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Avian Responses to Pine Barrens Prescribed Fire

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Avian responses to pine barrens prescribed fire

by

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ABSTRACT

Inland pine barrens support a rich community of plants and animals uniquely adapted to life in open-canopy, pyrogenic habitats. These environments also support a diverse array of birds, including critical populations of declining shrubland species. Active habitat management using prescribed fire is often necessary to maintain and restore pine barrens ecosystems. This study examines the effects of a prescribed burn on a resident pine barrens bird community at the Albany Pine Bush Preserve in East-central New York State. I used data collected during bird mist-netting to compare bird diversity, abundance, and body mass between the burn site and an adjacent site in the year before and after the burn. The post-fire burn site yielded the largest number of bird captures, but it had the lowest index of diversity. Relative abundance was analyzed for 12 species. Of these, four species were significantly more abundant and three species were significantly less abundant in the post-burn site. Differences in bird body mass varied between burned and non-burned treatments, suggesting altered food availability after the burn. Three species were significantly heavier in the post-burn site. No species were significantly less massive in this site. Given the positive responses of many bird species directly following the burn, I conclude that the short-term effects of prescribed fire do not pose a significant threat to resident bird populations in this pine barrens ecosystem. The bird community on this pyrogenic landscape demonstrates remarkable resilience and adaptation to fire disturbance.

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PREFACE

In ecology, an answer to one question simply leads to a dozen more. Traversing the path of perpetual questions is a delightfully confusing venture. I found myself beginning such a sojourn while working at the Albany Pine Bush Preserve. One question in particular struck me with such weight that I decided to pursue it in the body of text that follows: how do bird communities respond to prescribed fire as a management technique?

I by no means intend to present a complete answer to my question. Graphs, numbers, text—none can fully arrive at a complete understanding of the natural world. We like to observe the infinite complexity of nature and boil it down like maple sap in a steaming pot. Nature cannot be wholly understood with only a string of numbers or a colorful graph. Faced with such a conundrum, ecologists attempt to produce summaries and statistics that are just enough to provide a glimmer of an answer. And as always, each of these answers comes coupled with a new string of questions. Thus the march of knowledge proceeds. That may be the most important thing I learned from my honors project. Science is not the pursuit of answers. Science is the pursuit of questions. There is so much more to know, and it thrills the scientific mind to seek it. That's the fun part—the search.

Advice to future honors students

Your search can begin with an honors thesis. As an ESF honors student, you have the unique opportunity to invest in a topic which you personally choose. We are fortunate to have funding, equipment, and dedicated mentors ready to assist undergraduate research. Use the resources that are available.

Something to consider: everything takes longer than expected. Working through my analyses and collection of sources, it became quite clear that the project timeline in my head was a bit naïve. Give yourself ample time to pursue false positives, take wrong turns, and perform fruitless tests. In doing so, you will learn much more about your topic than you expected!

Something to remember: the honors thesis is what you make it. It doesn't have to be a novel contribution to your field. It doesn't have to challenge convention, open new doors, or publish in the journal *Nature*. It does, however, have to represent your best work and demonstrate a level of thinking beyond undergraduate coursework. When you give something your best, it can lead you to new places. My interests in avian communities and fire ecology led me further than I would have imagined: an internship, a summer job, a grad school position, and more.

ACKNOWLEDGEMENTS

I must begin by expressing my sincere gratitude to Bill Shields for his advice and guidance in almost every leg of this project. Bill's mentorship and advising has contributed phenomenally to my development as a scientist, thinker, and student of ecology.

I would like to thank Dr. Shannon Farrell for her assistance during the editing process and her willingness to aid with graphics in program R and Excel. I have benefitted immensely over the past two years through her teaching, leadership, and willingness to help.

Thank you to the permanent and seasonal staff of the Albany Pine Bush Preserve for contributing heavily to data collection. Special thanks to Neil Gifford, Amanda Dillon, and Dr. Steve Campbell for assisting with bird banding and experiment design.

Lastly, I would like to extend my deepest appreciation to my parents. From the bird feeder outside the kitchen window to bird research in the Sierra Nevada, they have continually supported and encouraged my exploration of the natural world.

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INTRODUCTION

Pine barrens habitats are considered one of the most threatened ecosystems in North America (Gifford, Deppen & Bried 2010; King *et al.* 2011). The inland pine barrens of the northeastern United States represent a valuable and unique assemblage of plant and animal biodiversity. Comprising fewer than 20 localities, inland pine barrens are the rarest shrubland habitat remaining in the Northeast. These declining habitats are postglacial relicts, restricted to well-drained sandy soils deposited during glacial retreats (Gifford, Deppen & Bried 2010). Most inland pine barrens occur on xeric sites with porous, acidic soils (Barnes 2003). These unique conditions foster a collection of species adapted to nutrient-poor soils and early-successional conditions. Unfortunately, the shallow soils and sparse cover of pine barrens ecosystems also make these habitats particularly attractive to development initiatives within the region. Many remaining inland pine barrens are heavily fragmented or damaged by development and fire suppression. By the year 2000, less than one-third of northeastern pine barrens continued to exist (Kurczewski 1998). Despite these pressures, inland pine barrens prominently contribute to regional biodiversity and shrubland species conservation.

Human disturbance to these ecosystems is not limited to development—fire suppression also reduces the quality of these conventionally pyrogenic systems. Wildfire alters habitat structure and resource availability. Pine barrens habitats depend on fire to maintain a successional mosaic of plant and animal communities (Gifford, Deppen & Bried 2010; King *et al.* 2011). Heterogeneity on the landscape level can be achieved by varying the frequency, severity, and extent of wildland fires (Brawn, Robinson & Thompson III 2001). Landscape variations resulting from disturbance regimes may be a

major factor increasing local biodiversity. Research in burned forest has revealed species in almost all taxa that respond positively to fire (Smucker 2005).

Pine barrens communities are adapted to wildfire disturbance. Within days of a burn, fire-resilient vegetation begins to sprout and incoming propagules colonize from adjacent habitat. Succession may continue unabated for decades until the next disturbance event. Without relatively frequent fires, pine barrens ecosystems will transition to closed-canopy forest (Bried & Gifford 2010). However, pine barrens habitats appear to have persisted with relative stability since the last retreat of the North American glaciers. The maintenance of fire on the landscape is crucial to the long-lasting nature of these otherwise ephemeral communities. Consequently, pine barrens provide a valuable and comparatively stable landscape for populations of early-successional plants and wildlife (Gifford, Deppen & Bried 2010). A history of fire suppression by humans has removed many pine barrens from their prehistoric cycle of disturbance and regeneration. Native Americans used fire on the landscape to clear land for agriculture and open forests for hunting. By the 1930's, however, fire suppression policies were well in place as a facet of forest conservation. Soon, nearly all wildland fires with low and medium intensity were quickly extinguished by fire protection organizations (Barnes 2003). Freed from natural disturbance, pine barrens community assemblages can be outcompeted by encroaching species that are not adapted for fire (Forman 1979; Elliott *et al.* 1999; King & Schlossberg 2014). Within the Albany sand plain of New York State, inland pine barrens are threatened by the encroachment of closed-canopy forest vegetation like aspen (*Populus spp.*), black cherry (*Prunus serotina*), and black locust (*Robinia pseudoacacia*).

Other species, including non-native honeysuckles (*Lonicera*), outcompete native understory vegetation.

Prescribed fire is a management technique that seeks to mimic the periodic disturbance of natural fires through controlled and well-planned burning. It has become an important habitat preservation method in many ecosystems that have historically depended on natural fire disturbance. Prescribed burns are also used to safely reduce the accumulation of fuels that build up during periods of fire-suppression. Elimination of excess fuels drastically reduces the risk of catastrophic, uncontrolled fire (Russell *et al.* 2009; King *et al.* 2011). This benefit of prescribed fire is crucially important in areas where fire-dependent habitat is fragmented and imbedded in a matrix of human development. In increasingly fragmented habitat, prescribed burning management is lauded for its ability to selectively burn predetermined tracts of land while leaving others to continue with natural succession. As such, land managers can achieve a desired mosaic of successional habitat according to the habitat preferences of the native pine barrens community (Elliott *et al.* 1999). This practice confers benefits to many pine barrens species, including shrubland birds (Brawn, Robinson & Thompson III 2001; Artman, Hutchinson & Brawn 2005; King & Schlossberg 2014).

Many early-successional avian species are currently at their lowest recorded populations (Brawn, Robinson & Thompson III 2001; Akresh 2012). These population declines stand apart from declines in other avian groups. Significant declines have affected 59% of shrubland species, while only 29% of forest-dwelling species have witnessed similar declines (Sauer *et al.*, 2008). Schlossberg and King (2007) report that half of the bird species (21 species) identified as core scrub-shrub birds in New England

have shown long or short-term declines. These declines have continued to increase in severity over the past few decades. Ailing populations of shrubland birds are linked to marked declines in early-successional habitat. In the Northeast, these habitats are maintained primarily by active management, making their associated species the most conservation-dependent group in the region (Schlossberg & King 2007; Gifford, Deppen & Bried 2010).

Conservation managers use a variety of methods to preserve habitat for early-successional plants and animals. Within the lands of the Albany Pine Bush Preserve (Albany Co., New York), inland pine barrens are maintained using an active and organized combination of selective clearing, herbicide application, mowing, and prescribed burns. Without frequently prescribed fire, this region's open pitch pine-scrub oak barrens are quickly overgrown into a thicket which gives way to dense tree oaks and invasive hardwoods (Bried & Gifford 2010). Indeed, scrub oak is notorious for its resilient ability to regenerate immediately following fire. Prescribed fire thins the scrub oak understory, reduces fuel load, and opens patches for native fire-dependent herbs. Hutchinson et al. (2005) describe evidence demonstrating an increase in small scale species richness as grasses, summer forbs, and seed banking species regenerate after fire. Burning also substantially reduces overstory and understory shrub density while facilitating prolific sprouting of low shrub species (Elliot et al.). These changes in vegetation structure following fire can be expected to affect the resident avian community. Reduced ground cover may benefit granivores and other ground-feeding species, and open canopies resulting from fire may provide more room for the maneuvers of aerial insectivores (Smucker, Dickson et al. 1995). Foliage gleaners and shrub

dependent birds may exhibit reduced populations until shrubs regrow (Artman, Hutchinson & Brawn 2005; Smucker, Hutto & Steele 2005).

Previous studies have indicated complex responses to fire within bird communities. Multiple studies suggest differing responses based on fire severity, time since fire, and total burn area (Saab & Powell 2005; Hawkins 2006). It is likely that all these factors confound the diverse results of past studies. Smucker et al. (2005) examined changes in bird abundance after a series of fires in Montana and concluded that (1) the magnitude of community change was associated with fire severity, (2) the abundance of many bird species was similar before and after the burn except at the most severely burned sample sites, and (3) differences occurred in bird communities between one year post-fire and two years post-fire. Saab & Powell (2005) reviewed available research on species responses to fire and found considerable variation between species, within a single species, and among different studies. However, appreciable trends occur based on species guild and behavior. Aerial, ground, and bark foragers demonstrate a general preference for burned areas. Ground, cavity, and canopy nesters tend to favor burned habitat more than shrub nesters. Perhaps most critical, however, is the lament of many authors that experimental work in this field is regrettably scarce (Artman, Hutchinson & Brawn 2005; Hawkins 2006). Thus, prescribed burns offer key opportunities to predict and examine the effects of fire on avian communities.

A majority of available studies investigate avian responses to high-intensity wildfire as opposed to low-intensity prescribed fire (Smucker, Hutto & Steele 2005). Unlike high-intensity wildfires, well-managed prescribed fires usually burn off ground-layer vegetation and duff without causing mortality in larger trees (Saab & Powell 2005).

Post-fire alterations to a community are largely dependent on fire severity. Thus, one might expect that birds will respond differently to prescribed fires as compared to wildfires. Past studies have addressed a large variety of fire-dependent habitats but have largely excluded rare inland pine barrens like the Albany Pine Bush. An early observational account from one pine barrens reported that the effects of a recent fire on the bird community were much less pronounced than anticipated. Brown thrashers and eastern towhees dispersed to nest on the outskirts of the burn area, while common nighthawks were found nesting where fire had cleared the ground. Pine and prairie warbler numbers were not affected, although the author estimated an increase in eastern bluebird and chipping sparrow densities (Ulmer 1926). Further research beyond observational reports is deficient for inland pine barrens ecosystems.

The effects of fire on birds are conventionally reported using estimates of species diversity, abundance, density, and biomass. These measures address community composition and relatively coarse numeric responses of local populations but fail to examine demographic effects or effects on individual physiology and fitness. Examinations of post-fire bird mass, age distribution, and demography are lacking in current literature. These qualities of a population are important considerations when reporting the effects of fire on birds. Mist netting allows investigators to collect direct measurements of mass, age, and condition from captured birds.

Two MAPS (Monitoring Avian Productivity and Survivorship) constant-effort bird banding stations exist within a native pitch pine-scrub oak ecosystem at the Albany Pine Bush Preserve. One of these stations was the site of a low-intensity prescribed burn in the early breeding season of 2014. The other bird banding site is located in a similar

habitat adjacent to the burned region. This provides an ideal opportunity to investigate the responses of several bird species to prescribed burning in this system. Specifically, my objectives were to compare (1) abundance, (2) species diversity, and (3) body mass of resident birds between the burned and unburned site and between pre- and post-burn stages. This approach aids a more complete discussion of avian responses to prescribed fire in an inland pine barrens ecosystem by providing analysis at the community, population, and organismal level.

METHODS

Study Site

The Albany Pine Bush Preserve is located on a glacial outwash sand plain between the cities of Albany and Schenectady in the Capitol District of New York State (42° 42' N, 73° 52' W). The 1,255 hectare preserve encompasses a unique ecoregion built on Aeolian parabolic sand dunes and xeric, well-drained soils. This postglacial relict contains one of the best preserved inland pine barrens remaining in the nation (Barnes 2003). In 2014, the preserve was designated as a National Natural Landmark in recognition of its commitment to conserving globally rare inland pine barrens ecosystems within the region. The preserve provides refuge for several rare plant species and a number of threatened insects, including the federally endangered Karner blue butterfly (*Lycaeides melissa samuelis*). Many declining shrub/scrub bird species are also present in significant numbers on preserve lands.

Protected areas of the pine bush are heavily fragmented by state and local highways (including Interstate 90) and urban expansion. Distinct patches of natural areas exist embedded within a matrix of commercial, residential, and agricultural land (Gifford, Deppen & Bried 2010). A large shopping mall and municipal landfill operate directly adjacent to protected lands. Native pitch pine-scrub oak barrens are also hindered by historical fire suppression and the encroachment of invasive successional vegetation. The non-native black locust (*Robinia pseudoacacia*) is a nitrogen-fixing hardwood that fundamentally alters nutrient availability in pine barrens, facilitating a community shift away from specialized xeric plant species. Historically, black locust has replaced up to 18% of pine barrens habitat in the preserve (Finton 1998). The preserve hosts ongoing

programs to restore and maintain pine barrens habitat through a variety of methods. Prescribed burning, in particular, is heavily utilized with planned frequencies and intensities to mirror natural disturbance.

The “Kings Road Barrens” MAPS station (here called KR) is divided into three separate management units. Two of these units were mowed in the spring of 2013 to reduce growth of dense woody understory plants. One of these units was subsequently treated with selective herbicide and prescribed fire in late autumn 2013. During the study, KR included a diverse patchwork of habitats undergoing different levels of restoration. The site is dominated by open-canopy scrub habitat with a diverse herbaceous layer. Grasses, scrub oak (*Quercus ilicifolia* and *Q. prinoides*), whorled loosestrife (*Lysmachia quadrifolia*), and greater prairie willow (*Salix humilis*) are present in large numbers. Seven of the twelve mist nets operated at KR were present within this habitat. Another large portion of the site is dominated by thick scrub oak and scattered pitch pine (*Pinus rigida*). This habitat, incorporating the other 5 mist nets, is bordered by a region of overgrown scrub oak (many individuals over 5 meters tall) and thick shrubs (*Rubus spp.*). The canopy of this patch is made up of pitch pine and dead standing hardwoods (*Populus spp.*). KR is bordered by paved roads to the West and South.

The “Karner Barrens West” management site and MAPS station (here called BW) was treated with a prescribed burn on June 2nd, 2014. Specific objectives of the burn included reducing one and ten-hour fuels, exposing mineral soil, and top-killing scrub oak and other woody shrub vegetation. The first day of bird banding occurred at this site just two days after the burn. During the summer of 2014, BW vegetation was dominated by a large region of low herbaceous cover and scattered pitch pine. This area is rimmed

by a belt of pitch pine and hardwood (mostly *Prunus serotina*) forest with a burned understory of scrub oak and other top-killed shrubs (*rubus spp.*, *Lonicera spp.*). The site is bordered by roadway on all sides, and a major interstate highway runs along the southern fringe. A portion of land in the northeast corner of the site was not burned. This area, bordering three of twelve mist nets, was dominated by grasses and shrubs (*Vaccinium pallidum*, *Lonicera spp.*).

Avian Sampling and Analysis

Constant effort mist netting offers the opportunity to gather information on relative bird abundance and diversity by creating a controlled random sample of captured birds (Nur, Geupel & Ballard 2000; Bibby *et al.* 2000). Constant-effort mist nets are employed across the United States as a standardized way to measure bird population trends (Osenkowski, Paton & Kraus 2012). In this study, Albany Pine Bush staff and I captured and processed birds following standard MAPS protocol (see Desante *et al.* 2014). We opened twelve meter, 30/32 millimeter mesh, four-tiered nylon mist nets one-half hour before sunrise and operated for at least six hours during a sampling period. Locations of the twelve mist nets were kept constant during both years of the study. Both stations operated seven times throughout the course of the breeding season. We fitted captured birds with United States Geological Survey aluminum bands. We identified the age and sex of each individual using plumage, breeding condition, body characteristics, and molt limits. The body mass of each individual was recorded in grams.

Recaptured birds were only counted for their first capture event in that season. Descriptive statistics were calculated for all variables. I used Chi-squared goodness-of-fit tests to compare bird abundance between the burned and unburned site for each species.

Comparative abundance was further analyzed using calculated percentages of each species frequency for the two sites in 2014. This approach controls for differences in capture rates to examine the abundance of each species relative to the community as a whole. I calculated community diversity for each site pre- and post-burn using two common diversity indices ($1/D$ and H) and evenness (E). While the Simpson's index (D) is recommended for systems with a wealth of rarely recorded species, the Shannon Weiner index (H) appears to be most appropriate for situations where a community is dominated by a few abundant species (Nagendra 2002). Because both scenarios were present, it was appropriate to utilize both diversity measures. I used a two-way analysis of variance (ANOVA) and Tukey comparisons to compare bird mass between sites and years. I examined main effects of the factors 'site' and 'age' using the response variable 'mass.' Two-sample T-tests were used to compare mass differences between sites for age groups within a species.

Vegetation Methods

I sampled vegetation characteristics for both sites in late July of 2014. I developed sampling plots within the vicinity of mist net placements using ArcGIS. The vegetation sampling area included all existing habitat within 100 meters of a mist net. After digitally laying a ten-by-ten meter grid over the area, I selected ten individual plots for each site using a random number generator. Coordinates of each plot were uploaded onto a Trimble® GIS unit.

Within a plot, I recorded the species, frequency, and diameter at breast height (DBH) of each tree. I measured density of woody shrubs (DBH < 5.0 cm, height > 1 m) using three 1.8 meter transects spanning each plot. I visually estimated percent ground

and overstory cover at 13 points distributed evenly across the plot. Vertical vegetation structure was quantified as the percent coverage of vegetation in four strata: 0-1 m, 1-3 m, 3-5 m, and >5 m (Campbell *et al.* 2012). I then calculated total basal area, stem density, and percent cover for both bird banding locations.

RESULTS

Avian Community

We captured 1,085 birds representing 51 species over the course of the study. The fewest captures occurred in KR 2013, and the greatest number of captures occurred after the prescribed burn in BW 2014. Productivity was high in both sites during 2014; over half of the birds captured had hatched during that season (Figure 1.). Gray catbirds were the most commonly captured species, with 225 individuals recorded over the two years. Seven other species had over 50 captures, including the American robin, field sparrow, Baltimore oriole, common yellowthroat, prairie warbler, black-capped chickadee, and song sparrow. While the gray catbird was the most captured species in three of the four samples, American robins were the most frequently captured species in the post-fire burn area.

Eleven species were captured frequently enough to include in a comparison of abundance between the two sites in 2014 (Table 1). Of these, 3 species were significantly more common in the burn site and 3 species were significantly more abundant in the non-burned site ($\alpha=0.05$). Chipping sparrows were recorded twice as often in the burn site, but this difference was not considered significant. Comparisons were also made for BW in the year before and the year after the burn (Table 2). American robins and chipping sparrows were captured significantly more frequently after the burn. However, significantly fewer common yellowthroats were captured after the burn. Prairie warblers were also considerably less abundant after the burn, although this difference was not significant. The eastern bluebird, although omitted from analysis due to low sample size,

was captured much more frequently in the burn site than in any other site (see Appendix A).

Seven species were recorded as contributing to >5% of the total sample for at least one site (Table 3). American robins and black-capped chickadees had much higher relative abundance in the burn site. Baltimore orioles, common yellowthroats, and song sparrows were considerably less abundant in the burn site. Similar trends become clear when examining the productivity of these two sites. Productivity was measured as the proportion of hatch-year birds in a population at a given site. American robins, black-capped chickadees, and gray catbirds had a higher proportion of hatch-year birds in the burn site. Baltimore orioles, common yellowthroats, field sparrows, and song sparrows had a lower productivity proportion in the burn site.

The number of species captured was similar between sites and years. Simpson's index of diversity ($1/D$) sums the squared proportions of each species' frequency. The highest value occurred in KR 2014; the lowest value occurred in the burn site BW 2014 (Table 4). Equitability (E) expresses evenness as diversity ($1/D$) over maximum diversity (n). This value indicates that the post-burn area had a relatively uneven capture rate between species. Gray catbirds and American robins dominated this sample. The Shannon Weiner index (H) sums the absolute value of each species proportion multiplied by the natural log of the proportion. Again, KR 2014 had the highest diversity and BW 2014 had the lowest diversity. Sample evenness, measured as H divided by the natural log of n , was lowest in the post-burn site.

Mean body mass and standard error were calculated for all birds with samples sizes over 15 captures (Table 5). ANOVA tests revealed no significant interactions

between body mass, site, and age. Gray catbirds were significantly heavier in the burn site than in the unburned site (Figure 2). On average, gray catbirds captured in the burn site were 0.41 to 1.88 grams heavier than birds captured in non-burned site (95% confidence interval). There was no significant difference in mean mass between ages for this species. No other species demonstrated significant main effects of site. Post hoc analysis revealed that hatch-year American robins were significantly heavier in burn site than the adjacent site ($p=0.024$). Likewise, hatch-year chipping sparrows were also heavier in the burn site ($p=0.42$). Other species (e.g. Baltimore orioles, common yellowthroat) display a trend of heavier birds in the burn site, although these comparisons were not significant (Figure 3). No species were significantly heavier in adjacent site when compared to the burn site. Species like song sparrows and field sparrows show no consistent trends between bird mass, age, and site in the year of the burn (Figure 4).

I also compared body mass between BW 2013 and 2014; the year before and the year directly following the fire. There were no significant main effects of site on mean mass. However, after-hatch-year American robins were heavier in 2013 than in 2014 ($p=0.038$). This trend is not seen when comparing mean mass for this species between sampling years at the site adjacent to the burn ($p=0.292$). Gray catbirds were the only species that tended to be heavier in BW in the year of the burn, but this difference was not significant. An opposite yet insignificant trend is apparent in black-capped chickadees. The majority of species analyzed did not demonstrate trends in mean mass between years: Baltimore oriole, field sparrow, prairie warbler, song sparrow (Figure 5). Finally, ANOVAs were conducted to compare mean mass of birds between the two sites

in 2013, the year before the burn. There were no significant differences in mean mass between sites for any species included in the analysis.

Vegetation Summary

Vegetation sampling began at BW 45 days after the prescribed burn. In that time, a rich herbaceous layer had rapidly regrown to an average ground cover of 64%. About one quarter of observation points (n=130) had an estimated 100% ground cover. Canopy cover at BW was estimated at 21%. Tree basal area was calculated to be 81 square meters per hectare. The adjacent site (KR) had an average ground cover of 73% and an average canopy cover of 16%. Tree basal area at KR was estimated to be 107 square meters per hectare. Vertical vegetation structure was similar between sites at each of the four height classes. Vegetation cover was least from three to five meters above the ground greatest from zero to one meter above the ground. Although the two sites had different mean ground cover percentages, they had equal median ground cover values of 80%.

The unburned site had a significantly greater shrub stem density (~87200 stems per hectare) compared to the burn site (~29900 stems per hectare). However, the majority of standing stems at the unburned site were dead (58.6% of total). This proportion is due to scrub oak herbicide management at the site in the previous year. 37.1% of standing shrub stems were dead at the burn site. Both live and dead stems were recorded within the boundaries of the burn; only live stems were found in areas missed by the fire. At both sites, the shrub zone was dominated by scrub oak. Other shrubs included sapling hardwoods, honeysuckle, and hazelnut (*Corylus spp.*).

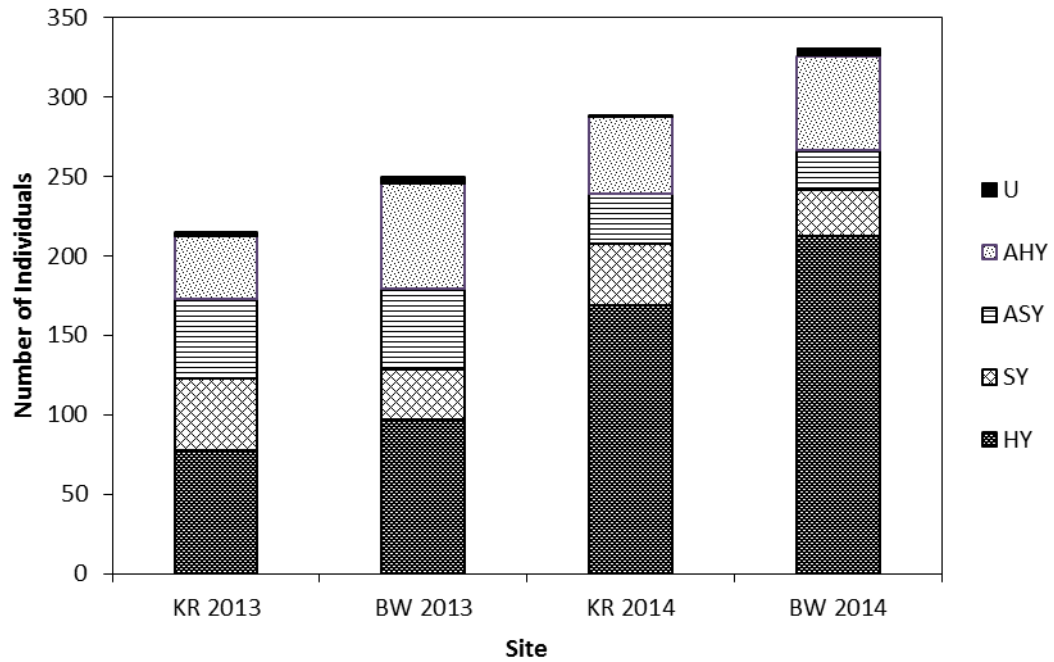


Figure 1. Frequency of bird captures separated by age and site. The post-burn location (BW 2014) yielded the highest number of individuals. In both sites, 2014 experienced a higher proportion of hatch-year birds than 2013.

Table 1. Comparison of species abundance between KR and BW in 2014 ($\alpha=0.05$).

| Species | n | χ^2 test stat. | p-value |
|----------------|----------|---------------------------------------|----------------|
| AMRO | 103 | 79.64 | <0.001* |
| BAOR | 41 | 23.36 | <0.001† |
| BCCH | 29 | 13.0 | <0.001* |
| CHSP | 23 | 16.78 | 0.061 |
| COYE | 33 | 8.59 | 0.003† |
| EATO | 15 | 3.80 | 0.796 |
| FISP | 50 | 0.79 | 0.572 |
| GRCA | 112 | 0.72 | 0.571 |
| NOMO | 16 | 11.41 | 0.046* |
| PRAW | 18 | 3.06 | 0.637 |
| SOSP | 36 | 9.50 | 0.008† |

*Species abundance significantly greater in the post-burn site (3 species)

†Species abundance significantly lower in the post-burn site (3 species)

Table 2. Comparison of species abundance between 2013 and 2014 in BW ($\alpha=0.05$).

| Species | n | χ^2 test stat. | p-value |
|----------------|----------|---------------------------------------|----------------|
| AMRO | 100 | 49.00 | <0.001* |
| BAOR | 19 | 1.32 | 0.251 |
| BCCH | 37 | 3.27 | 0.071 |
| CEDW | 16 | 0.25 | 0.617 |
| CHSP | 16 | 16 | <0.001* |
| COYE | 26 | 3.85 | 0.050† |
| EATO | 23 | 2.13 | 0.144 |
| FISP | 44 | 0.09 | 0.763 |
| GRCA | 121 | 0.07 | 0.785 |
| PRAW | 24 | 2.67 | 0.102 |
| SOSP | 22 | 0.18 | 0.670 |

*Species abundance significantly greater in BW 2014 (2 species)

†Species abundance significantly lower in BW 2014 (1 species)

Table 3. Species frequency expressed as percent of total captures followed by percent of hatch-year birds (total frequency: HY frequency).

| Species | KR 2014 | BW 2014 |
|----------------|-----------------|-----------------|
| AMRO | 6.23% ; 77.78% | 25.68% ; 85.88% |
| BAOR | 11.76% ; 76.47% | 2.11% ; 57.14% |
| BCCH | 1.73% ; 20.00% | 7.25% ; 54.17% |
| COYE | 8.65% ; 64.00% | 2.24% ; 25.00% |
| FISP | 9.34% ; 51.85% | 6.95% ; 34.78% |
| GRCA | 18.34% ; 60.38% | 17.82% ; 74.58% |
| SOSP | 9.00% ; 88.46% | 3.02% ; 60.00% |

Table 4. Diversity indices by sampling location.

| Diversity Index | KR 2013 | BW 2013 | KR 2014 | BW 2014 (post-burn) |
|------------------------------------|----------------|----------------|----------------|--------------------------------|
| Number of Species Captured (n) | 30 | 33 | 35 | 33 |
| Simpon's Index of Diversity (1/D) | 13.152 | 12.942 | 13.994 | 9.867 |
| Equitability (E): | 0.438 | 0.392 | 0.400 | 0.299 |
| Shannon-Weiner Diversity Index (H) | 2.628 | 2.626 | 2.782 | 2.568 |
| Evenness (E): | 0.773 | 0.751 | 0.782 | 0.736 |

Table 5. Mean body mass calculated for selected species, divided into hatch-year and after-hatch-year ages groups (mean mass \pm S.E.).*

| <u>Species</u> | | <u>Mean Mass (g\pmSE)</u> | | | |
|----------------|------|--|------------------|------------------|------------------------|
| | | KR 2013 | BW 2013 | KR 2014 | BW 2014 (post-burn) |
| <u>AMRO</u> | HY: | 75.14 \pm 1.51 | 76.46 \pm 1.46 | 72.27 \pm 1.29 | 75.73 \pm 0.54 |
| | AHY: | 76.92 \pm 1.98 | 80.72 \pm 1.35 | 78.00 \pm 1.30 | 76.27 \pm 1.01 |
| <u>BAOR</u> | HY: | 31.68 \pm 0.72 | 32.87 \pm 0.32 | 32.20 \pm 0.28 | 33.08 \pm 0.75 |
| | AHY: | 33.36 \pm 0.45 | 33.89 \pm 0.83 | 32.18 \pm 0.93 | 32.60 \pm 0.50 |
| <u>BCCH</u> | HY: | 10.45 \pm 0.15 | 10.90 \pm 0.36 | ---- | 10.44 \pm 0.18 |
| | AHY: | 10.47 \pm 0.50 | 10.72 \pm 0.30 | 10.75 \pm 0.12 | 10.64 \pm 0.14 |
| <u>CHSP</u> | HY: | ---- | ---- | 10.87 \pm 0.23 | 11.72 \pm 0.21 |
| | AHY: | ---- | ---- | 12.03 \pm 0.54 | 12.17 \pm 0.29 |
| <u>COYE</u> | HY: | 9.67 \pm 0.35 | 10.87 \pm 0.37 | 9.53 \pm 0.19 | 9.75 \pm 0.15 |
| | AHY: | 9.35 \pm 0.50 | 10.10 \pm 0.23 | 9.71 \pm 0.18 | 10.15 \pm 0.21 |
| <u>EATO</u> | HY: | 37.80 \pm 0.36 | 37.30 \pm 0.50 | 37.82 \pm 2.10 | ---- |
| | AHY: | 42.15 \pm 1.48 | 41.0 \pm 0.90 | 42.05 \pm 2.95 | 39.87 \pm 1.01 |
| <u>FISP</u> | HY: | 11.97 \pm 0.46 | 11.63 \pm 0.39 | 11.80 \pm 0.20 | 12.00 \pm 0.26 |
| | AHY: | 12.92 \pm 0.30 | 12.42 \pm 0.26 | 12.63 \pm 0.23 | 12.28 \pm 0.18 |
| <u>GRCA</u> | HY: | 35.17 \pm 0.44 | 34.56 \pm 0.30 | 34.30 \pm 0.34 | 35.27 \pm 0.27 |
| | AHY: | 36.02 \pm 0.44 | 35.71 \pm 0.40 | 34.54 \pm 0.48 | 36.09 \pm 0.57 |
| <u>NOMO</u> | HY: | ---- | ---- | 45.55 \pm 1.85 | 45.89 \pm 0.89 |
| | AHY: | ---- | ---- | 48.20 \pm 1.60 | 46.65 \pm 0.25 |
| <u>PRAW</u> | HY: | ---- | 7.73 \pm 0.29 | ---- | 7.50 \pm 0.20 |
| | AHY: | 7.95 \pm 0.27 | 7.72 \pm 0.12 | 7.61 \pm 0.12 | 7.62 \pm 0.31 |
| <u>SOSP</u> | HY: | 18.88 \pm 0.52 | ---- | 18.67 \pm 0.29 | 18.76 \pm 0.31 |
| | AHY: | ---- | 19.94 \pm 0.65 | 20.80 \pm 0.15 | 20.50 \pm 0.91 |

*Blank fields do not have enough records for relevant calculations.

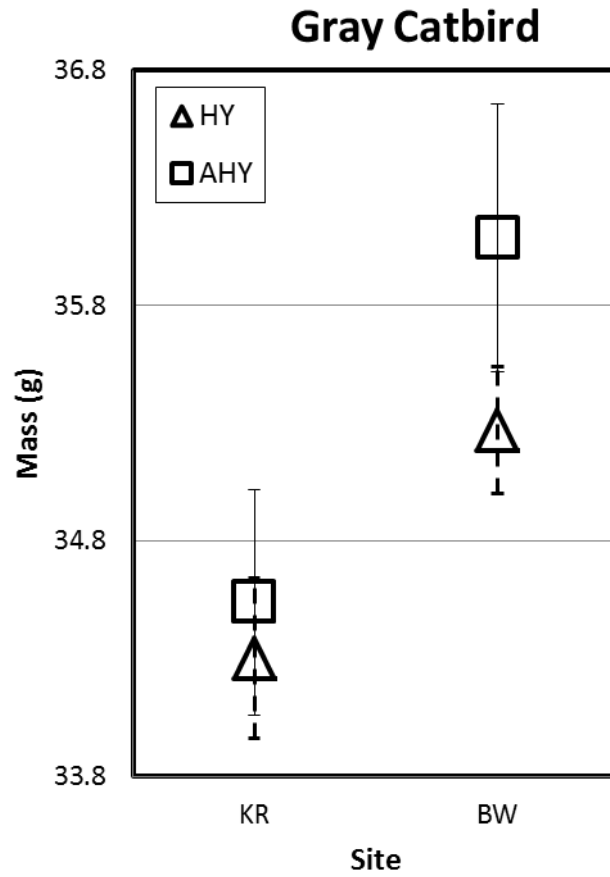


Figure 2. Mean mass with standard error bars for gray catbirds captured from both sites in 2014. Site is a significant main effect on mass ($p=0.003$). Birds captured at the burn site (BW) averaged 1.9 grams to 0.4 grams heavier than birds captured at KR (95% confidence interval).

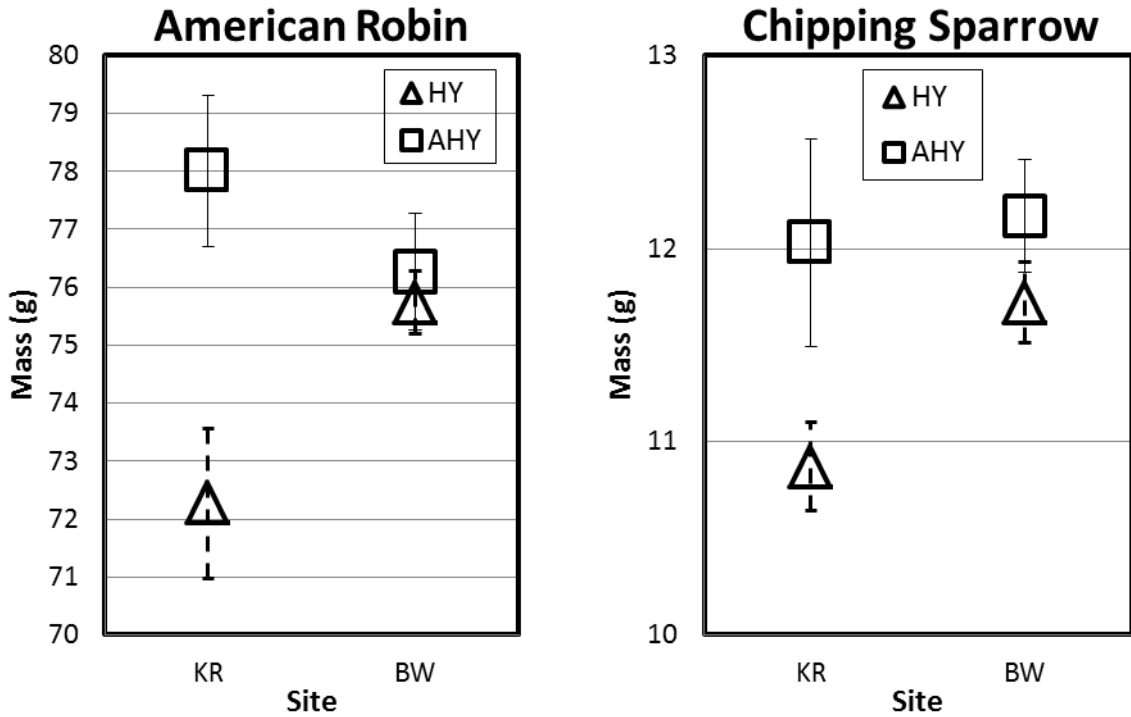


Figure 3. Mean body mass with standard error bars for selected species in 2014. There were no significant interactions between variables. Although individuals from the burn site tended to be more massive, site was not a significant main effect on mass. Hatch-year American robins and hatch-year chipping sparrows were significantly heavier in the burn site than the adjacent site ($p=0.024$; $p=0.042$). Mean mass was significantly different between ages in American robins ($p=0.035$) and nearly significant in chipping sparrows ($p=0.051$).

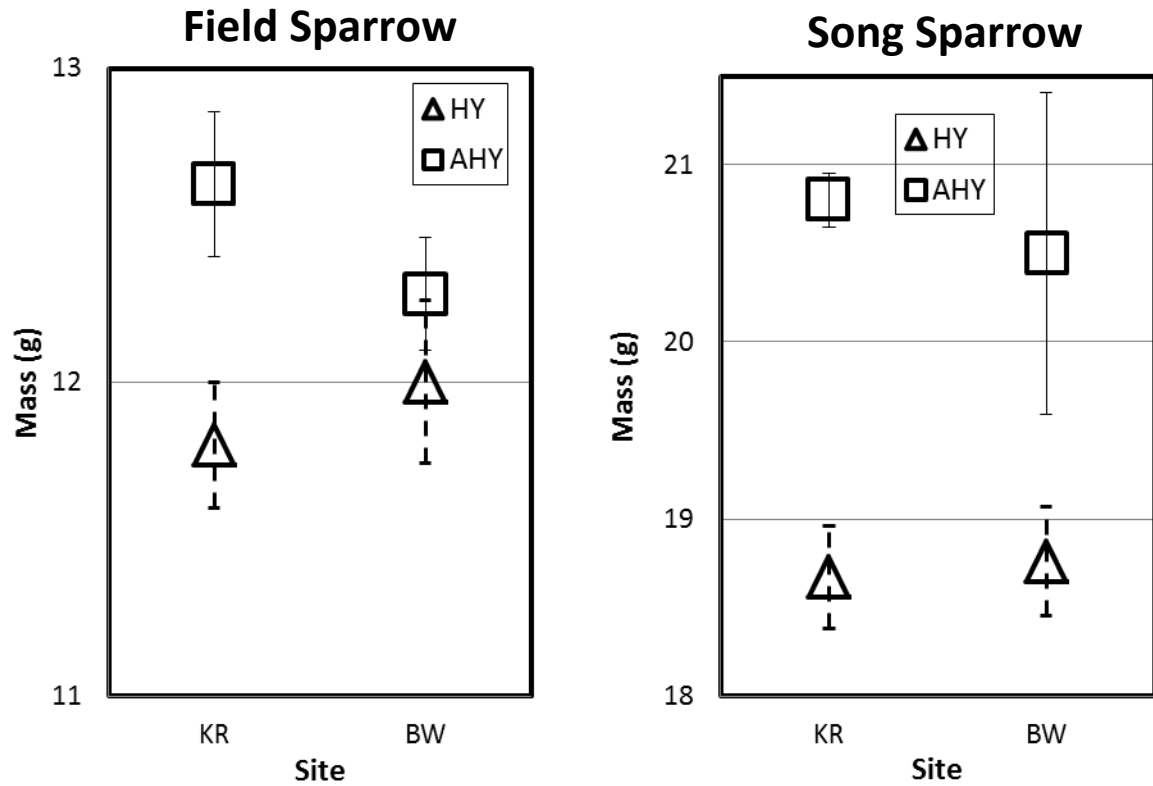


Figure 4. Mean body mass with standard error bars for two species in 2014 which demonstrated no discernable trend in mean mass between sites. For both species, the only significant main effects were age (FISP, $p=0.010$; SOS, $p=0.002$).

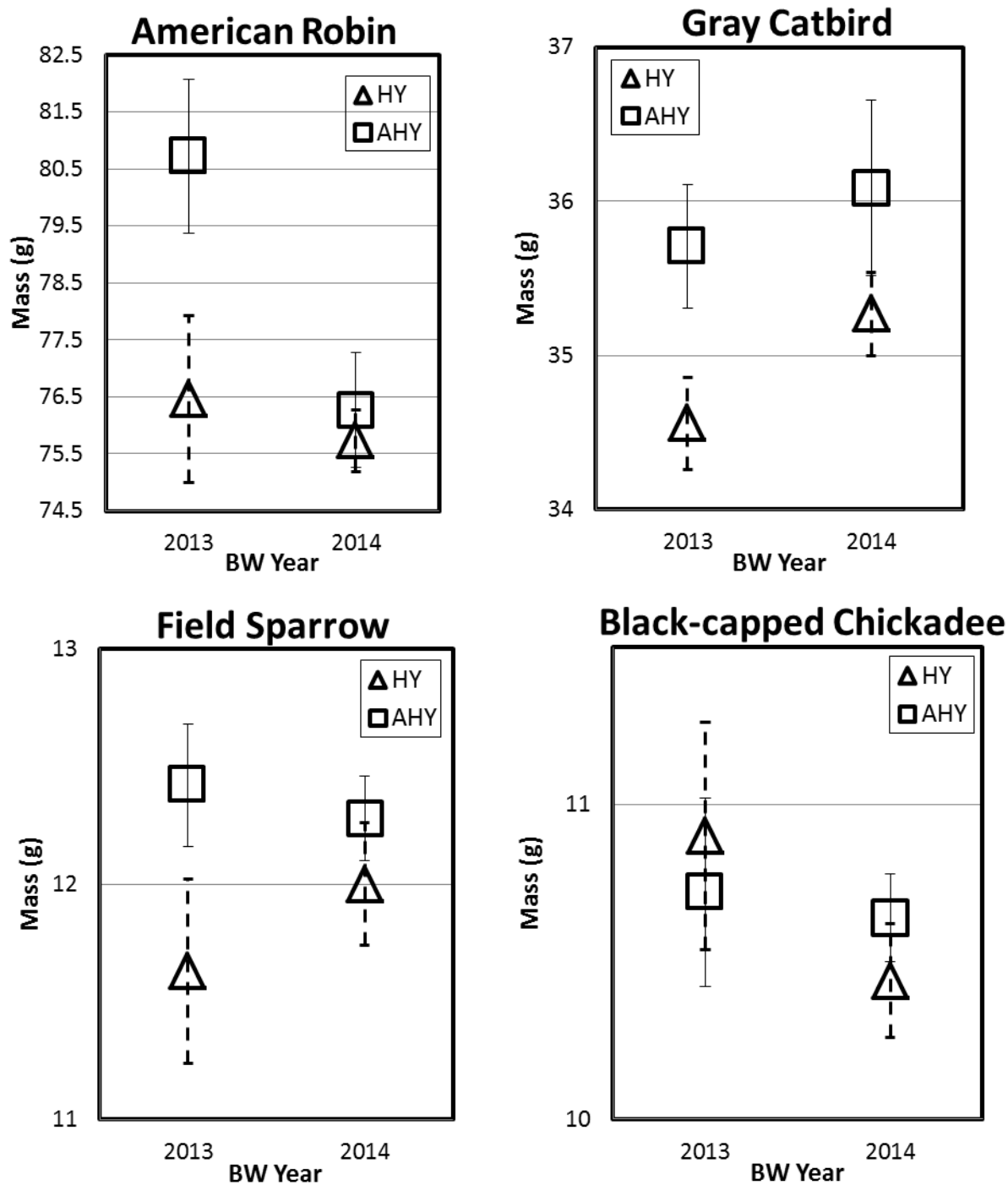


Figure 5. Mean body mass with standard error bars for selected species in BW 2013 and 2014. There were no significant interactions between variables. Three trends are depicted: birds averaged heavier in 2013 (AMRO, BCCH), birds averaged heavier in 2014 (GRCA), and mixed results (FISP). Year was not a significant main effect on bird mass. However, after-hatch-year American robins were significantly heavier in 2013 ($p=0.038$). Age was a significant source of variation for gray catbirds ($p=0.012$).

DISCUSSION

Avian Diversity

Previous studies indicate increases in avian species richness after fire (Bock & Lynch 1970; Apfelbaum & Haney 1981; Hutto 1995). These studies cite the total number of species observed at a research site without incorporating calculations of species evenness. The diversity indices used here incorporate richness and evenness into a final value. Thus, differences in diversity can also provide information on which species dominate the community. The low diversity indices for the post-burn site were strongly influenced by abundant populations of certain species at the site. This factor is also reflected in the low evenness/equitability values for the site. These results suggest that certain fire-adapted species may dominate post-fire bird communities in the first weeks after a burn. American robins were particularly abundant in the post-burn location.

Many environmental factors sway calculations of community diversity. Although both study sites are located in similar habitat, they accommodate differing microhabitats and border divergent vegetation communities. Avian diversity at BW may be limited by the site's proximity to major roadways. KR extends to roads on two sides, but the site also borders a more continuous natural area. This may explain some of the sites more peculiar captures, including yellow-throated vireos, wood warbler species, and orchard orioles.

The mist netting method provides a limited view into the avian diversity present at each site. By providing a constant-effort random sample, mist netting increases the chances of detecting well-hidden or non-vocal birds, including fledglings (Bibby *et al.* 2000). In a sense, this method exchanges the observer bias associate with conventional

point counts for a 'net bias'. Birds that forage in foliage or on the ground are more likely to be sampled than birds that forage on the wing. While useful for making comparisons, this method underestimates species richness. Multiple species of aerial insectivores and raptors were common at both sites, but these species are difficult to sample with passive mist netting. Individuals of these species are not represented in the data for this study.

Comparative Abundance

Significantly more birds were captured at both sites in 2014 than in 2013, despite consistency in sampling effort. At the Albany Pine Bush locality, many factors exist as possible manipulators of bird population trends: weather, seasonal changes, management practices, and food availability. Short-term population fluctuations may also be attributed to stochastic population trends; avian populations are known to fluctuate from year to year across large spatial scales and at small-scale breeding localities (Keitt & Stanley 1998). Productivity was also relatively higher for both sites during the 2014 breeding season. Measures of productivity are useful indicators of breeding success in the catchment area sampled by mist nets (Nur, Geupel & Ballard 2000). In 2013 65% of sampled birds hatched in the same year as capture, while 79% of birds were hatch-year in 2014. The number of hatch-year birds captured in 2014 was more than double that of the previous year. Age structure did not differ between the two sites in 2014; the prescribed burn did not seem to cause significant alteration to population age distributions. Further analysis of past bird-banding/monitoring datasets may provide the information needed to make informed hypotheses regarding the fluctuations witnessed over the two years of this study.

The number of birds captured in post-burn BW was 14% more than the number of birds captured in KR 2014 and 32% more than the number of individuals captured in BW 2013. Increases in abundance were apparent in American robins, black-capped chickadees, chipping sparrows, and northern mockingbirds. Baltimore orioles, common yellowthroats, prairie warblers and song sparrows were less abundant in the burn site. Foraging and nest guild designations for each of these species follows those of Maurer et al. (1981) and Saab & Powell (2005). Larger ground insectivore species (American robin and northern mockingbird) were more abundant after the burn. With the exception of black-capped chickadees, smaller foliage gleaners like common yellowthroats and prairie warblers were less abundant in the burn site. Transitions in post-fire bird communities to heavier, ground-foraging bird species have been described in other study systems (Apfelbaum & Haney 1981; Saab & Powell 2005). Baltimore orioles, a species preferring tall, open deciduous forest, were also less abundant in the burn site.

Decreases in shrub-nesting birds following fire have been evidenced in past studies (Saab & Powell 2005). Indeed, certain shrub-nesters were less abundant in the post-burn site. Others, like the gray catbird and chipping sparrow, did not experience declines in the year following the fire. The abundance of bird captures at the post-burn site suggests a notable presence of transient non-territory holding birds, sometimes called “floaters.” Site-faithful territory holders within the catchment area are more likely to remain within a defined area. Floaters, however, move from place to place as food is available (Nur, Geupel & Ballard 2000). The burn may have altered the established territories of after-hatch-year birds in a way that facilitated use of the site by foraging flocks of hatch-year birds and other transient individuals. The demographic shift

associated with this phenomenon would be expected to increase overall abundance and the number of hatch-year birds captured at the site.

Previous research in inland pine barrens have concluded that active management at these sites provides habitat for scrub-shrub bird species while causing slight reductions in local forest dwelling species (King *et al.* 2011). In such a case, management objectives seek to address the needs of high-priority shrubland birds. Gifford, Deppen & Breid (2010) suggest that the prairie warbler is the strongest avian indicator of ecosystem health at the Albany Pine Bush Preserve. This species is included among seven other avian indicators of early-successional shrubland habitat in the region (Bried *et al.* 2011). While reductions of obligate forest birds were apparent in the burn site, many shrubland indicator species did not experience short-term increases in abundance. Some, like the common yellowthroat and prairie warbler, were considerable less common directly following the burn. These birds were likely displaced to neighboring habitat, including the site KR. Despite the effort associated with relocation, shrubland birds often show remarkable resilience after disturbance. Displaced prairie warblers enjoy breeding success that is similar to resident conspecifics in their new habitat patch. Newly-created habitat is colonized in the years following disturbance by second-year birds, while site-faithful older birds maintain territories in more mature habitat (Akresh 2012). It is likely that shrubland birds will increase in abundance at the burn site in subsequent years. Some indicator species, including chipping sparrows, field sparrows, and gray catbirds, did not show decreases in abundance following the fire.

Differences in abundance between the sites mirror differences in productivity for each species. Of the species that increased in abundance after the burn, American robins

and black-capped chickadees also experienced a rise in productivity at the burn site. Gray catbirds, although similar in abundance between sites, also had higher productivity in the post-burn site. Similarly, species that were less abundant in the burn site (Baltimore oriole, common yellowthroat, and song sparrow) experienced greater productivity in the adjacent site. This suggests that hatch-year birds make up a significant proportion of breeding season abundance calculations. Measurements of abundance depend strongly on the success and failure of breeding attempts.

Bird Mass

Hatch-year American robins and chipping sparrows were heavier in the post-burn site than in the adjacent site. Both age groups of gray catbird were also heavier at this site. No species or age groups were significantly heavier in the adjacent site during 2014. These differences in average mass are not due to reduced abundance at the burn site; American robins, chipping sparrows, and gray catbirds were actually more abundant in the burn site. This indicates that more food was available for ground-foraging species after the burn in BW.

Survival and health of offspring is positively correlated with food availability during the breeding season (Martin 1987). Young American robins, chipping sparrows, and gray catbirds depend primarily on insects during the summer. In fact, insectivorous diets characterize the majority of species present in post-fire bird communities (Hutto 1995). Insect mortality after a fire is related to fire severity, degree of exposure, and mobility of the insect. However, insect populations are known to rapidly increase as vegetation regenerates. Much of the catchment area sampled by mist nets in the burn site was dominated by dense regenerating growth. Post-burn flora can support impressive

populations of recolonizing insects. It is speculated that vegetation success results from the temporary herbivore release of re-sprouting plants following insect reductions after fire (Swengel 2001). To some degree, this phenomenon may culminate in abundant insect populations shortly after a fire passes—and more food for breeding/fledging birds.

Ecology and Management

Avian community responses to fire are often mixed and difficult to discern. Results and trends are complicated by time, region, and environmental factors (Saab & Powell 2005). Severe differences in results have even been recorded for members of the same species. American robins show severe variation in responses to fire in the Western United States; studies have cited both more and less abundance in burned plots (Smucker, Hutto & Steele 2005). Most studies have dealt with unpredictable wildfire, making it nearly impossible to gather before-and-after bird community data. Thus, prescribed fires should be treated as valuable opportunities for controlled observations of bird responses.

Bird communities are known to respond differently to various fire intensities. Smucker, Hutto & Steele (2005) demonstrated that American robins and chipping sparrows increased in abundance only following low to moderately severity burns in Montana. The 2014 burn in BW represented a relatively low intensity, controlled fire. This management method can be expected to provide benefits to ground-foraging and open canopy birds while mitigating negative effects on shrub-nesting and foliage gleaning birds. These results of the BW prescribed burn evidence the benefit of maintaining prescribed fires at controlled levels in relatively small management units. Compared to large-scale fires (like wildfires in the Western U.S.), small prescribed burns contribute to a landscape mosaic of early to mid-successional shrub habitat. Smaller

scales can also be expected to increase recolonization rates of insects, thereby increasing food supply for birds. Such considerations are especially important when managing a constricted habitat imbedded in an urban matrix.

CONCLUSIONS

Inland pine barrens support a rich avian community inhabiting a fire-dependent ecosystem. Pitch pine-scrub oak barrens, like those at the Albany Pine Bush Preserve, offer important habitats for many scrub-shrub birds. Prescribed burning is useful for the management and maintenance of this unique ecosystem. However, it is important to consider the effects of this practice on the avian community.

The small scale, low intensity prescribed fire at site BW had diverse effects on resident birds. Compared to an adjacent and similar habitat patch, most bird species did not show significant differences in abundance. Heavy-bodied ground foragers tended to be more abundant in the burn site. American robins were the most common bird species after the burn; hatch-year robins were particularly abundant. Small foliage-gleaning insectivores, like prairie warblers, were less abundant in the burn zone. Differences in abundance reflected differences in productivity between sites. The burn zone had a smaller index of diversity than the other sites. This suggests that the post-fire abundance of certain species disrupted the evenness and equitability of the bird community. Certain age classes of American robins, gray catbirds, and chipping sparrows were significantly heavier in the burn site.

More research is necessary to track the long-term effects of the prescribed burn at BW. This study presents preliminary results offering conclusions about short-term avian responses to prescribed fire. Previous authors note a need for studies in the months immediately following fire (Lyon *et al.* 1978; Apfelbaum & Haney 1981). However, these data can be enhanced by continuing studies into subsequent years. Decreases in certain species following the prescribed burning may be followed by marked increases in

those species in the years following fire. The effects of shrubland management, which are varied in the short-term, are quite beneficial to early-successional species in the long-term (Akresh 2012). One might expect increases in foliage-gleaning and shrub-nesting bird populations at BW within the next few years. The inland pine barren avian community present at the Albany Pine Bush is remarkably resilient to ecosystem restoration by prescribed fire. The adaptations of birds and other taxa present on this ever-shifting mosaic evidence the remarkable tenacity of this imperiled ecosystem.

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APPENDICES

Appendix A. Capture records (BBL code) for each species by site.

| Species | KRB 2013 | KBW 2013 | KRB 2014 | KBW 2014 |
|---------|----------|----------|----------|----------|
| AMGO | 6 | 6 | 3 | 0 |
| AMRO | 14 | 15 | 18 | 85 |
| AMWO | 0 | 1 | 0 | 0 |
| BAOR | 18 | 12 | 34 | 7 |
| BBCU | 0 | 2 | 2 | 3 |
| BCCH | 9 | 13 | 5 | 24 |
| BHCO | 0 | 2 | 0 | 2 |
| BLJA | 0 | 4 | 5 | 2 |
| BRTH | 3 | 2 | 6 | 1 |
| BWWA | 1 | 0 | 0 | 0 |
| CARW | 0 | 0 | 2 | 0 |
| CEDW | 11 | 7 | 4 | 9 |
| CHSP | 2 | 0 | 7 | 16 |
| COGR | 0 | 1 | 5 | 1 |
| COYE | 11 | 18 | 25 | 8 |
| CSWA | 0 | 9 | 3 | 2 |
| DOWO | 6 | 0 | 3 | 0 |
| EABL | 1 | 3 | 4 | 10 |
| EAKI | 0 | 0 | 0 | 2 |
| EAPH | 4 | 5 | 4 | 5 |
| EATO | 10 | 15 | 7 | 8 |
| EAWP | 0 | 0 | 0 | 1 |
| FISP | 13 | 21 | 27 | 23 |
| GRCA | 51 | 62 | 53 | 59 |
| HAWO | 0 | 0 | 0 | 1 |
| HOFI | 0 | 0 | 4 | 6 |
| HOWR | 7 | 3 | 8 | 6 |
| INBU | 2 | 1 | 6 | 1 |
| MODO | 0 | 1 | 0 | 0 |
| NAWA | 0 | 0 | 1 | 0 |
| NOCA | 0 | 0 | 2 | 0 |
| NOMO | 4 | 1 | 4 | 12 |
| NOWA | 0 | 1 | 0 | 0 |
| OROR | 3 | 0 | 0 | 0 |
| OVEN | 1 | 0 | 1 | 2 |
| PIWA | 0 | 0 | 1 | 3 |
| PRAW | 20 | 16 | 10 | 8 |
| PUFI | 1 | 0 | 0 | 0 |
| RBGR | 1 | 4 | 1 | 0 |
| RBNU | 0 | 2 | 0 | 3 |
| RBWO | 0 | 1 | 0 | 0 |
| SOSP | 3 | 12 | 26 | 10 |
| SWSP | 3 | 0 | 0 | 0 |
| TRFL | 4 | 2 | 0 | 3 |
| TUTI | 0 | 3 | 0 | 0 |
| VEER | 3 | 2 | 1 | 3 |
| WBNU | 0 | 1 | 3 | 4 |
| WIWA | 1 | 0 | 0 | 0 |
| WOTH | 0 | 0 | 0 | 1 |
| YEWA | 0 | 0 | 1 | 0 |
| YSFL | 1 | 2 | 1 | 0 |
| YTVI | 0 | 0 | 2 | 0 |

Appendix B. Reference for species BBL codes.

| Species Code | Common Name | Scientific Name |
|---------------------|-------------------------|-----------------------------------|
| AMGO | American goldfinch | <i>Spinus tristis</i> |
| AMRO | American robin | <i>Turdus migratorius</i> |
| AMWO | American woodcock | <i>Scolopax minor</i> |
| BAOR | Baltimore oriole | <i>Icterus galbula</i> |
| BBCU | black-billed cuckoo | <i>Coccyzus erythrophthalmus</i> |
| BCCH | black-capped chickadee | <i>Poecile atricapillus</i> |
| BHCO | brown-headed cowbird | <i>Molothrus ater</i> |
| BLJA | blue jay | <i>Cyanocitta cristata</i> |
| BRTH | brown thrasher | <i>Toxostoma rufum</i> |
| BWWA | blue-winged warbler | <i>Vermivora cyanoptera</i> |
| CARW | Carolina wren | <i>Thryothorus ludovicianus</i> |
| CEDW | cedar waxwing | <i>Bombycilla cedrorum</i> |
| CHSP | chipping sparrow | <i>Spizella passerina</i> |
| COGR | common grackle | <i>Quiscalus quiscula</i> |
| COYE | common yellowthroat | <i>Geothlypis trichas</i> |
| CSWA | chestnut-sided warbler | <i>Dendroica pensylvanica</i> |
| DOWO | downy woodpecker | <i>Picoides pubescens</i> |
| EABL | eastern bluebird | <i>Sialia sialis</i> |
| EAKI | eastern kingbird | <i>Tyrannus tyrannus</i> |
| EAPH | eastern phoebe | <i>Sayornis phoebe</i> |
| EATO | eastern towhee | <i>Pipilo erythrophthalmus</i> |
| EAWP | eastern wood-pewee | <i>Contopus virens</i> |
| FISP | field sparrow | <i>Spizella pusilla</i> |
| GRCA | gray catbird | <i>Dumetella carolinensis</i> |
| HAWO | hairy woodpecker | <i>Picoides villosus</i> |
| HOFI | house finch | <i>Carpodacus mexicanus</i> |
| HOWR | house wren | <i>Troglodytes aedon</i> |
| INBU | indigo bunting | <i>Passerina cyanea</i> |
| MODO | mourning dove | <i>Zenaida macroura</i> |
| NAWA | Nashville warbler | <i>Vermivora ruficapilla</i> |
| NOCA | northern cardinal | <i>Cardinalis cardinalis</i> |
| NOMO | northern mockingbird | <i>Mimus polyglottos</i> |
| NOWA | northern waterthrush | <i>Seiurus noveboracensis</i> |
| OROR | orchard oriole | <i>Icterus spurius</i> |
| OVEN | ovenbird | <i>Seiurus aurocapilla</i> |
| PIWA | pine warbler | <i>Dendroica pinus</i> |
| PRAW | prairie warbler | <i>Dendroica discolor</i> |
| PUFI | purple finch | <i>Carpodacus purpureus</i> |
| RBGR | rose-breasted grosbeak | <i>Pheucticus ludovicianus</i> |
| RBNU | red-breasted nuthatch | <i>Sitta canadensis</i> |
| RBWO | red-bellied woodpecker | <i>Melanerpes carolinus</i> |
| SOSP | song sparrow | <i>Melospiza melodia</i> |
| SWSP | swamp sparrow | <i>Melospiza georgiana</i> |
| TRFL | Trail's flycatcher | <i>Empidonax alnorum/traillii</i> |
| TUTI | tufted titmouse | <i>Baeolophus bicolor</i> |
| VEER | veery | <i>Catharus fuscescens</i> |
| WBNU | white-breasted nuthatch | <i>Sitta carolinensis</i> |
| WIWA | Wilson's warbler | <i>Wilsonia pusilla</i> |
| WOTH | wood thrush | <i>Hylocichla mustelina</i> |
| YEWA | yellow warbler | <i>Dendroica petechia</i> |
| YSFL | yellow-shafted flicker | <i>Colaptes auratus</i> |
| YTVI | yellow-throated vireo | <i>Vireo flavifrons</i> |