernel) home ranges ($n = 11$ of 13 in 2013, $n = 16$ of 22 in 2014). I found no difference among burn treatments or age/sex classes either in the size of 50% home range areas (MANOVA: Pillai = 0.01, $F_{2,29} = 0.1$, $P = 0.90$) or 90% home range areas (MANOVA: Pillai = 0.13, $F_{2,29} = 2.2$, $P = 0.13$; Figure 9).

Departure time

I determined detection probability and departure dates for 35 kinglets ($n = 13$ in 2013, and $n = 22$ in 2014). The detection probability, reported here as a measure of sampling bias for the spring departure period, differed between the age/sex classes: ASY males had the highest detection due to their singing behavior ($p = 0.86 \pm 0.04$ SE; 95% CI: 0.77–0.93), followed by ASY females ($p = 0.66 \pm 0.07$ SE; 95% CI: 0.52–0.78), and SY bird had the lowest ($p = 0.56 \pm 0.07$ SE; 95% CI: 0.41–0.69) There was considerable overlap between ASY females and SY birds, due to similarities in behavior.

All variables included in the global model influenced departure time in all the top weighted models that had values of $\Delta AIC_c < 4$. The most important variable was body size which was present in all the highest weighted models ($\Delta AIC_c < 4$) (Table 12). Model averaged coefficients showed that body size was the only significant predictor of departure timing (Table 13), indicating that larger body size was related to earlier departure time. Habitat, home range size, and attack rate had equal weights and very similar ΔAIC_c values indicating these variables do contribute to departure time. However, given the lack of significance for these other variables in the model averaged coefficients, I generated a model that included only body size as a fixed effect and year as a random effect.

Small and large body birds for each age/sex class differed from 3 (ASY female) to 5 days (SY) in day of departure. The medium body-sized birds' departure times overlapped with both large and small-bodied birds in all groups. There was considerable overlap in departure time between ASY females and SY kinglets (Figure 10).

DISCUSSION

Several lines of evidence from my study suggest that fire-based management can create both high- and low-quality habitat for kinglets depending on the fire regime. My results indicate that fire-based management has implications for kinglet abundance, foraging behavior, and sex-based occupancy. The effect of controlled burning on habitat structure was most apparent at the understory and midstory strata (height of 1–5 m), where the density of vegetation differed strongly between unburned and burned habitats (Table 5). The midstory strata includes both shrubs and trees which have been shown to

be essential to kinglet habitat, and the density of the horizontal structure which is directly manipulated with fire management.

Habitat Associations

Kinglet abundance was lowest in habitats with either frequent low-intensity burns (i.e., longleaf pine) or high-intensity stand-replacing burns (i.e., burned oak scrub). Abundance was highest in non-burned habitats, and in the infrequently burned pine oak scrub. Kalinowski and Johnson (2010) found wintering kinglets in suburban areas in northern California were positively associated with increasing shrub and tree cover (vegetation height > 3 m). My results support Kalinowski and Johnsons (2010) since I found increased abundance of kinglets in habitats with established shrub and tree layers, as in the pine oak scrub and mixed forest habitats. I found decreased abundance in long leaf pine habitat, which lacks a shrub layer and the oak scrub that does not have trees. These structural differences are direct effects of fire management and its impacts on the micro- and macro-habitat.

Both micro- and macro-habitat characteristics have been shown to influence habitat occupancy of kinglets in Alabama (McClure et al 2012). Important microhabitat variables include canopy cover and basal area, and the macro-habitat variable that was significant was natural pine forest (McClure et al. 2012). McClure et al. (2012) found kinglets were positively associated with more canopy cover, and negatively associated with increasing basal area, suggesting that kinglets prefer young stands of pine with high density of small trees (i.e. approximately 10 years old). I found similar results, kinglets were more abundant in intermittently disturbed areas with dense shrub layers. In my site the stands were older than 10 years, however, disturbance at the site from strong storms

and intermittent fire creates areas of regeneration that create habitats that function similar to these younger stands of pine.

Kinglets that occupied the burned oak scrub habitat were predominantly SY and ASY females, suggesting that there is habitat segregation between age sex classes. Habitat segregation among age and sex classes has been described as an outcome of dominance relationships for model systems in winter ecology of passerines, such as American Redstarts (*Setophaga ruticilla*) (Marra and Holmes 2001). The dominance-mediated social system observed in redstarts has been experimentally linked to variation in habitat quality (Marra and Holmes 2001). Such patterns have also been observed in kinglets in coastal California, where mist-net captures resulted in a ratio two females per male (Humble et al. 2001). My results suggest a similar response, with ASY males apparently forcing ASY females and SY kinglets to use habitats of presumably lower quality.

Home Range

The different sex/age classes of kinglets all occupy home ranges throughout the mid- and late-winter periods until spring migration. Although I found no significant patterns of variation in home range sizes among burn treatments or age/sex classes, ASY male kinglets tended to have larger home ranges in burned habitat than in unburned habitat, whereas ASY female and SY kinglets tended to have larger home ranges in unburned habitat. If this is indeed a real pattern, it may be explained by differences in structure between burned and unburned habitats. It is possible that unburned areas have increased foraging substrate so that a smaller home range is sufficient, but ASY males could occupy higher quality sites with more food availability and thus need even smaller

home ranges. Diggs et al (2010) found that larger birds, predominantly males, occupy home ranges with higher arthropod availability, and regularly exhibit territorial behaviors. It is also possible that in burned areas body size is important for maintaining a larger area that would provide access to more resource opportunities, and would help explain the larger home ranges of ASY males compared to ASY females and SY kinglets. Bechtoldt and Stouffer (2005) found no difference in home range size of Henslow's Sparrows among burn treatments despite differences in vegetation structure. However, abundance of Henslow's Sparrows differed among patches based on the number of years since last burn, which suggests a numerical response.

There was some evidence of winter floaters in the population of kinglets at Naval Live Oaks. Namely, some banded birds were not re-sighted again, and in one case I was able to follow a kinglet over a number of days, and map a home range that was much larger than all other kinglets in this study. Other evidence of floaters includes occasional observations of unbanded kinglets that were otherwise not regularly observed in an area. It may be that these observations are related to the occasional behavior of kinglets joining mixed species flocks. Winter floating behavior has been described in other species, and in some cases it may be an advantageous strategy for acquiring resources although it is more commonly associated with exclusion from resources by dominants (Brown and Long 2007; Brown and Sherry 2008).

Foraging Behavior

Foraging rates differed between years, with kinglets in the winter of 2014 having higher rates. Faster foraging rates have been suggested to indicate food scarcity (Zach and Falls 1976), although it may also occur in situations of increased prey availability

(Morrison et al. 1990), especially when physiological demands are high, such as prior to migration (Moore and Simons 1992, Heise and Moore 2003). My findings may be related to the reduced food availability following the ice storm in 2014, which may have had lasting effects throughout the season. However, the total rate of attack was similar between years and was also similar to other insectivorous species in exhibiting 3 to 4 attacks per min (Lyons 2005).

Kinglets had higher sally-hover rates in burned than unburned areas. This foraging difference is most likely based on the different structures of the habitats. The dense structure of unburned areas may be more suitable for hops and gleanings, whereas the relatively open structure in burned areas may facilitate aerial maneuvers, such as the sally hover. More hops contribute to more search time for prey whereas more flights contribute to less time searching (Lyons 2005). This may also partly explain why kinglets have smaller home ranges in areas with increased horizontal structure and larger home ranges in more open habitats, like the one managed by fire. The results of studies conducted during the breeding season suggest that the foraging behavior of aerial insectivorous songbirds differs between open and closed canopy areas, but foliage gleaning songbirds did not show such differences (Hartung and Brawn 2005). My results contradict these findings, because they show that a gleaning insectivore can change foraging strategies in response to the resource conditions. American Redstarts show distinct differences in foraging behaviors between breeding and winter seasons, including differences in rates and substrates (Lovette and Holmes 1995). Further work is needed to understand how kinglets change behavior among life cycle stages, but difference would not be surprising given they show differences between habitats within a season.

Departure Time

Structural body size was the only significant indicator of spring departure time, with larger birds in each respective age/sex class departing earlier. This suggests that body size mediated dominance is an important factor for all age/sex classes. Because departure time is likely related to the accumulation of events over the winter season, dominance is likely related to habitat use. In wintering Hermit Thrushes, structurally larger birds occupied areas with higher arthropod abundance, suggesting that larger birds defend and maintain higher quality habitat (Diggs et al 2010).

There was evidence that habitat, home range size, and attack rate contribute to departure time. All of these variables relate to habitat quality, An increase sample size would help define these patterns further. In my study, males tended to depart prior to females. Areas vacated by departed males were then occupied by neighboring or immigrating kinglets throughout the departure season. American Redstarts have been shown to upgrade habitat in response to vacancies in higher-quality habitat (Studds and Marra 2005). In my study, the continual occupancy of vacated areas was likely attributable to a combination of habitat upgrade, home range expansion, and the arrival of migrant kinglets from other areas.

The kinglets that returned to burned areas and were followed both years had later departure times in 2014 than in 2013. This may be due to environmental factors such as the ice storm in February 2014. Environmental factors such as rainfall have been shown to influence spring migration timing, specifically in years with high March rainfall, during what is otherwise the dry season, when food availability was greater, American Redstarts departed earlier on spring migration (Studds and Marra 2011).

Conclusions and Management Implications

My results suggest that kinglets demonstrate age/sex class segregation into different burn treatment areas. The segregation is largely dependent on the age/sex class; larger-bodied birds, which tend to be males, appear to occupy preferred habitats. Both burned and unburned habitats appear to provide high-quality habitat, and differences in patterns of abundance appear to depend largely on the habitat structure and fire history, with relatively few birds in frequently burned habitats that lack a well-developed scrub layer. Kinglets benefit in the winter season from a mix of both shrub and tree cover, which is consistent with McClure et al.'s (2012) observation that Ruby-crowned Kinglets prefer young-staged pine forest. More research is needed to understand how specific time-since-fire successional trends affect the winter ecology of kinglets, and other species. In addition, more observations are needed in the unburned sand pine scrub to better quantify the habitat use. Studies addressing flock behavior, territorial overlap, microhabitats characteristics, body condition, and food quantification would help us to further understand the effects of variation in winter habitat quality. Additional years of study would allow for a better understanding of how variations in environmental conditions affect the winter ecology of kinglets. Laurenzi et al. (1982) suggested that winter population limitation of kinglets was related to weather patterns. My study included just two winters, one of which was unusually cold and wet, and included a severe ice storm that likely led to reduced apparent survival, suggesting the importance of environmental conditions.

The effects of fire-based habitat management at Naval Live Oaks on the behavior of Ruby-crowned Kinglets appears to vary based of the intensity and frequency of fire

and the pre-burn habitat (i.e. oak vs. pine). A management approach designed to maintain a variety of habitats in relatively small patches may maximize conditions for kinglets of multiple demographic classes. My own anecdotal observations suggest that ecotones can provide high quality conditions; however, I did not specifically test this pattern. Kinglets appear to be at least partially limited by events occurring in the winter period. These events include extreme weather, such as ice storms, but also more subtle differences in habitat type that affect abundance, foraging, age/sex class ratios, and departure timing. Kinglet populations are declining in the eastern region of North America. Although it is remains uncertain to what degree winter events are contributing to population changes, we can now recommend kinglets would benefit from management strategies which promote younger stage pine, or at least early successional forest stages, with a floristically diverse shrub layer such as that found in the sand pine scrub at Naval Live Oaks. High intensity and frequent fires appears to have negative effects on habitat quality for kinglets at this site given a lower abundance observed compared to other habitats tested. From this study, low-intensity burns at low frequencies to allow development of dense shrub layers in some patches would be best suited to promote high quality habitat in areas with fire management. GUIs, and in particular Naval Live Oaks is managed for a variety of flora and fauna, including Gopher Tortoise (*Gopherus polyphemus*), and long leaf pine which in some case does not promote suitable habitat. However, the current management practices at Naval Live Oaks appear do create suitable habitat for kinglets, but more work may be necessary to identify the best habitat management approach for this species.

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Appendix A: Tables

Table 1. Mean abundance and standard errors of bird species observed in fixed area searches at Naval Live Oaks during 2013.

Species	Status	Guild	Unburned	Burned	Burned	Unburned	Burned
			Mixed Canopy	Longleaf Pine	Oak Scrub	Sand Pine Scrub	Sand Pine Scrub
<i>Pandion haliaetus</i> (Osprey)		PE	0.08 ± 0.08				
<i>Zenaida macroura</i> (Mourning Dove)		GR	0.25 ± 0.16	0.08 ± 0.08	0.11 ± 0.11	0.13 ± 0.08	0.33 ± 0.24
<i>Melanerpes carolinus</i> (Red-bellied Woodpecker)		OM	0.75 ± 0.16	0.50 ± 0.22	0.11 ± 0.11	0.13 ± 0.08	0.24 ± 0.10
<i>Sphyrapicus varius</i> (Yellow-bellied Sapsucker)	M	OM	0.08 ± 0.08				0.05 ± 0.05
<i>Picoides villosus</i> (Hairy Woodpecker)		OM	0.08 ± 0.08			0.07 ± 0.07	0.05 ± 0.05
<i>Dryocopus pileatus</i> (Pileated Woodpecker)		OM					0.05 ± 0.05
<i>Sayornis phoebe</i> (Eastern Phoebe)	M	AI				0.07 ± 0.07	
<i>Cyanocitta cristata</i> (Blue Jay)		OM	1.50 ± 0.62		0.33 ± 0.33	0.33 ± 0.21	0.29 ± 0.13
<i>Poecile carolinensis</i> (Carolina Chickadee)		OM	0.67 ± 0.27	0.42 ± 0.25	1.00 ± 0.38	0.67 ± 0.28	1.14 ± 0.45
<i>Baeolophus bicolor</i> (Tufted Titmouse)		OM	0.25 ± 0.08			0.07 ± 0.07	0.10 ± 0.10
<i>Sitta pusilla</i> (Brown-headed Nuthatch)		OM		0.17 ± 0.10	0.11 ± 0.11	0.13 ± 0.13	0.14 ± 0.14
<i>Troglodytes aedon</i> (House Wren)	M	FI					0.05 ± 0.05
<i>Thryothorus ludovicianus</i> (Carolina Wren)		FI	0.17 ± 0.17	0.25 ± 0.16		0.33 ± 0.15	0.24 ± 0.16
<i>Poliophtila caerulea</i> (Blue-gray Gnatcatcher)		FI	0.25 ± 0.16			0.13 ± 0.08	0.10 ± 0.06
<i>Regulus calendula</i> (Ruby-crowned Kinglet)	M	FI	0.42 ± 0.25		0.33 ± 0.19	0.73 ± 0.07	0.24 ± 0.12
<i>Sialia sialis</i> (Eastern Bluebird)		OM		1.08 ± 0.37			
<i>Catharus guttatus</i> (Hermit Thrush)	M	OM	0.08 ± 0.08		0.22 ± 0.22		
<i>Turdus migratorius</i> (American Robin)	M	OM	0.50 ± 0.50	0.83 ± 0.63			0.19 ± 0.14
<i>Dumetella carolinensis</i> (Gray Catbird)	M	OM	0.42 ± 0.16		0.33 ± 0.19	0.07 ± 0.07	0.05 ± 0.05
<i>Toxostoma rufum</i> (Brown Thrasher)		OM					0.05 ± 0.05
<i>Mimus polyglottos</i> (Northern Mockingbird)		OM	0.17 ± 0.17	0.33 ± 0.33	0.33 ± 0.19		0.24 ± 0.16
<i>Bombicilla cedrorum</i> (Cedar Waxwing)	M	FR/IN		2.50 ± 2.50			0.48 ± 0.48
<i>Oreothypis celata</i> (Orange-crowned Warbler)	M	FI/FR	0.17 ± 0.10	0.08 ± 0.08		0.07 ± 0.07	
<i>Setophaga palmarum</i> (Palm Warbler)	M	FI/FR					0.10 ± 0.10
<i>Setophaga pinus</i> (Pine Warbler)		OM		0.33 ± 0.14		0.13 ± 0.08	
<i>Setophaga coronata</i> (Yellow-rumped Warbler)	M	OM	0.67 ± 0.45	2.58 ± 0.37	2.89 ± 0.99	2.60 ± 0.68	1.38 ± 0.46
<i>Pipilo erythrophthalmus</i> (Eastern Towhee)		OM		0.08 ± 0.08	1.44 ± 0.68	0.20 ± 0.08	0.33 ± 0.13
<i>Spizella passerina</i> (Chipping Sparrow)	M	OM					0.05 ± 0.05
<i>Passerculus sandwichensis</i> (Savannah Sparrow)	M	OM					0.05 ± 0.05
<i>Melospiza melodia</i> (Song Sparrow)	M	OM			0.22 ± 0.22		
<i>Zonotrichia albicollis</i> (White-throated Sparrow)	M	OM	0.08 ± 0.08		2.89 ± 0.68	1.47 ± 0.50	0.76 ± 0.55
<i>Cardinalis cardinalis</i> (Northern Cardinal)		OM	1.75 ± 0.50	0.17 ± 0.17	1.00 ± 0.51	1.00 ± 0.35	0.95 ± 0.33

Table 2. Mean abundance and standard errors of bird species observed in fixed area searches at Naval Live Oaks during 2014.

Species	Status	Guild	Unburned	Burned	Burned	Unburned	Burned
			Mixed Canopy	Longleaf Pine	Oak Scrub	Sand Pine Scrub	Sand Pine Scrub
<i>Accipiter striatus</i> (Sharp-shinned Hawk)	M	CA			0.11 ± 0.11		
<i>Buteo lineatus</i> (Red-shouldered Hawk)		CA					0.05 ± 0.05
<i>Scolopax minor</i> (American Woodcock)	M	IN			0.11 ± 0.11		
<i>Zenaida macroura</i> (Mourning Dove)		GR	0.17 ± 0.17	0.08 ± 0.08	0.22 ± 0.22	0.27 ± 0.16	0.24 ± 0.14
<i>Melanerpes carolinus</i> (Red-bellied Woodpecker)		OM	0.50 ± 0.10	0.25 ± 0.16	0.11 ± 0.11	0.20 ± 0.08	0.29 ± 0.09
<i>Sphyrapicus varius</i> (Yellow-bellied Sapsucker)	M	OM	0.08 ± 0.08				0.05 ± 0.05
<i>Picoides pubescens</i> (Downy Woodpecker)		OM	0.17 ± 0.10		0.11 ± 0.11	0.27 ± 0.07	0.19 ± 0.19
<i>Picoides villosus</i> (Hairy Woodpecker)		OM	0.17 ± 0.17				
<i>Colaptes auratus</i> (Northern Flicker)		IN/OM		0.08 ± 0.08			0.10 ± 0.06
<i>Falco sparverius</i> (American Kestrel)	M	CA/IN					0.05 ± 0.05
<i>Sayornis phoebe</i> (Eastern Phoebe)	M	AI		0.08 ± 0.08			0.05 ± 0.05
<i>Cyanocitta cristata</i> (Blue Jay)		OM	0.42 ± 0.16	0.33 ± 0.33	0.33 ± 0.00	0.67 ± 0.21	0.48 ± 0.27
<i>Poecile carolinensis</i> (Carolina Chickadee)		OM	0.50 ± 0.29	0.25 ± 0.16	0.22 ± 0.22	0.53 ± 0.23	0.43 ± 0.28
<i>Baeolophus bicolor</i> (Tufted Titmouse)		OM	0.25 ± 0.16	0.17 ± 0.17	0.11 ± 0.11	0.27 ± 0.19	0.24 ± 0.14
<i>Sitta pusilla</i> (Brown-headed Nuthatch)		OM		0.50 ± 0.22			0.19 ± 0.12
<i>Troglodytes aedon</i> (House Wren)	M	FI		0.08 ± 0.08			
<i>Tryothorus ludovicianus</i> (Carolina Wren)		FI	0.75 ± 0.16	0.50 ± 0.22	0.89 ± 0.22	1.00 ± 0.24	0.57 ± 0.20
<i>Poliopitila caerulea</i> (Blue-gray Gnatcatcher)		FI	0.25 ± 0.08			0.20 ± 0.13	0.05 ± 0.05
<i>Regulus calendula</i> (Ruby-crowned Kinglet)	M	FI	0.50 ± 0.22		0.22 ± 0.11	0.60 ± 0.19	0.67 ± 0.15
<i>Sialia sialis</i> (Eastern Bluebird)		OM		0.92 ± 0.32			0.05 ± 0.05
<i>Catharus guttatus</i> (Hermit Thrush)	M	OM	0.08 ± 0.08				
<i>Turdus migratorius</i> (American Robin)	M	OM	1.00 ± 0.58	0.75 ± 0.28	0.22 ± 0.11	0.60 ± 0.19	0.43 ± 0.14
<i>Dumetella carolinensis</i> (Gray Catbird)	M	OM	0.17 ± 0.17	0.17 ± 0.10	0.56 ± 0.40	0.13 ± 0.08	0.19 ± 0.12
<i>Toxostoma rufum</i> (Brown Thrasher)		OM	0.33 ± 0.24	0.17 ± 0.17	1.44 ± 0.80	0.40 ± 0.19	0.24 ± 0.10
<i>Mimus polyglottos</i> (Northern Mockingbird)		OM		0.42 ± 0.42		0.07 ± 0.07	0.10 ± 0.10
<i>Bombycilla cedrorum</i> (Cedar Waxwing)	M	FR/IN	1.25 ± 1.25	0.08 ± 0.08			0.14 ± 0.14
<i>Vermivora celata</i> (Orange-crowned Warbler)	M	FI/FR	0.17 ± 0.10			0.07 ± 0.07	0.05 ± 0.05
<i>Dendroica palmarum</i> (Palm Warbler)	M	FI/FR		0.08 ± 0.08			0.19 ± 0.19
<i>Dendroica pinus</i> (Pine Warbler)		OM		0.67 ± 0.47			
<i>Dendroica coronata</i> (Yellow-rumped Warbler)	M	OM	1.25 ± 0.34	4.58 ± 2.31	0.78 ± 0.48	2.13 ± 1.05	1.33 ± 0.10
<i>Pipilo erythrophthalmus</i> (Eastern Towhee)		OM	0.17 ± 0.10	0.42 ± 0.21	1.44 ± 0.59	0.93 ± 0.36	0.71 ± 0.18
<i>Melospiza georgiana</i> (Swamp Sparrow)	M	OM		0.17 ± 0.17			
<i>Zonotrichia albicollis</i> (White-throated Sparrow)	M	OM	0.42 ± 0.16	0.17 ± 0.17	3.11 ± 1.16	1.27 ± 0.12	0.57 ± 0.33
<i>Zonotrichia leucophrys</i> (White-crowned Sparrow)	M	OM		0.17 ± 0.17			
<i>Cardinalis cardinalis</i> (Northern Cardinal)		OM	1.33 ± 0.30	0.67 ± 0.27	0.44 ± 0.22	0.33 ± 0.18	0.14 ± 0.14
<i>Haemorhous mexicanus</i> (House Finch)		GR/IN				0.13 ± 0.08	

Table 3. Summary statistics for bird communities among habitats based on fixed area searches during 2013. CA/IN = Carnivore/Insectivore, GR/IN = Granivore/Insectivore, IN/OM = Insectivore/Omnivore, FI/FR = Foliage Insectivore/Frugivore, and FR/IN = Frugivore/Insectivore.

	Unburned Mixed Canopy	Burned Longleaf Pine	Burned Oak Scrub	Unburned Sand Pine Scrub	Burned Sand Pine Scrub
Richness and Diversity					
Total Number Counted	94	113	102	118	160
Total Richness 0-50 m	18	13	14	18	25
Average Richness	9.75 ± 0.63	6.75 ± 0.63	9.00 ± 1.15	7.50 ± 0.45	8.14 ± 1.34
Simpson (D)	0.85 ± 0.02	0.72 ± 0.03	0.81 ± 0.01	0.67 ± 0.02	0.77 ± 0.05
Shannon (H)	2.06 ± 0.10	1.54 ± 0.09	1.87 ± 0.07	1.58 ± 0.10	1.74 ± 0.18
ENS	7.85	4.66	6.49	4.85	5.70
Migratory Status					
Migrants (%)	7 (38.88)	3 (23.07)	6 (42.86)	6 (33.33)	11 (44.00)
Migrants Counted (%)	29 (30.85)	72 (63.72)	62 (60.78)	68 (57.63)	71 (44.38)
Foraging Guild					
Aerial Insectavore	.	.	.	1	.
Foliage Insectavore	3	1	1	3	4
Granivore	1	1	1	1	1
Omnivore	12	10	12	12	18
Piscivore	1
Carnivore
Insectovore
CA/IN
GR/IN
IN/OM
FI/FR	1	.	.	1	1
FR/IN	.	1	.	.	1

Table 4. Summary statistics for bird communities among habitats based on fixed area searches during 2014. CA/IN = Carnivore/Insectivore, GR/IN = Granivore/Insectivore, IN/OM = Insectivore/Omnivore, FI/FR = Foliage Insectivore/Frugivore, and FR/IN = Frugivore/Insectivore.

	Unburned Mixed Canopy	Burned Longleaf Pine	Burned Oak Scrub	Unburned Sand Pine Scrub	Burned Sand Pine Scrub
Richness and Diversity					
Total Number Counted	120	141	94	152	164
Total Richness 0-50 m	22	24	17	19	27
Average Richness	12.50 ± 1.32	10.75 ± 1.70	10.33 ± 0.88	12.60 ± 0.68	10.29 ± 1.15
Simpson (D)	0.86 ± 0.01	0.79 ± 0.04	0.79 ± 0.04	0.85 ± 0.04	0.83 ± 0.01
Shannon (H)	2.27 ± 0.05	1.95 ± 0.13	2.03 ± 0.03	2.27 ± 0.16	2.13 ± 0.13
ENS	9.68	7.03	7.61	9.68	8.41
Migratory Status					
Migrants (%)	9 (40.90)	10 (41.66)	7 (41.18)	6 (31.58)	11 (40.74)
Migrants Counted (%)	59 (24.17)	76 (53.90)	46 (48.94)	72 (47.37)	78 (47.56)
Foraging Guild					
Aerial Insectivore	.	1	.	.	1
Foliage Insectivore	3	2	2	3	3
Granivore	1	1	1	1	1
Omnivore	15	17	12	13	16
Piscivore
Carnivore	.	.	1	.	1
Insectivore	1
CA/IN	1
GR/IN	.	.	.	1	.
IN/OM	.	1	1	.	1
FI/FR	1	1	.	1	2
FR/IN	1	1	.	.	1

Table 5. Means and standard errors of vegetation measurements for five habitats at Naval Live Oaks area.

		Unburned Mixed Canopy	Burned Longleaf Pine	Burned Oak Scrub	Unburned Sand Pine Scrub	Burned Sand Pine Scrub
Cover Estimates	Tree	4.25 ± 0.25	2.50 ± 0.29	1.00 ± 0.58	2.80 ± 0.37	2.29 ± 0.36
	Shrub	3.50 ± 0.29	3.00 ± 0.41	4.67 ± 0.33	4.40 ± 0.40	3.14 ± 0.26
	Herb	1.75 ± 0.25	3.00 ± 0.00	2.00 ± 0.58	2.00 ± 0.32	3.00 ± 0.22
	Grass	0.75 ± 0.25	3.00 ± 0.41	1.33 ± 0.33	0.80 ± 0.37	2.00 ± 0.31
	Forbs	1.25 ± 0.25	2.00 ± 0.41	1.33 ± 0.33	2.00 ± 0.32	2.71 ± 0.29
	Fruit Abundance	1.00 ± 0.71	1.75 ± 0.75	0.00 ± 0.00	0.80 ± 0.37	1.14 ± 0.34
	Nut Abundance	2.25 ± 0.48	0.75 ± 0.25	1.33 ± 0.33	1.20 ± 0.20	1.43 ± 0.30
Ground Composition	Leaf Litter	5.00 ± 0.00	1.75 ± 0.25	4.00 ± 0.00	4.20 ± 0.37	2.86 ± 0.26
	Moss/Lichen	0.50 ± 0.29	0.00 ± 0.00	0.00 ± 0.00	2.40 ± 0.24	0.57 ± 0.20
	Bare Ground	1.00 ± 0.00	1.75 ± 0.48	3.00 ± 0.00	1.20 ± 0.37	2.29 ± 0.29
	Vegetation	1.25 ± 0.25	2.25 ± 0.48	1.00 ± 0.00	1.40 ± 0.24	2.00 ± 0.22
	Woody Debris	2.00 ± 0.00	2.00 ± 0.00	2.00 ± 0.00	2.60 ± 0.24	2.29 ± 0.18
	Pine Needles	0.25 ± 0.25	4.00 ± 0.71	0.33 ± 0.33	2.80 ± 0.37	2.43 ± 0.61
Robel Pole in % Obstructed	4-m 0.00-0.50 m	6.25 ± 3.15	16.25 ± 6.88	45.00 ± 16.07	43.00 ± 9.70	18.57 ± 5.85
	4-m 0.51-1.00 m	1.25 ± 1.25	3.75 ± 2.39	31.67 ± 24.21	29.00 ± 15.03	7.86 ± 4.06
	4-m 1.01-1.50 m	3.75 ± 2.39	5.00 ± 3.54	5.00 ± 5.00	11.00 ± 7.48	2.14 ± 2.14
	4-m 1.51-2.00 m	3.75 ± 3.75	0.00 ± 0.00	0.00 ± 0.00	17.00 ± 10.20	0.71 ± 0.71
	Avg. Total at 4-m	3.75 ± 1.53	6.25 ± 2.22	20.42 ± 11.19	25.00 ± 9.57	7.32 ± 2.35
	12-m 0.00-0.50 m	61.25 ± 10.68	62.50 ± 9.24	93.33 ± 3.33	90.00 ± 5.24	77.14 ± 8.37
	12-m 0.51-1.00 m	32.50 ± 10.51	16.25 ± 8.26	75.00 ± 14.43	80.00 ± 14.58	46.43 ± 11.64
	12-m 1.01-1.50 m	35.00 ± 16.71	2.50 ± 2.50	45.00 ± 25.17	59.00 ± 16.39	16.43 ± 8.14
	12-m 1.51-2.00 m	28.75 ± 10.28	3.75 ± 3.75	23.33 ± 20.88	40.00 ± 13.60	7.14 ± 4.06
	Avg. Total at 12-m	39.38 ± 11.14	21.25 ± 4.87	59.17 ± 15.57	67.25 ± 11.91	36.79 ± 7.26
Tree	Canopy Closure	77.73 ± 3.64	29.53 ± 8.46	8.17 ± 6.83	35.87 ± 11.19	17.42 ± 5.97
	Snag Count	2.00 ± 1.35	15.00 ± 3.11	22.00 ± 15.50	16.60 ± 4.65	27.14 ± 4.76
	Avg. Canopy Ht.	6.99 ± 1.50	13.70 ± 1.76	3.64 ± 1.87	4.92 ± 0.30	6.44 ± 1.08
	Tallest Tree	19.60 ± 2.23	20.31 ± 2.28	8.57 ± 1.64	14.73 ± 0.97	9.41 ± 1.17
	Tallest Tree Dbh	44.13 ± 10.16	42.78 ± 4.88	20.85 ± 8.55	27.54 ± 3.74	19.36 ± 2.75

Table 6. Loading scores from the principal component analysis of vegetation measurements measured in 2013. Variable loadings with considerable influence (> 0.4) are in bold.

		PC1	PC2	PC3	PC4
Tree	Tree Cover	0.890	-0.106	-0.166	-0.086
	Canopy Closure	0.948	0.115	-0.150	-0.037
	Tallest Tree	0.804	-0.266	0.298	0.240
	Tallest Tree DBH	0.710	-0.247	0.221	0.362
Shrub	Shrub Cover	0.000	0.890	0.245	0.167
	Total 4-m Robel	0.050	0.701	0.604	0.033
	Total 12-m Robel	0.018	0.869	0.305	0.083
Ground	Total Ground Vegetation	-0.192	-0.484	0.205	0.659
	% Leaf Litter	0.364	0.770	-0.421	0.053
	% Bare Ground	-0.859	0.026	-0.127	0.402
	% Pine Needles	0.172	-0.449	0.723	-0.339
	Snag Count	-0.651	-0.077	0.281	-0.262

Table 7. Spearman correlation coefficients (r_s values) and p values of bird abundance by species with the vegetation principal component scores for 2013. Statistically significant correlations ($P < 0.05$) are indicated by bold font.

Species	PC 1	PC 2	PC 3	PC 4
<i>Zenaida macroura</i> (Mourning Dove)	0.106	-0.184	0.192	-0.370
<i>Melanerpes carolinus</i> (Red-bellied Woodpecker)	0.520	-0.366	-0.058	0.003
<i>Cyanocitta cristata</i> (Blue Jay)	0.145	0.115	-0.289	-0.354
<i>Poecile carolinensis</i> (Carolina Chickadee)	-0.103	0.214	-0.110	-0.063
<i>Sitta pusilla</i> (Brown-headed Nuthatch)	-0.079	0.099	0.003	0.442
<i>Thryothorus ludovicianus</i> (Carolina Wren)	-0.104	-0.106	0.149	0.133
<i>Poliophtila caerulea</i> (Blue-gray Gnatcatcher)	0.152	-0.254	-0.293	0.106
<i>Regulus calendula</i> (Ruby-crowned Kinglet)	-0.393	-0.223	-0.289	0.006
<i>Sialia sialis</i> (Eastern Bluebird)	0.140	-0.174	0.283	0.432
<i>Turdus migratorius</i> (American Robin)	0.421	-0.233	0.182	0.054
<i>Dumetella carolinensis</i> (Gray Catbird)	-0.023	-0.340	-0.488	-0.426
<i>Mimus polyglottos</i> (Northern Mockingbird)	0.018	0.168	0.037	0.104
<i>Bombycilla cedrorum</i> (Cedar Waxwing)	0.164	-0.420	0.441	-0.040
<i>Setophaga coronata</i> (Yellow-rumped Warbler)	-0.497	-0.279	0.056	0.305
<i>Pipilo erythrophthalmus</i> (Eastern Towhee)	-0.456	0.434	-0.218	0.130
<i>Zonotrichia albicollis</i> (White-throated Sparrow)	-0.561	0.223	-0.052	-0.177
<i>Cardinalis cardinalis</i> (Northern Cardinal)	-0.035	-0.363	-0.314	-0.379

Table 8. Spearman correlation coefficients (r_s values) and p values of bird abundance by species with the vegetation principal component scores for 2014. Statistically significant correlations ($P < 0.05$) are indicated by bold font.

Species	PC 1	PC 2	PC 3	PC 4
<i>Melanerpes carolinus</i> (Red-bellied Woodpecker)	0.382	-0.156	-0.340	-0.120
<i>Sphyrapicus varius</i> (Yellow-bellied Sapsucker)	0.140	-0.302	-0.163	0.047
<i>Picoides pubescens</i> (Downy Woodpecker)	-0.129	-0.220	-0.057	-0.175
<i>Colaptes auratus</i> (Northern Flicker)	0.097	-0.253	0.370	-0.156
<i>Sayornis phoebe</i> (Eastern Phoebe)	0.279	-0.203	0.163	0.116
<i>Cyanocitta cristata</i> (Blue Jay)	-0.304	-0.408	-0.204	0.009
<i>Poecile carolinensis</i> (Carolina Chickadee)	0.011	-0.001	-0.246	0.348
<i>Baeolophus bicolor</i> (Tufted Titmouse)	0.038	0.253	-0.067	0.383
<i>Sitta pusilla</i> (Brown-headed Nuthatch)	0.381	0.154	0.505	0.112
<i>Tryothorus ludovicianus</i> (Carolina Wren)	-0.400	0.058	-0.249	0.040
<i>Poliophtila caerulea</i> (Blue-gray Gnatcatcher)	0.135	0.138	-0.297	-0.001
<i>Regulus calendula</i> (Ruby-crowned Kinglet)	-0.036	-0.211	-0.031	-0.054
<i>Sialia sialis</i> (Eastern Bluebird)	0.154	-0.201	0.329	0.157
<i>Turdus migratorius</i> (American Robin)	0.138	-0.269	-0.074	0.046
<i>Dumetella carolinensis</i> (Gray Catbird)	-0.195	-0.114	-0.040	-0.221
<i>Toxostoma rufum</i> (Brown Thrasher)	-0.314	0.251	-0.160	-0.367
<i>Mimus polyglottos</i> (Northern Mockingbird)	-0.004	-0.185	0.238	-0.034
<i>Bombycilla cedrorum</i> (Cedar Waxwing)	0.374	-0.192	-0.090	-0.096
<i>Vermivora celata</i> (Orange-crowned Warbler)	0.138	0.121	-0.277	0.156
<i>Dendroica palmarum</i> (Palm Warbler)	0.109	-0.333	0.317	0.240
<i>Dendroica pinus</i> (Pine Warbler)	0.257	0.016	0.378	0.245
<i>Dendroica coronata</i> (Yellow-rumped Warbler)	-0.025	-0.493	0.279	-0.014
<i>Pipilo erythrophthalmus</i> (Eastern Towhee)	-0.538	0.233	0.059	-0.167
<i>Zonotrichia albicollis</i> (White-throated Sparrow)	-0.601	0.166	-0.170	-0.136
<i>Cardinalis cardinalis</i> (Northern Cardinal)	0.304	-0.074	-0.313	-0.243

Table 9. Morphometrical summary of Ruby-crowned Kinglets and the one-way ANOVA statistics among age/sex classes. The body size variable is a composite score based on PC1. The second line for each variable shows the source table values for the error term. Groups labeled as Male and Female are ASY age-class birds.

	Mean \pm Standard Error			DF	ANOVA Table			
	Male	Female	SY		Sum Sq.	Mean Sq.	F	P
Mass	6.44 \pm 0.06	6.14 \pm 0.14	5.94 \pm 0.10	2	2.257	1.1283	7.245	0.002
				50	7.786	0.1557		
Muscle Score	2.12 \pm 0.07	1.97 \pm 0.03	1.93 \pm 0.07	2	0.42	0.21019	2.83	0.068
				51	3.788	0.07427		
Fat Score	2.21 \pm 0.29	2.5 \pm 0.34	2.96 \pm 0.28	2	5.05	2.527	1.243	0.243
				51	88.69	1.739		
Wing Chord	58.92 \pm 0.28	55.62 \pm 0.34	55.39 \pm 0.49	2	154.5	77.27	34.52	<0.001
				51	114.2	2.24		
Tail Length	43.65 \pm 0.47	41.16 \pm 0.50	40.39 \pm 0.63	2	112.3	56.17	11.29	<0.001
				51	253.7	4.97		
Tarsus Length	18.98 \pm 0.10	18.77 \pm 0.13	18.24 \pm 0.16	2	4.904	2.4519	9	<0.001
				51	13.891	0.2724		
Body Size	0.83 \pm 0.13	-0.47 \pm 0.13	-0.88 \pm 0.22	2	30.98	15.492	35.89	<0.001
				51	22.02	0.0432		

Table 10. Overall survival and detection probability for all kinglets for the winter and non-winter periods with standard errors.

Period	Time Frame	Survival Probability	Detection Probability
Mid-winter 2013	Jan-Feb	0.83 \pm 0.08	0.94 \pm 0.06
Late-winter 2013	Feb-Mar	1.00 \pm 0.27 x 10 ⁻⁷	0.89 \pm 0.07
Pre-migratory/Migratory/Breeding	Mar-Jan	0.88 \pm 0.04	0.50 \pm 0.23
Mid-winter2014	Jan-Feb	0.68 \pm 0.08	0.78 \pm 0.09
Later-winter 2014	Feb-Mar	1.00 \pm 0.00	1.00 \pm 0.00

Table 11. Mean foraging rates and standard errors for kinglets between years, age/sex classes and burn treatments. Groups labeled as Male and Female are ASY age-class birds.

	n	Proportion of				
		Time Foraging	Foraging Rate	Glean Rate	Sally Hover Rate	Attack Rate
2013	13	97.36 ± 0.60	22.76 ± 0.84	2.09 ± 0.28	1.41 ± 0.22	3.65 ± 0.45
2014	19	95.99 ± 1.22	27.38 ± 0.76	2.36 ± 0.34	1.11 ± 0.20	3.73 ± 0.34
Male (All)	16	94.67 ± 1.32	24.47 ± 0.88	2.28 ± 0.33	1.36 ± 0.23	3.86 ± 0.38
Male (Burn)	11	94.90 ± 1.83	24.5 ± 1.21	2.11 ± 0.36	1.65 ± 0.26	3.96 ± 0.48
Male(Unburn)	5	94.19 ± 1.50	24.41 ± 1.14	2.66 ± 0.74	0.72 ± 0.3	3.64 ± 0.67
Female (All)	9	98.82 ± 0.60	26.08 ± 1.43	1.83 ± 0.3	1.09 ± 0.27	3.2 ± 0.46
Female (Burn)	5	99.1 ± 0.68	23.53 ± 1.63	1.96 ± 0.51	1.66 ± 0.26	3.91 ± 0.67
Female(Unburn)	4	98.47 ± 1.14	29.26 ± 1.36	1.66 ± 0.29	0.38 ± 0.19	2.32 ± 0.18
SY ALL	7	97.90 ± 0.75	27.13 ± 1.55	2.72 ± 0.62	1.11 ± 0.27	3.95 ± 0.63
SY (Burn)	4	98.37 ± 0.89	27.49 ± 1.15	2.26 ± 0.8	1.45 ± 0.31	3.77 ± 1.02
SY(Unburn)	3	97.28 ± 1.45	26.65 ± 3.73	3.33 ± 1.04	0.67 ± 0.39	4.19 ± 0.79

Table 12. The 10 best models of departure timing based on ΔAIC_c of all possible variable combinations relative to the full global model (+ sign indicates categorical variable).

Intercept	Attack Rate	Habitat	Burn Treatment	Sex	Body Size	Home Range	Attack Rate * Body Size	Habitat * Body Size	Burn Treatment * Body Size	Sex * Body	Body Size * Home Range	df	logLik	AICc	Delta AICc	Weight
34.59					-5.81							3	-107.56	222	0	0.205
33.75		+			-5.9							4	-107.36	224	2.23	0.067
35.55					-5.62	-0.246						4	-107.38	224	2.25	0.067
36.08	-0.404				-5.82							4	-107.38	224	2.27	0.066
34.62			+		-5.81							4	-107.56	225	2.62	0.055
31.65				+	-3.63							5	-106.4	225	3.13	0.043
33.82		+			-4.33			+				5	-106.73	226	3.79	0.031
28.12				+								4	-108.41	226	4.33	0.024
35.48	-0.536	+			-5.96							5	-107.07	227	4.48	0.022
31.57		+	+		-5.95							5	-107.1	227	4.52	0.021
37.65	-0.504				-5.59	-0.307						5	-107.11	227	4.55	0.021
34.73		+			-5.7	-0.274						5	-107.14	227	4.61	0.021
36.59	-0.53				-7.06		0.3156					5	-107.21	227	4.76	0.019
34.59					-4.83	0.0618					-0.2446	5	-107.24	227	4.8	0.019
30.67				+	-2.65							7	-104.14	227	4.98	0.017

Table 13. Model averaged coefficients for departure timing from the average of the best five models. Negative coefficient estimates indicate variables that were negatively related to departure date. For instance, larger body sized birds departed earlier.

	Estimate	SE	Adjusted SE	z value	Pr(> z)
Intercept	34.82	1.78	1.85	18.84	< 0.001
Body Size	-5.80	0.94	0.98	5.92	< 0.001
Habitat	1.35	2.14	2.23	0.60	0.547
Home Range	-0.25	0.40	0.41	0.59	0.552
Attack Rate	-0.40	0.67	0.70	0.57	0.567
Burn Treatment	-0.07	2.13	2.22	0.03	0.974

Appendix B: Figures



Figure 1. Naval Live Oaks Area showing sites of the 25 fixed area searches.

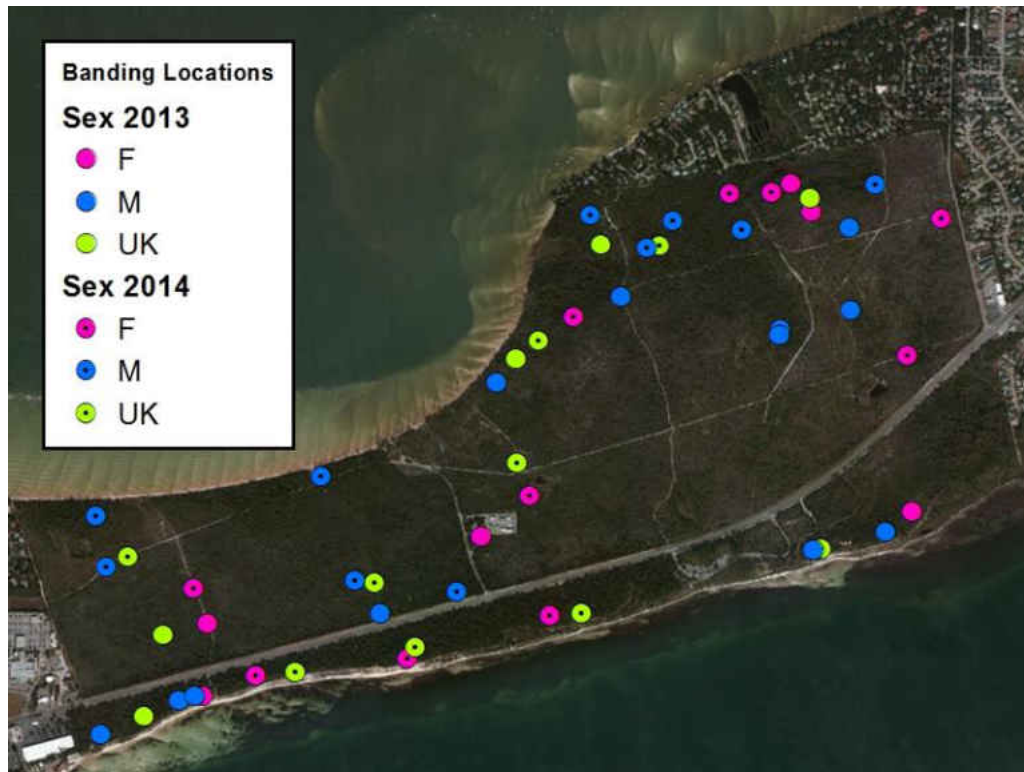


Figure 2. Naval Live Oaks area map showing the banding locations of all the birds captured.

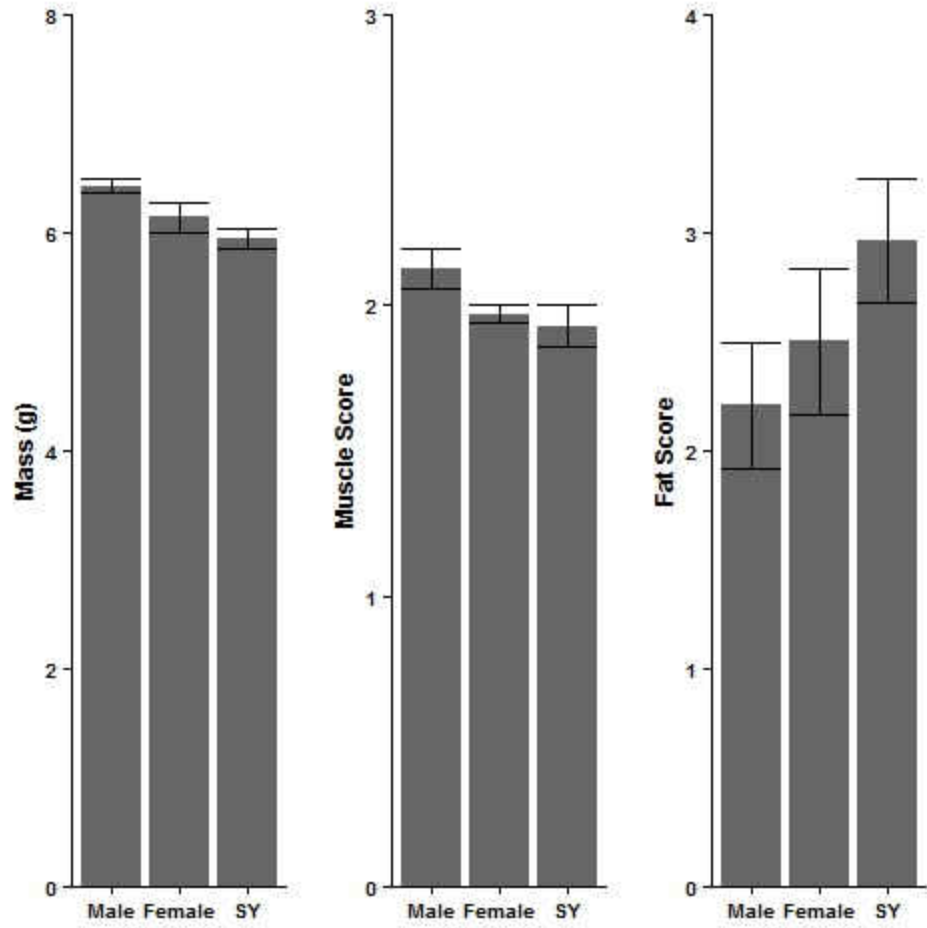


Figure 3. Mean \pm standard error for the total mass, muscle, and fat scores for each age/sex class. Groups labeled as Male and Female are ASY age-class birds.

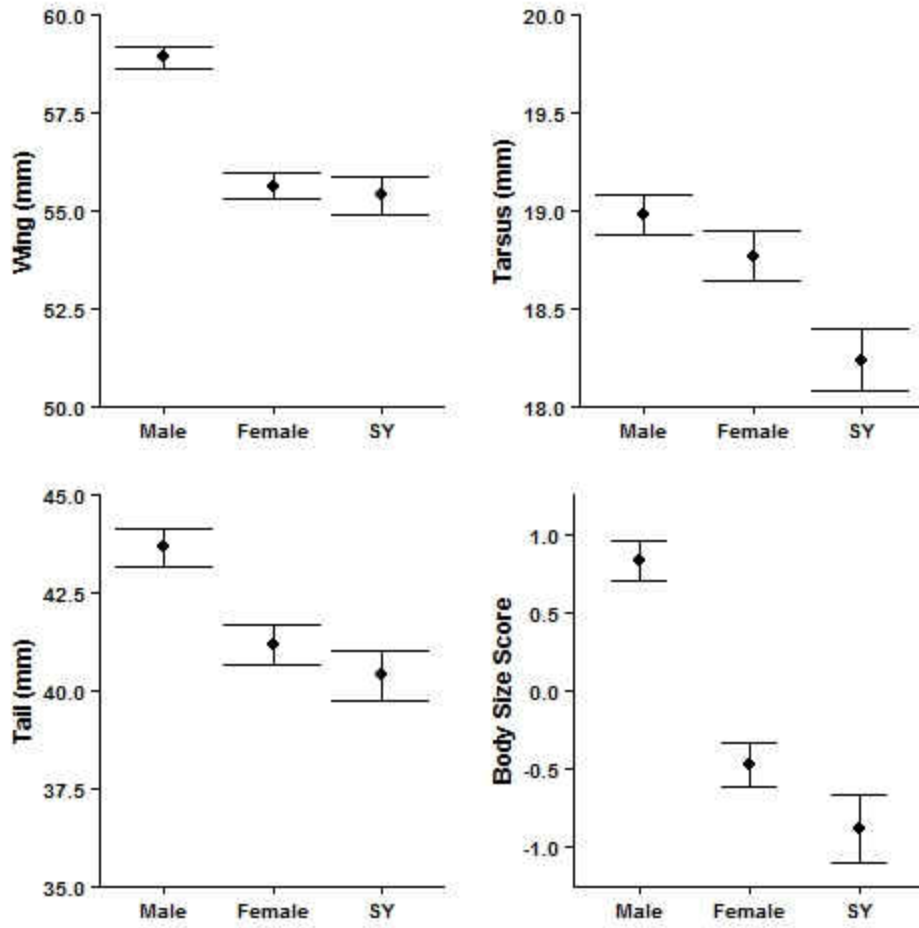


Figure 4. Mean \pm standard error for the wing chord, tail length, tarsus length and structural body size component score for each age/sex class. Groups labeled as Male and Female are ASY age-class birds.

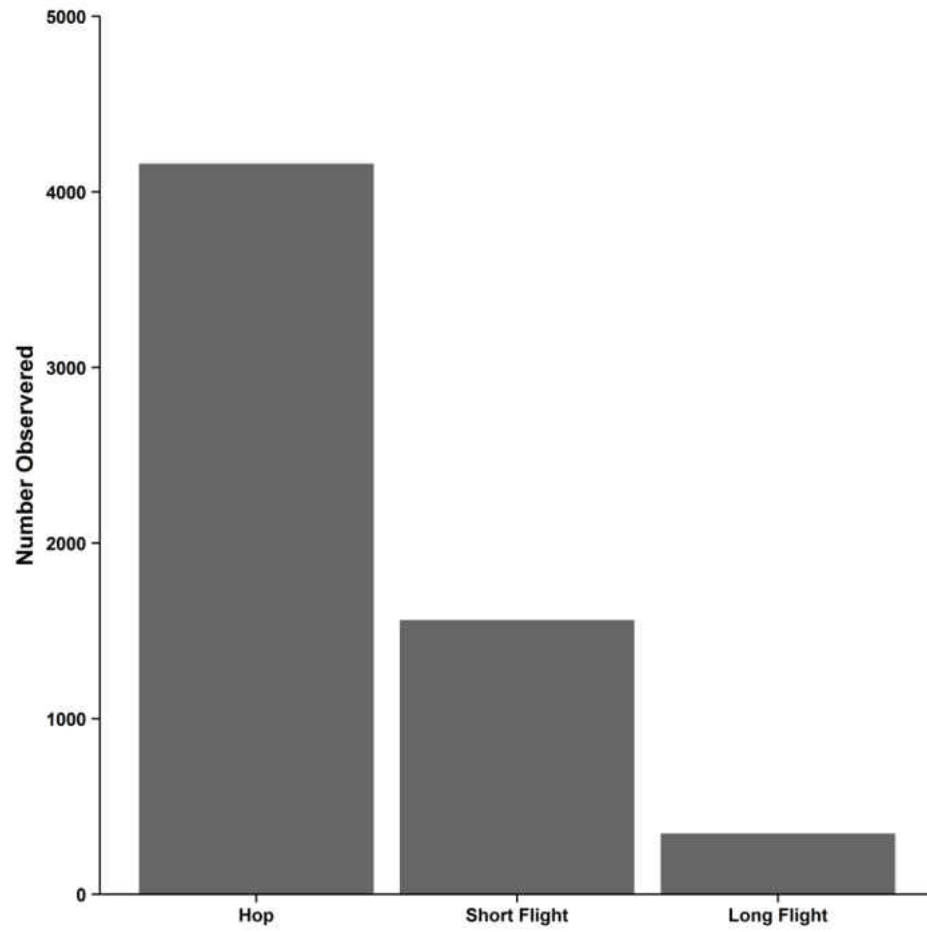


Figure 5. Total frequency of all foraging movements observed over the two years of study.

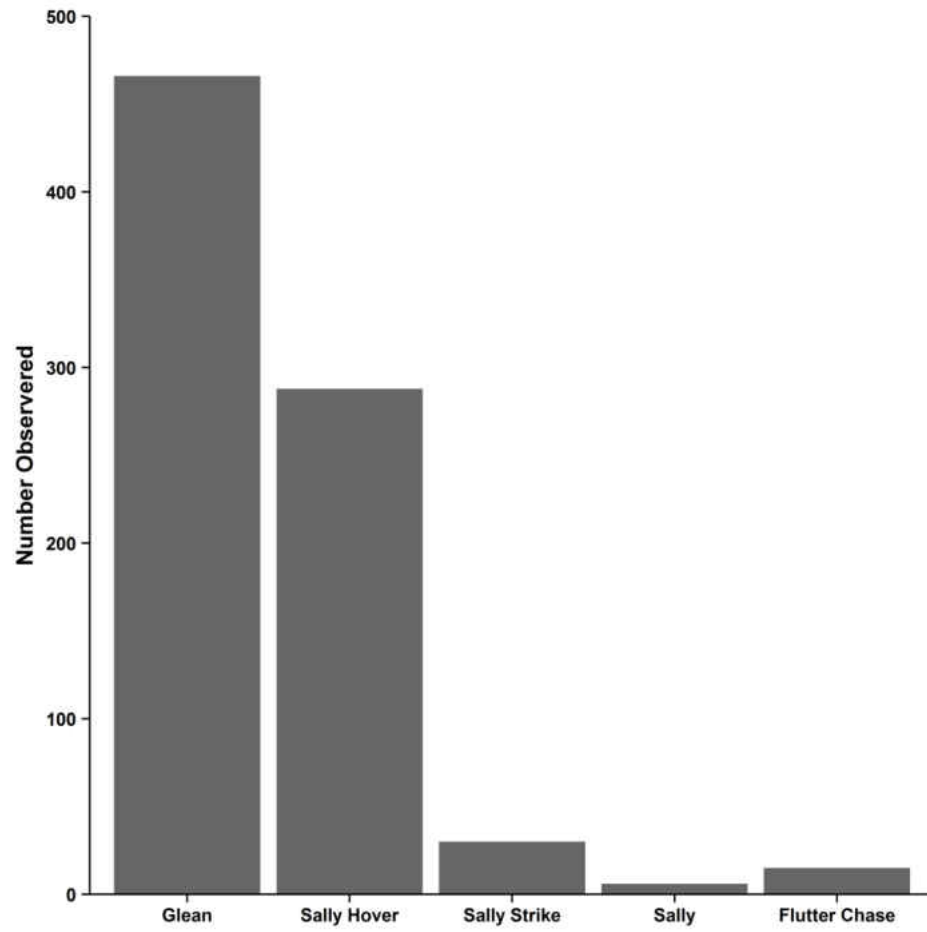


Figure 6. Total frequency of all foraging attacks observed over the two years of study.



Figure 7. Typical home range overlap shown by kinglets at Naval Live Oaks. The lines and circles represent different banded birds (black=ASY male, white=ASY female). The inner line is the core area and the outer line is the 90% kernel. The triangles represent sightings of unbanded birds.



Figure 8. Typical home range overlap shown by kinglets at Naval Live Oaks. The lines and circles represent different banded birds. The inner line is the core area and the outer line is the 90% kernel. The triangles represent sightings of unbanded birds

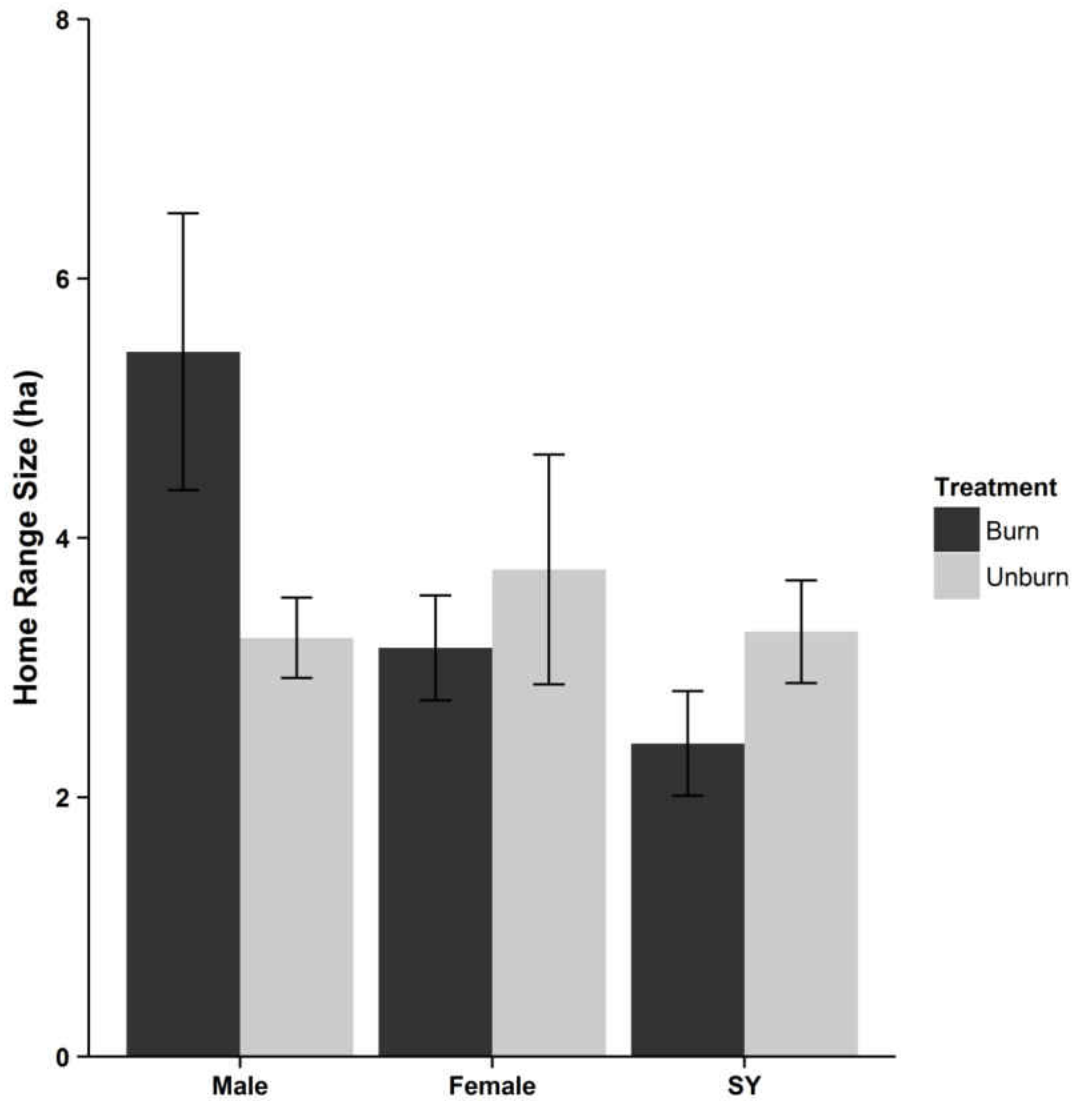


Figure 9. Overall home range size (based on 90% kernel) in hectares with standard error bars between age/sex classes and burn treatment. Groups labeled as Male and Female are ASY age-class birds.

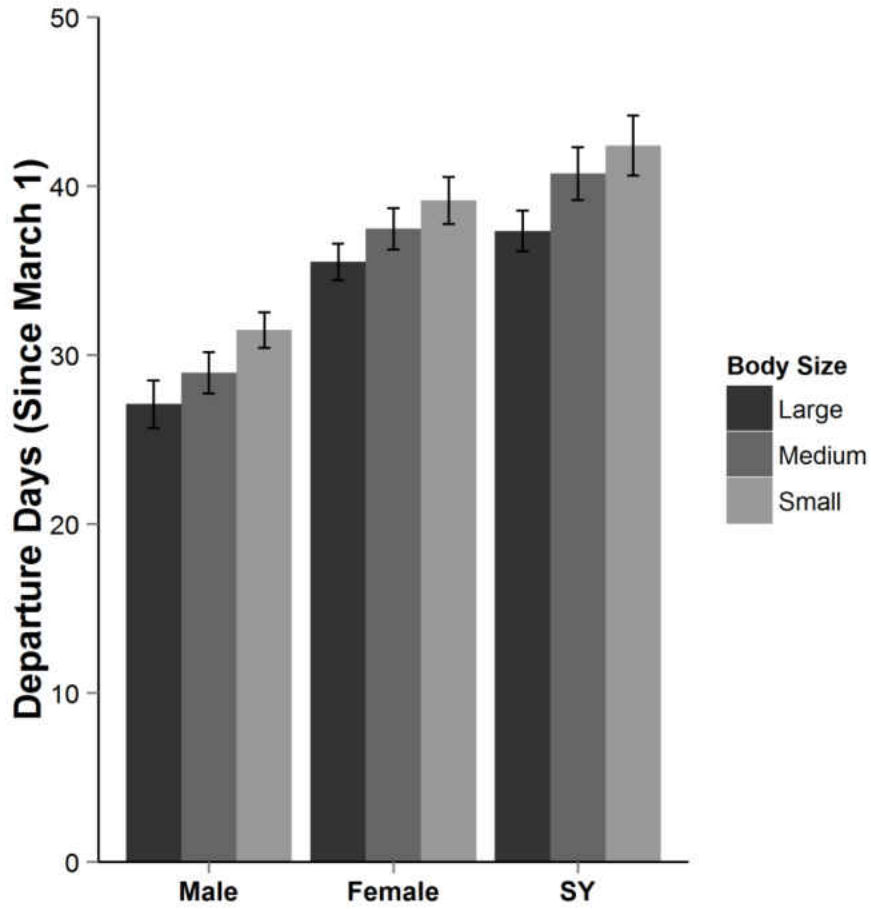


Figure 10. Relative timing of departure of each age/sex class based on structural body size scores from PC1. Groups labeled as Male and Female are ASY age-class birds.