# Avian Use Of Riparian Habitats And The Conservation Reserve Program:

# Migratory Stopover In Agroecosystems

THESIS

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## ABSTRACT

Although contemporary studies have emphasized the role of breeding and wintering grounds in sustaining populations of Nearctic-Neotropical migratory birds, migration is increasingly recognized as the most perilous and energetically demanding period in a bird's annual cycle. This presents difficulties for conservation biologists because even ambitious efforts to protect breeding and wintering habitat for birds may be ineffective if they fail to include adequate stopover habitats. Habitat conservation is especially challenging within agriculturally dominated regions of the midwestern U.S., where remnant habitats make up less than 15% of the landscape, suggesting that stopover habitat may be a limiting resource for migrant landbirds. Remnant natural habitats and restored habitats may provide important refueling opportunities and cover for migrants, yet few studies have examined migrant habitat use in agricultural landscapes.

Riparian habitat restoration in landscapes dominated by agriculture is aimed at reducing soil erosion and agricultural runoff and improving water quality, but evidence is mounting that restored habitats are also important for wildlife. The Conservation Reserve Program (CRP) and Conservation Reserve Enhancement Program (CREP), administered as part of the 1985 Food Security Act (Farm Bill), provide financial incentives to landowners who restore croplands to natural vegetative cover. The CRP has been shown to impart benefits to a variety of bird species, but these studies have focused

on grassland passerines and upland game birds during the breeding season. Recent studies suggest that early successional forest habitats contain a greater abundance and diversity of migrating birds and more food resources than mature forest interior. CRP and CREP habitats in agriculturally dominated regions may provide early successional habitat for Nearctic-Neotropical and temperate migratory landbirds, but the use of these riparian habitat restorations by landbird migrants is essentially unknown.

I examined associations between migratory land bird use and vegetation across a gradient of riparian forest conditions using transect surveys and mist-netting from late August through late October in 2009 and 2010 at 19 sites in Hancock and Allen counties, northwestern Ohio. Riparian habitats ranged from mature forest, early successional shrub-sapling and young forest restorations containing trees 2-15 m in height (CRP and CREP developed through the U.S. Department of Agriculture Farm Bill and other habitat conservation programs). An information theoretic approach was used to identify small-scale (microhabitat structure) and large-scale (percent forest cover within 500 m) habitat characteristics associated with bird abundances. Restoration habitats had fewer total bird numbers than remnant habitats, although restoration habitats were vegetatively similar to early-successional habitats at similar stages of succession. Woody stem densities and fruit-bearing shrub cover were positively associated with nearly all species and guilds studied, while pole-stage (8-23 cm dbh) trees were negatively associated. Relationships with percent forest cover within 500 m depended on the ecological requirements of the species, with Neotropical migrants and forest breeding species having a positive

relationship and early successional breeders and White-throated Sparrows (*Zonotrichia albicollis*) being negatively associated. My results indicate that vegetatively complex habitats tend to contain higher densities of fall migrants, and that some guilds and species are sensitive to large-scale forest cover surrounding stopover sites. Thus, benefits of riparian forest restorations to stopover migrant landbirds can be improved if managers (1) incorporate fruit-bearing shrubs, (2) manage for high structural heterogeneity, and (3) promote strategies to increase forest cover within the matrix surrounding restoration sites. Because both fine-scale and large-scale attributes were important predictors of use by migrating birds, my findings illustrate that effective strategies to provide stopover habitat must work across multiple spatial scales.

For my mother, Bonnie Colleen Condon-Gill

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## Fields of Study

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

# CHAPTER 1

# INTRODUCTION

Habitat loss and fragmentation are critical conservation issues for wildlife species worldwide. Migratory wildlife should be more vulnerable to these anthropogenic changes given that they use habitats at multiple locations during their annual cycles. For migratory birds, habitat loss on the breeding and wintering grounds has traditionally been considered to have the greatest impact on bird populations (Rappole and McDonald 1994, Askins 1995, Sherry and Holmes 1995). However, recent evidence suggests that mortality during the migratory period may have a greater effect on the long-term viability of populations (Moore et al. 1995, Sillett and Holmes 2002). Migration mortality may result from inclement weather conditions encountered during migratory flights, collision with manmade structures (Banks 1979), and resource competition at stopover sites (Newton 2006), but a paucity of adequate stopover habitats along the migratory route may play an important role.

Migration is the most energetically demanding phase in the annual cycle of a migratory bird (Bairlein 2002, Mehlman et al. 2005) and requires that birds periodically stopover in appropriate habitats to feed and rest. Stopover habitats are especially scarce in fragmented landscapes dominated by agriculture (Kirby et al. 2008). Riparian corridors in agricultural landscapes have also been recognized as valuable stopover habitat for Neotropical migrants (Skagen et al. 2005, Pennington et al. 2008); however, many riparian corridors have been marginalized by human activity, necessitating habitat restoration to provide benefits to wildlife and improve ecological services.

In the United States, habitat restoration within agricultural landscapes has been spearheaded by two programs established under the 1985 Food Security Act (Farm Bill): the Conservation Reserve Program (CRP) and the Conservation Reserve Enhancement Program (CREP), which provide financial incentives to private landowners who leave natural areas undisturbed or restore cropland to non-crop vegetative cover (Gray and Teels 2006). Habitats restored through the CRP and CREP reduce soil erosion and agricultural runoff and may play a key role in the reversal of population declines of many species of migratory birds. However, research to date has focused on use of CRP and CREP restorations by grassland species and upland game birds during the breeding season. Very few studies have investigated migrant landbird responses to riparian habitat restorations during the non-breeding season, and to my knowledge no studies have examined the use of riparian habitats restored through CRP or CREP.

Restored habitats have the potential to provide stopover habitats for migratory birds in fragmented agroecosystems. In northwestern Ohio, CRP and CREP riparian forest restorations are generally younger forest habitats, similar in height and tree size-class to shrub-sapling (early successional) forest. Recent studies indicate that early successional forests have a greater density and diversity of migrant birds during fall

(Rodewald and Brittingham 2004) and spring (Smith and Hatch 2008). Loss of early successional forests due to changes in logging practices (i.e. clearcutting) has had negative consequences for many bird species (North American Bird Conservation Initiative 2009). Because only 4-7% of the landscape in the lower Midwest is seedling-sapling stage (Trani et al. 2001), early successional habitat may be a limiting resource for birds migrating through this area.

The primary objectives of my research were to: 1) determine the abundance and diversity of fall migrant landbirds across a gradient of riparian stopover habitats in an agricultural matrix, 2) examine relationships between migrant abundance and fine-scale (microhabitat structure) and large-scale (percent forest cover within 500 m) habitat characteristics in restored habitats and remnant habitats, 3) determine whether abundance and diversity of stopover migrants in habitat restorations differed from remnant riparian habitats, and 4) develop habitat management recommendations based on these findings. I expected that 1) migrant abundance and diversity would be positively associated with dense vegetation and overall habitat complexity and to have no relationship with percent forest cover within 500 m, and 2) restored habitats would be vegetatively similar to and have similar abundance and diversity of migrants as remnant habitats at a similar stage of succession.

# LITERATURE REVIEW

## *Migration, stopover ecology and habitat use*

Although most contemporary studies have emphasized the importance of events occurring during breeding and wintering periods in sustaining populations of Nearctic-Neotropical (hereafter Neotropical) migrants and other migrant birds (Askins 1995, Brown and Sherry 2006, Keller and Yahner 2006), migration has recently been recognized as one of the most perilous and energetically demanding periods in a bird's annual cycle (Bairlein 1983, Mehlman et al. 2005). In Black-throated Blue Warblers (*Setophaga caerulescens*) nearly 85% of annual mortality occurs during migration, suggesting this period may have the greatest impact on long-term population viability (Sillett and Holmes 2002). Hence, benefits imparted to bird populations by protecting breeding and wintering habitat may be neutralized or even reversed without adequate conservation of stopover habitats (Martin and Karr 1986, Moore and Kerlinger 1987, Moore and Simons 1992, Parrish 2000, Mehlman et al. 2005, Newton 2006).

Migration is "a seasonal to-and-fro movement of a population between regions where conditions are alternately favorable or unfavorable" (Dingle and Drake 2007). Most species of Neotropical migrant landbirds spend less than 4 months per year on North American breeding grounds, and spend the remainder on tropical wintering grounds or traveling between the two. The amount of time that a bird spends in stopover (up to 2-3 months annually) far exceeds its time in flight and may ultimately determine the duration of its migration (Alerstam 1993). During the migratory period a migrant landbird will utilize several stopover sites, and individuals may spend a few hours to over a week at a single stopover site replenishing fat reserves and resting (Moore et al. 1995).

Stopover habitats must provide migrants with high-quality food items (Kirby et al. 2008) and sufficient cover in which to rest and evade predators as they traverse unfamiliar landscapes (Nemeth and Moore 2007). Numerous studies suggest that migrants actively select stopover sites (reviewed in Petit 2000), although the mechanisms behind habitat selection remain unclear (Moore and Aborn 2000). In highly modified landscapes, stopover migrants may be forced to utilize stopover habitat patches that are unlike their preferred breeding habitat, but may provide adequate food resources and vegetative cover to support short term needs. Mehlman et al. (2005) classified stopover habitat into three types: "full-service hotels" that provide plentiful food resources and relatively low risk of predation where birds may linger for several days or more before continuing migration, "convenience stores" where birds have access to sufficient food and water to continue migration within about 24 hours, and "fire escapes" that confer little benefit aside from providing cover in ecologically inhospitable landscapes.

Remnant habitat is especially scarce in many agricultural regions and riparian corridors are recognized as valuable stopover habitat in other landscapes, e.g. in urban areas of the eastern United States (Pennington et al. 2008) and in the Southwest (Skagen et al. 2005). During the breeding season, riparian corridors in fragmented landscapes contain a greater diversity of birds than upland woodlots (Gentry et al. 2006), and riparian forest area rather than width has been shown to be an important determinant of such diversity (Groom and Grubb 2002). This pattern may not hold during migration for some species, as abundance of Bay-breasted Warblers (*Setophaga castanea*) and Blackpoll Warblers (*Setophaga striata*) was not associated with patch size during migration (Keller and Yahner 2007). Similarly, avian diversity and abundance did not differ between upland and riparian boreal forest during migration (Mosley et al. 2006). The surrounding matrix and woody species composition within a habitat may also influence habitat choice since mature upland forest may be more heavily used by migrating landbirds than riparian forest in urban areas of the Midwest (Rodewald and Matthews 2005). However, many riparian areas have been compromised by human activity, especially within urban and agricultural landscapes (Chazdon 2008).

In addition to riparian forest habitats, there is mounting evidence that shrub-sapling habitats have higher densities of migrants than mature woodland during both fall (Rodewald and Brittingham 2004) and spring (Smith and Hatch 2008), regardless of the species' preferred breeding habitat. Many Neotropical migrants exhibit less habitat specificity during migration, possibly in response to food resource availability (Wang and Finch 2002). Migrant density and diversity are also strongly related to vegetative structure, which is often diverse in shrub-sapling (early successional) habitats and in turn supports a larger abundance of insect prey (Smith and Hatch 2008). Birds also appear to select early successional habitats during the post-breeding season because of higher fruit and arthropod prey abundance as well as dense vegetation for cover (Suthers et al. 2000, Keller et al. 2003, Vitz and Rodewald 2007). However, experimental manipulation of arthropod abundance in early successional habitats had no effect on capture or foraging rates of landbirds in South Carolina during spring and fall migration or breeding periods, suggesting that habitat choice may be driven by factors other than prey availability (Champlin et al. 2009). Since lower survival and reproductive success necessitates a higher population density to maintain population size, wildlife abundance may not accurately indicate habitat quality during the breeding season (Van Horne 1983, Donovan and Thompson 2001), but whether this holds true for stopover migrants is unclear. Though the mechanisms behind habitat selection and the meaning of this increased abundance remain unclear, evidence suggests that early successional habitats are heavily used by migratory landbirds during stopover.

### *Habitat restoration and land conservation incentives*

Humans have altered the structure and function of nearly all ecosystems on earth by increasing nutrients and pollutants, suppressing fire events, diverting water, changing precipitation patterns via the release of greenhouse gasses, and converting undisturbed land to row crops (Dodds et al. 2008). Landscape-scale habitat fragmentation is a major wildlife conservation issue in many parts of the world and is cited as one of the main causes of species declines and extinctions across taxa, from endemic Hawaiian birds (Boyer 2008) and perennial herbaceous plants in central Europe (Winter et al. 2008) to

Malagasy amphibians (Andreone et al. 2005) and freshwater mussels in the Midwestern United States (Lyons et al. 2007). In the United States less than 10% of native ecosystems remain (Dodds et al. 2008), and avian population trends in North America are closely linked to land use changes (Murphy 2003). Conversion of forest and grassland ecosystems to agriculture has been associated with regional declines of migratory landbirds (Kirby et al. 2008, North American Bird Conservation Initiative 2009) and among those hardest hit are long-distance migrants that nest in forested habitats (Askins 1995). This is potentially problematic in the lower Midwestern United States, where agriculture comprises up to 75% of the landscape (Trani et al. 2001), but few studies have assessed stopover habitat selection in inland agricultural landscapes (Ewert and Hamas 1995, Ewert et al. 2006).

In areas where anthropogenic changes have marginalized landscapes for native species, habitat restoration is necessary to improve ecological services and restore biodiversity (Chazdon 2008, Luther et al. 2008, Wade et al. 2008). Riparian areas are often targeted for restoration in marginal or fragmented landscapes because they usually retain some natural characteristics or can be more easily restored to a near-natural state, even in areas that have been heavily modified by human activity (Luther et al. 2008). Although restored lands are not equivalent to native lands with respect to biodiversity (Chazdon 2008) they have been shown to provide 31-93% of the ecological benefits of native lands within ten years of restoration (Dodds et al. 2008). Similarly, rainforest restoration sites in Australia increasingly resembled native rainforest in terms of beetle species assemblages as the age of sites increased, but were intermediate between native rainforest and pasture (Grimbacher et al. 2007). Brockerhoff et al. (2008) reviewed a variety of plantation forest studies in Europe and North America and concluded that the wildlife species diversity of restored plantation forest is inferior to that of native forest but almost always exceeds that of agriculture or pastureland. In addition, plantation forests can facilitate succession in areas where natural succession has been hampered (Brockerhoff et al. 2008). Although plantation habitats are often viewed as suboptimal, shade coffee plantations that retain a canopy of native tree species harbored a higher bird abundance and species richness as they aged and as stem numbers increased (Wunderle 1999). In highly agricultural areas of Switzerland, ecological compensation areas (ECAs) positively influenced pollinator diversity and pollinator services in the intensively managed meadows adjacent to them (Albrecht et al. 2007).

Growing concern over compromised ecological services and lost biodiversity has spurred widespread efforts to restore native habitats. On a global scale, forest cover has increased through both assisted restoration and natural succession, and ecosystem decline within agricultural landscapes has resulted in a variety of management programs aimed at restoring biodiversity (Chazdon 2008). In the United States, this task was greatly facilitated by the 1985 Food Security Act, otherwise known as the Farm Bill. The Farm Bill has become one of the most significant acts of legislation in the United States regarding land conservation in the private sector (Gray and Teels 2006). Among its most successful programs, the Conservation Reserve Program (CRP) provides contractual financial incentives to private landowners who either protect tillable natural areas or convert existing pastures and croplands into areas of non-crop vegetative cover to reduce soil erosion and runoff. Amendments to the Farm Bill in the 1990s and early 2000s added provisions to enhance habitat for fish and wildlife, such that high-priority conservation practices including riparian buffers, wildlife habitat buffers, herbaceous filter strips, contour grass strips, wetland restoration and wetland buffers became eligible for enrollment in the program.

The Conservation Reserve Enhancement Program (CREP), another incentive-based program under the Farm Bill, differs from the CRP in that the specific goal of CREP is to protect environmentally sensitive lands that suffer from increased risk of erosion. CREP was initiated in 2002 to create partnerships with federal, state and nongovernmental entities to address conservation on landscape scales (Gray and Teels 2006) and has generally garnered positive reactions from landowners (Suter et al. 2008). Unlike upland grass-dominated CRP, CREP lands are almost exclusively riparian and oftentimes planted with trees. Because CREP habitats in this study are indistinguishable from CRP, the term "restoration" includes both CRP and CREP habitats unless otherwise stated.

Although direct benefits of the CRP to wildlife are difficult to quantify due to the complex and dynamic nature of agricultural landscapes (see Giudice and Haroldson 2007), CRP restorations are positively correlated with wildlife abundance and arthropod prey availability. For example, arthropod abundance and diversity across grassland CRP sites in Texas was less than that of native prairie but greater than adjacent croplands, and may explain recent reversing population declines in a few species of grassland breeding birds (McIntyre and Thompson 2003). CRP fields in Kansas contained more arthropod prey than surrounding cultivated fields, providing better foraging opportunities for Ring-necked Pheasant (*Phasianus colchicus*) and Northern Bobwhite (*Colinas virginianus*) broods (Doxon and Carroll 2007). The amount of CRP enrollment within a 1 km buffer of Breeding Bird Survey (BBS) routes was positively correlated with Ring-necked Pheasant abundance in nine Great Plains states (Nielson et al. 2008). Similarly, although restored CRP grassland comprises only 3.5% of the landscape in the Midwest, population trends of grassland-nesting birds were strongly linked to the presence of restored grassland within the landscape surrounding BBS point counts (Veech 2006). Furthermore, rangewide population increases of Henslow's Sparrows (*Ammodramus henslowii*), a species of high conservation concern, were positively correlated with CRP enrollment within 3 km of BBS routes (Herkert 2007). Conversion of cropland to grassland CRP in Minnesota was associated with higher abundances of Ring-necked Pheasant and *Sturnella* meadowlarks (Haroldson et al. 2006) and grassland CRP appears to be important habitat for Eastern Meadowlarks (*Sturnella magna*) during the post-breeding season in Wisconsin (Guzy and Ribic 2007).

Few studies have assessed potential benefits of CRP or CREP habitat restorations to wildlife, and to my knowledge few studies have been published on the use of riparian forest restorations by birds; however, several studies demonstrate the positive effect of habitat restorations on bird communities. Avian abundance and diversity was significantly higher in CRP filter strips than non-buffered field edges (Blank et al. 2011). Smiley et al. (2007) found that migrant landbirds were most abundant during the migratory period in restorations with more woody vegetation. Twedt et al. (1999) demonstrated that replacing agriculture with non-CRP cottonwood plantations in the Mississippi valley increased the number of breeding bird territories 6-9 years after planting, but studies of this kind during the migratory period have not yet been published.

Although the CRP is often regarded as a panacea for a variety of environmental ailments, many challenges remain, among them the planting of non-native species, a renewed interest in biofuels and disagreements over the application of proper disturbance and management regimes to CRP lands. While CRP riparian buffer strips may provide important breeding habitat to grassland species of conservation concern, their narrow width may reduce avian nesting success due to predation, and future CRP enrollments for wildlife should consider increasing buffer widths to mitigate this effect (Henningsen and Best 2005). As interest in alternative energy increases, landowners may choose to convert expiring CRP contracts into biofuel crops, potentially reversing the recent populations increases of some species of grassland breeding birds. However, rotating harvest regimes of switchgrass (*Panicum virgatum*) plots may be a viable alternative to corn since winter-harvested switchgrass can provide breeding habitat for Grasshopper Sparrows (*Ammodramus savannarum*), Northern Harriers (*Circus cyaneus*) and Ring-necked Pheasants while still providing economic benefit to the landowner (Murray and

Best 2003); also, such harvest regimes may mimic natural disturbance. A study of habitat selection in Grasshopper Sparrows in Maryland revealed that the birds were absent from CRP grassland plots that were not treated with fire regimes or herbicide within the last 2-3 years (Gill et al. 2006). This complements the finding that species richness of migratory passerines is greater in CRP plots with appropriate disturbance regimes, such as fire and herbicides, suggesting that CRP may need continued management to provide maximum benefit to wildlife (Sladek et al. 2008).

### *Large-scale effects on habitat choice*

Another factor to consider when evaluating stopover habitats in agricultural landscapes is the effect of the surrounding matrix on habitat choice. Both microhabitat features as well as landscape attributes are known to influence stopover habitat use (Buler et al. 2007, Packett and Dunning 2009), although landscape-scale effects on habitat selection have been less commonly examined. In agricultural areas of Sweden, effects of landscape structure (homogeneity vs. heterogeneity) and agricultural practices (conventional vs. organic) on the density and species richness of stopover migrants depended on the ecological requirements of the species, with passerines being more numerous in heterogeneous landscapes and organic agriculture (Dänhardt et al. 2010). Conversely, surveys in agricultural landscapes of Indiana found that distance to riparian area and woodlot isolation had no effect on migrant bird diversity or abundance, suggesting that migrants selected habitat based on local habitat characteristics and particularly food availability; however, woodlot size within a 5 km radius did have a moderate positive effect on migrant abundance (Packett and Dunning 2009). Buler et al. (2007) also found that microhabitat features rather than large-scale features appeared to influence stopover migrant abundance. In addition, forest- and edge-breeding species respond differently to the amount of mature forest cover in the landscape, with early-successional breeding species showing an inverse relationship to forest cover and forest-breeding species showing a positive relationship (Perkins et al. 2003).

### *Diet and energetics*

During migration birds can exhibit plasticity not only in habitat preference, but also in dietary choices. Many insectivorous species become partially or largely frugivorous during the fall migratory period (Martin and Karr 1986). This shift presumably occurs in order to meet energetic demands (Bairlein 2002) or compensate for variable resource abundance (Parrish 1997a), but some birds exhibit seasonal shifts in dietary preference spontaneously, even under laboratory conditions where a wide variety of foods are available (Bairlein 2002). Switching to an omnivorous diet may reduce catabolism of bodily reserves of protein (i.e. muscle tissue; Gannes 2001). A shift in diet and habitat preference underscores how migrating birds have different physiologies relative to those utilized in the breeding or wintering periods (Parrish 2000). For example, migrating Garden Warblers (*Sylvia borin*) undergo a shrinking and subsequent regrowth of the reproductive and digestive systems and muscle groups (Biebach 1998, Bauchinger et al. 2005) depending on the phase of their migration.

Extent of frugivory in several species of fall migrants was positively correlated with average change in body condition and fat scores, suggesting that fruit consumption allowed migrants to gain mass more efficiently (Parrish 1997b). Quantification of plasma triglyceride and uric acid levels in free-living migrating passerines suggest that for some species there may be a relationship between fruit abundance within stopover sites, protein metabolism and fat deposition, but metabolite studies on free-living birds are fraught with difficulties and results can be difficult to interpret (Smith and McWilliams 2010). Efforts to understand how energetic condition and resource availability influence stopover duration and habitat choice, though challenging to quantify, are nonetheless necessary to identify the habitats that provide maximum energetic benefits to migrants.

Migration is energetically costly for small passerines (Bairlein 2002) and the procurement of high-quality food items prior to and during migration is paramount, especially if resources are limited at the onset of migration. Manipulated and natural variation in food availability had measurable effects on the body condition of wintering Ovenbirds (*Seiurus aurocapilla*), which may have fitness consequences that carry over into the migratory and breeding periods and ultimately influence population trends over time (Brown and Sherry 2006). Food availability during the overwintering period was positively correlated with bird abundance, body condition and habitat quality of both *in* *situ* and relocated American Redstarts (*Setophaga ruticilla*; Studds and Marra 2005 and Johnson et al. 2006, respectively). Food may not be the only resource limiting successful fat deposition; there is some evidence that limited water availability can also impede a migrant's ability to deposit fat (Tsurim et al. 2008).

How individual birds decide when to depart stopover habitats appears to be largely dependent on fat stores upon arrival or its ability to replenish those fat stores (Schaub et al. 2008). A study on Horn Island, Mississippi revealed that after crossing the Gulf of Mexico spring migrant Summer Tanagers (*Piranga rubra*) in lean condition were more active and departed the island sooner than fatter individuals (Moore and Aborn 2000). In contrast, lean captive spring migrant thrushes (*Hylocichla* and *Catharus*) exhibited more activity during the day (inferred feeding behavior) and less nocturnal migratory restlessness than fatter individuals, and became more active at night as they gained mass (Yong and Moore 1993). Similarly, Arizaga et al. (2008) found that stopover length of stay of Blackcaps (*Sylvia atricapilla*) during autumn migration in Spain was negatively correlated with body mass at first capture.

Given the cost of migration and the potential reproductive advantages incurred by early arrival at the breeding grounds in spring (Smith and Moore 2005), a migrating bird should minimize its stopover time and hence the duration of its migration (Alerstam and Lindström 1992). An individual should concurrently maximize its energy stores by using stopover sites that have an abundance of food resources (Moore et al. 1995) and choosing food items that provide the greatest amount of energy for the lowest cost to procure them

(Gannes 2001, Schaub et al. 2008). Early arrival at the breeding grounds with excess fat stores imparts higher reproductive success in terms of clutch size and nestling mass (Smith and Moore 2003). Recent studies of fat-deposition strategies suggest that migrating landbirds may be responding to daily energetic needs rather than energetic needs of days to come (Benson and Winker 2005). This contrasts the generally accepted idea that birds "fatten up" at the breeding or wintering grounds before departing on long-distance migrations, although this pattern is seen where birds must cross ecological barriers. For example, migrating Garden Warblers (*Sylvia borin*) are known to increase their body mass 50-100% immediately prior to crossing the Mediterranean Sea or the Sahara Desert, where there are limited opportunities to forage (Ottosson 2005). Agriculture and landscapes heavily modified by human activity may represent a similar ecological barrier to some migrant species.

## STUDY AREA

My study was conducted in northwestern Ohio from mid August through October in 2009 and 2010. Study sites were located in southern Hancock County and northeastern Allen County on public (County Parks, Ohio Division of Wildlife property) and private lands (Boy Scouts of America, private landowners) along the Blanchard River and its tributaries, at elevations ranging from 238 m to 248 m. Land cover in these areas is dominated by agriculture (corn, soybeans and wheat) but also includes, in descending order, mature riparian and upland woodlots with open understory, urban development, shrub-sapling habitat, pasture, and wetland.

The vegetation density and structure, size, age and management history of study sites display a great deal variability and overlap. Dominant tree species in the study area include ashes (*Fraxinus* spp.), elms (*Ulmus* spp.) box elder (*Acer negundo*), common hackberry (*Celtis occidentalis*) maples (sugar, *Acer saccharum*; silver, *A. saccharinum*; and black, *A. nigrum*), and oaks (red, *Quercus rubra*; white, *Q. alba*; pin, *Q. palustris*; chinquapin, *Q. muehlenbergii*; and bur, *Q. microcarpum*). For purposes of this study, "riparian" study sites ranged from natural, permanent flowing water to intermittent anthropological waterways, i.e. drainage ditches; many CRP habitats fell into the latter category. Site habitats included mature riparian forest, naturally succeeded shrub-sapling habitat, and young riparian habitat restorations (CRP). Mature riparian forests were 2-82 ha in area and contained trees with mean dbh > 23 cm and mean height > 15 m. Shrub-sapling habitats were rare in the study area and 2-12 ha in area, 8-15 years in age, and had > 50% shrub cover. Riparian restoration habitats (CRP and CREP) ranged from 2-32 ha in area and contained sapling (3-8 cm dbh) to pole stage (8-12 cm dbh) trees planted at 3 m intervals.

Study sites restored using CRP practices included CP11 (re-enrolled young trees), CP22 (riparian buffer strips), CP31 (bottomland hardwood) and CP4D (wildlife habitat). All study sites met the following criteria: more than 50% of trees were over 2 m tall, distance to riparian area was less than 150 m, distance to nearest road was less than 500 m, and permission to use the site was granted by landowners. The study area was chosen because there are sufficient numbers of both enrolled CRP lands and fragmented native habitats for sampling. CREP sites were used for the study as much as possible, however because few were available in the study area, CRP sites similar in habitat structure and hydrology were also used.

## SIGNIFICANCE

To reverse population declines of migratory landbirds, the conservation of stopover habitat deserves as much attention as breeding and wintering habitat (Parrish 2000, Mehlman et al. 2005, Newton 2006, Deppe and Rotenberry 2008). Understanding avian habitat relationships during stopover and the energetic consequences of habitat choice is necessary to make informed conservation decisions to maximize benefits to landbird populations. Within agriculturally dominated landscapes the availability of isolated patches of early successional stopover habitat may be of great importance to migrant populations, especially if they allow them to successfully refuel during stopover. Habitat restoration within agricultural systems has the potential to provide stopover habitat, yet to my knowledge few studies have investigated the importance of riparian forest restorations to woodland passerines during migratory stopover. Such research will become increasingly important as remnant habitats are modified for human use and replaced with cropland or restored habitat. This study expands our current understanding of stopover habitat selection in midwestern agricultural landscapes and investigates the extent to which CRP and CREP riparian restorations provide suitable stopover habitat for Neotropical migrants. Thus, results of this study ultimately can be applied in other agricultural landscapes to create or restore riparian habitats specifically for landbirds during stopover.

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# CHAPTER 2

# FALL MIGRANT LANDBIRD USE OF NATIVE AND RESTORED HABITATS IN AN AGRICULTURALLY DOMINATED LANDSCAPE

## ABSTRACT

Stopover habitat may be a limiting resource for forest birds in agriculturally dominated regions of the Midwestern U.S. In these landscapes, remnant natural and restored habitats may provide important refueling opportunities and cover for migrants, yet few studies have examined migrant abundance in these habitats. I studied migratory land bird use and vegetative structure across a gradient of riparian forest conditions that included mature forest, early successional shrub-sapling habitat and young forest restorations containing trees 2-15 m in height developed through Farm Bill habitat conservation programs (CRP and CREP). I conducted transect surveys and mist-netting from late August through late October in 2009 and 2010 at 19 sites in Hancock and Allen counties, northwest Ohio. An information theoretic approach was used to identify fine-scale (structural and floristic) and large-scale (forest cover within 500 m) habitat characteristics associated with bird abundances. Overall, capture and transect detection rates for Neotropical and temperate migrants were lowest in restored habitats and highest

in early successional habitats. Restoration habitats had fewer total bird numbers than remnant habitats although restoration habitats were vegetatively similar to early-successional habitats at similar stages of succession. Woody stem hits were positively associated with nearly all migrant species and guilds studied, while pole-stage (8-23 cm dbh) trees were negatively associated. Relationships with percent forest cover within 500 m depended on the ecological requirements of the species, with Neotropical migrants and forest breeding species having a positive relationship, whereas early successional breeders were negatively associated. My results indicate that vegetatively complex habitats tend to contain higher densities of fall migrants, and that abundances of some guilds and species are sensitive to forest cover surrounding stopover sites. My research suggests that riparian forest restorations should increase structural heterogeneity by incorporating more shrub layer vegetation and that effective strategies to provide stopover habitat for stopover migrant landbirds must work across multiple spatial scales.

# INTRODUCTION

Habitat loss and fragmentation on breeding and wintering grounds are widely recognized as critical threats to birds (Rappole and McDonald 1994, Askins 1995, Sherry and Holmes 1995). However, conditions encountered during the migratory period have been historically overlooked despite evidence that the migratory period can mediate the long-term viability of bird populations (Sillett and Holmes 2002, Newton 2006). Migration is energetically costly, and landbird migrants must stopover periodically in appropriate habitats to meet their energetic demands and rest (Hussell and Lambert 1980, Bairlein 2002, Kirby et al. 2008). For a forest bird, stopover habitat is limited in landscapes dominated by agriculture. This may be especially true in some areas of the lower midwestern United States (Ewert and Hamas 1995), where agriculture comprises up to 85% of the landscape (Trani et al. 2001). Riparian forest corridors in agricultural landscapes have been recognized as valuable stopover habitat for Neotropical migrants (Skagen et al. 2005, Pennington et al. 2008); however, most riparian forests have been degraded by human activity, and habitat restoration may be necessary to provide benefits to wildlife and improve ecological services.

The decline of ecosystems within agricultural landscapes has led to a variety of conservation programs aimed at restoring biodiversity (Chazdon 2008). In the United States, habitat restoration was spearheaded by the Conservation Reserve Program (CRP) and the Conservation Reserve Enhancement Program (CREP) established in 2002 under the 1985 Food Security Act (i.e. Farm Bill). These programs provide financial incentives to landowners who restore cropland to non-crop vegetative cover or leave natural areas undisturbed (Gray and Teels 2006). CRP and CREP (henceforth CRP) habitat restorations are designed to reduce soil erosion and agricultural runoff, but also confer benefits to wildlife (McIntyre and Thompson 2003, Kamler et al. 2005, Herkert 2007). A growing number of studies demonstrate positive responses of breeding birds to increasing vertical structure of riparian forest restorations (Twedt et al. 2002, Norris 2009, Twedt et al. 2010). Riparian forest restorations such as CRP and CREP may have the potential to provide stopover habitats for migratory birds in fragmented agroecosystems, but no studies have addressed the extent to which migrants use and benefit from these habitat restorations. Many riparian CRP restorations in northwestern Ohio are similar in height and tree size-class to early successional (i.e. shrub-sapling) habitat. Recent studies suggest that shrub-sapling habitats have a greater abundance and diversity of migrants during fall (Rodewald and Brittingham 2004) and spring (Smith and Hatch 2008), regardless of migrants' preferred breeding habitat.

Local as well as landscape-scale habitat characteristics can strongly affect bird abundance and habitat choice during stopover, although few studies have examined landscape-scale effects on landbird stopover habitat selection (Buler et al. 2007, Packett and Dunning 2009). Distance to riparian area and woodlot isolation in an agricultural landscape of Indiana was not associated with migrant songbird diversity or abundance, suggesting that habitat selection was based on local habitat characteristics and food availability (Packett and Dunning 2009). This is consistent with Buler et al. (2007), who also reported that hardwood forest cover within 5 km had a moderately positive effect on migrant abundance during stopover in coastal Mississippi and Louisiana. In agricultural areas of Sweden, migrating passerines responded both to landscape structure (homogeneity vs. heterogeneity) and agricultural practice (conventional vs. organic) of stopover habitats, and were most numerous in heterogeneous landscapes and organic agriculture (Dänhardt et al. 2010).

The primary objectives of my research were to: 1) determine the abundance and diversity of fall migrant landbirds across a gradient of riparian stopover habitats in an agricultural matrix, 2) examine relationships between migrant abundance and fine-scale (microhabitat structure) and large-scale (percent forest cover within 500 m) habitat characteristics between remnant and restored habitats, and 3) compare abundance and diversity of stopover migrants in remnant and restored riparian habitats. I expected that 1) migrant abundance and diversity would be positively associated with dense vegetation and unrelated to percent forest cover within 500 m, and 2) restored habitats would be vegetatively similar to and have similar abundance and diversity of migrants as remnant habitats at similar stages of succession.

## METHODS

### *Study area and sites*

The study was conducted at 19 sites in Hancock County and northeastern Allen County of northwestern Ohio (40.96°N, -83.65ºW) from August 26 to October 29, 2009 and August 22 to October 25, 2010. Study sites were located on public (County and State Parks) and private lands (Boy Scouts of America, individual landowners) along the Blanchard River and its tributaries (Table 2.1), at elevations ranging from 238 m to 248 m. Land cover in this area was approximately 71% agriculture (corn, soybeans and wheat), 12% riparian and upland woodland, 9% low-intensity development (e.g. suburban yards, golf courses) 5% pasture, 1% urban development, 1% open water, and 1% wetland (Figure 1). Study sites met the following criteria: more than 50% of trees were over 2 m tall, distance to the nearest riparian area was less than 150 m, the nearest road was within 500 m and accessible by foot, and permission to use the site was granted by landowners. Study sites restored using CRP and CREP practices included CP11 (re-enrolled young trees), CP22 (riparian buffer strips), CP31 (bottomland hardwood) and CP4D (wildlife habitat).

Study sites varied widely in terms of vegetation density and structure, size, age and management history. Dominant tree species in the study area include ashes (*Fraxinus* spp.), elms (*Ulmus* spp.) box elder (*Acer negundo*), common hackberry (*Celtis occidentalis*) maples (sugar, *Acer saccharum*; silver, *A. saccharinum*; and black, *A.*

*nigrum*), and oaks (red, *Quercus rubra*; white, *Q. alba*; pin, *Q. palustris*; chinquapin, *Q. muehlenbergii*; and bur, *Q. microcarpum*). For purposes of this study, I considered riparian to include the full range of natural, permanent flowing water to intermittent anthropological waterways, including drainage ditches that were commonly adjacent to CRP habitats. Habitats included mature riparian forest, naturally succeeded shrub-sapling habitat, and young riparian habitat restorations (CRP and CREP). Mature riparian forests were 2-82 ha in area and contained trees with mean dbh > 23 cm and mean height > 15 m. Shrub-sapling habitats were rare in the study area and 2-12 ha in area, 8-15 years in age, and had > 50% shrub cover. Riparian restoration sites ranged from 2-32 ha in area and contained sapling (3-8 cm dbh) to pole stage (8-12 cm dbh) trees planted at 3 m intervals.

### *Avian sampling*

I sampled birds using both line transects and mist-nets (Bibby et al. 2000) as several studies have demonstrated the value of using mist-nets in tandem with distance sampling (Dunn et al. 1997). Distance-based sampling methods adjust for differences in bird detectability among sites of greatly different habitat structure, whereas mist-netting can improve detections, particularly during fall when migrant songbirds are much less vocal (Rappole et al. 1998). The methods can complement one another, especially when multiple habitat types are compared (Rappole et al. 1998, Pagen et al. 2000, Wang and Finch 2002).

When possible, transects (100 m, n = 39) were placed at least 75 m apart at a site to eliminate the likelihood of double-counting individual birds. Surveys took place from 15 minutes after sunrise until 4 hours thereafter and were not conducted during periods of heavy wind or rain. I surveyed sites 1-2 times per week with at least two days between visits for a total of 13-17 visits per season. Observers walked each transect at ~1 km/hr and recorded compass bearings and distances to each observation with range finders. I used the distance and bearing to each observation to calculate perpendicular distances from the bird to the transect line using trigonometric functions.

I used mist nets to sample non-game migratory birds at 6 sites, two each in early successional shrub-sapling, mature forest and CRP habitat restorations. Sites selected for netting contained vegetation at least as tall as the mist nets (2.6 m) and all but two sites were at least 1 km apart (Figure 2.1). I used a standardized protocol for operating banding stations based on that developed by the Institute for Bird Populations (IBP) for Monitoring Avian Productivity and Survivorship (MAPS) to ensure bird safety and consistent mist net operation (DeSante et al. 2001). Eight 12 m mist nets (30 mm mesh) were operated at each site every 2-3 days from sunrise to five hours thereafter. We did not operate mist nets in extreme temperatures (less than 2° C or greater than 35° C) or during periods of heavy wind or rain. I removed captured birds from mist nets at 30 min intervals and held them (< 30 min) in fabric drawstring bags until processing. I sexed and aged individuals according to Pyle (1997) (and skull pneumatization when possible) and fitted captures with a USGS aluminum band (Northern Cardinals and Rose-breasted Grosbeaks with stainless steel). I recorded tarsus length to the nearest 0.1 mm with calipers, unflattened wing chord to the nearest 0.5 mm with a wing rule, fat score (scale of 0-5, Helms and Drury 1960), and mass to the nearest 0.1 g with a digital scale for each capture. Field methods were approved by the OSU Institutional Animal Care and Use Committee (IACUC permit 2009A0034).

### *Microhabitat characterization*

Vegetation data were collected at all sites in August 2010 following a modified James and Shugart (1970) protocol. Sampling occurred within 0.04 ha circular (11.3 m radius) plots located at the center of net locations and three at 25, 50 and 75 m along each transect. Tree and woody shrub species were counted in three diameter-at-breast-height (dbh) size classes (3-8 cm, 8-23 cm, and > 23 cm) within each plot. Every 2 m along transects oriented N-S and E-W within each plot, I recorded canopy cover hits with an ocular tube (0 or 1), ground cover less than 0.5 m tall (litter, soil, rock, moss and forb), and woody stem and forb hits between 0.5-3 m on a vertical pole. I measured canopy height in meters with a rangefinder at the center of each plot. Vegetation data were averaged over the three plots for each transect prior to analysis. In cases where mist nets were less than 10 m apart, one vegetation plot was conducted for the adjacent nets and capture rates were averaged.

Fruit resources are positively associated with bird abundance during the autumn migratory and postbreeding season (Suthers et al. 2000 and Vitz and Rodewald 2007, respectively). Within each vegetation plot, I estimated the percent cover of autumn fruit-bearing shrubs, vines and trees < 3 m tall at both mist-net locations and transects since counting individual ripe fruits weekly at all sites not logistically possible. Species included poison ivy (*Toxicodendron radicans*) highbush cranberry (*Viburnum trilobum*), grape (*Vitus* spp.), honeysuckle (*Lonicera* spp.), dogwood (*Cornus* spp.) and blackberry (*Rubus* spp.). All of the above species had ripe fruits and were consumed by migrants during the field season.

### *Forest cover quantification*

As migrating birds end nocturnal flights and descend into habitats, their ability to evaluate local microhabitat conditions is likely compromised due to darkness and they likely rely more heavily on broad-scale cues (Hutto 1985), such as percent forest cover in the surrounding landscape (Buler et al. 2007). I used National Land Cover Database (NLCD 2006) raster layers with 30 m resolution and GIS software (ESRI 2010, ArcGIS v. 9.3.1) to extract percentage of mature forest cover within 500 m of transects post-hoc to explore impacts of large-scale habitat features of potential importance to migrants. In cases where transects were less than 100 m apart and located within the same habitat type, microhabitat features were averaged across all plots and survey data from transects within the buffer were combined.

*Avian data treatment*

I examined the total number of birds detected per season on transect surveys or captured in mist-nets after Packett and Dunning (2009). Capture rates were standardized as the number of birds captured per 1000 net hours and then rounded to the nearest whole number for analysis. Detection rates were calculated as total detections per 100 min to account for slight differences in sampling times across sites and then rounded to the nearest whole number for analysis. I truncated survey data at 25 m from the transect line because number of detections decreased sharply beyond this distance. In doing so, I fixed the area sampled by each 100 m transect at 0.5 ha.

I conducted analyses for the top five most abundant transient species. To examine patterns for ecologically similar species and boost sample sizes, I classified bird species into guilds based on migratory behavior (Nearctic-Neotropical vs. temperate migrants), breeding habitat (mature forest vs. early successional habitat) and diet (obligate frugivores). Species were classified into migrant and breeding habitat guilds following DeGraff and Rappole (1995), and obligate frugivores included those with diets having > 66% fruit according to Parrish et al. (1997). American Robins (*Turdus migratorius*) were excluded from the transect data analysis because they comprised over 50% of the obligate frugivore guild and largely defined guild patterns. To determine which habitat features are associated with transient species diversity I calculated first-order jackknife estimates after Gonzalez-Oreja et al. (2010).

Because the sampling distributions of bird and habitat data were non-Gaussian, I used non-parametric two-sample t-tests to examine annual differences in capture and detection rates. I considered $p < 0.05$ to be statistically significant. Finding no annual differences in total numbers of birds captured in nets ($t = 0.55$, $df = 67.99$, $p = 0.581$) or detected along transects ($t = 0.01$, $df = 75.75$, $p = 0.99$), I pooled across years in analyses. To detect differences in capture and detection rates and habitat features across habitat types I used a one-way ANOVA and Tukey's Honestly Significant Difference with a familywise error rate of 0.05.

### *Vegetation data treatment*

I used a Spearman correlation matrix to identify highly correlated vegetation measures and chose variables to include in the models that both captured microhabitat variation and reflected features important to migrants for transect data (Table 2.2) and mist-netting data (Table 2.3). Canopy height and percent canopy cover were excluded from the modeling because they were strongly correlated with Trees > 23 cm dbh. Percent fruit-bearing shrub cover ("% Berry") was excluded from transect data modeling since it was strongly correlated with woody stem hits ($r^2 = 0.73$), suggesting that most woody stem hits were likely fruit-bearing shrubs; % Berry was included in the modeling of mist-netting data since it was not strongly correlated with any other vegetation measures, and fruit-bearing shrub cover immediately surrounding a mist net likely influences capture rates within that net. Forb stem hits were excluded since forb cover was negatively correlated with trees > 23 cm dbh ($r^2 = -0.77$). Variables used for modeling detection rate data were percent forest cover within 500 m, all three tree size classes, woody stem hits, and bank fullwidth of the adjacent riparian area ("water"). Variables used for modeling mist-netting data were all three tree size classes, woody stem hits, water, and % Berry.

*Modeling*

Using transect data, I constructed a series of 36 generalized linear models using detection rates of individual species and guilds as dependent variables and the six habitat features as independent variables. Using the mist netting data I constructed a series of 36 generalized linear models that examined relationships between capture rates of individual species or guilds of interest as the dependent variable and six habitat features as independent variables. I used a Poisson distribution in all cases to correct for overdispersion. I tested model sets using an information theoretic approach and Akaike's Information Criterion (AIC, Akaike 1973) with an adjustment for small samples ($AIC_c$, Anderson and Burnham 2002). Models with $\Delta$ $AIC_c < 2$ were considered to be equally plausible in light of the data. Model averaging was performed for all models with $\Delta$ $AIC_c < 2$ on coefficients and standard errors that differed by > 0.01. Analyses were performed with the statistical program R (R Development Core Team 2010 v. 2.12.0, Vienna, Austria).

## RESULTS

### *Vegetation analyses*

Vegetation associated with transects in restoration sites was similar to transects in remnant early successional habitats (see Table 2.4), but had significantly lower canopy height (mean = 4.47 ± 1.25 SE, $p < 0.05$) and narrower bank fullwidth (mean = 3.3 ± 1.33 SE, $p < 0.01$). Restored habitats had significantly lower canopy height, canopy cover, and trees > 23 cm dbh than remant mature forest habitats. There were significant differences in percent forest cover within 500 m between all three habitat types, with restorations having the least forest cover (mean = 9.63 ± 1.79 SE), early successional habitats having intermediate amounts (mean = 21.86 ± 3.44 SE) and mature forest having the most (mean = 35.18 ± 4.35 SE). Numbers of pole-stage trees, percent fruit-bearing shrub cover, woody stem hits, forb stem hits and woody species diversity were not significantly different between any habitat type.

With respect to vegetation associated with mist netting sites, there were no statistical differences between restored and remnant early successional habitats for any habitat feature that I measured (Table 2.5). Restored habitats had significantly higher forb (mean = 32.5 ± 10.91 SE, $p < 0.05$) and woody stem hits (mean = 74.25 ± 13.64 SE, $p < 0.05$) than remnant mature forest habitats, while mature forest habitats had significantly wider bank fullwidth (mean = 103.5 ± 28.83 SE, $p < 0.001$).

*Avian transect surveys*

I recorded 9,744 individual detections of 99 species on transect surveys from late August to late October 2009 and 2010 (Appendix A), representing over 165 hours of survey effort. Transient species (i.e., those not breeding locally) included 28 species of Neotropical migrants and 26 species of temperate migrants. Local breeders included 12 Neotropical migrants, 15 temperate migrants and 18 year-round residents. The five most common species detected on transects included (in decreasing abundance) Golden-crowned Kinglet (*Regulus satrapa*), White-throated Sparrow (*Zonotrichia albicollis*), Myrtle Warbler (*Setophaga coronata coronata*), Ruby-crowned Kinglet (*Regulus calendula*) and Magnolia Warbler (*Setophaga magnolia*).

The top-ranked model explaining transect abundance of Golden-crowned Kinglet included forest cover ($\beta$ = 0.019 ± 0.006 SE), trees 3-8 cm ($\beta$ = 0.031 ± 0.006 SE), trees 8-23 cm ($\beta$ = -0.046 ± 0.022) and trees > 23 cm ($\beta$ = 0.296 ± 0.036) and contained 66.3%

of the Akaike weight; the second-ranked model included the same variables but also woody stem hits ($\beta$ = -0.005 ± 0.013) (Table 2.6). The top-ranked model explaining abundance for White-throated Sparrows included forest cover ($\beta$ = -0.025 ± 0.006), woody stem hits ($\beta$ = 0.039 ± 0.003) and bank fullwidth ("water", $\beta$ = 0.029 ± 0.010) and held 35.1% of the Akaike weight; the second-ranked model included these terms as well as trees > 23 cm ($\beta$ = 0.071 ± 0.023) and held 27.2% of the weight (Table 2.7). There were four models with $\Delta AIC_c < 2$ for Ruby-crowned Kinglet (Akaike weights from 8.6 to 18.1%) with the top three models containing trees > 23 cm ($\beta$ = 0.145 ± 0.043) (Table 2.8). There were three models with $\Delta AIC_c < 2$ for Myrtle Warbler, but all had little support (Akaike weights from 9.7 to 25.4%); all three models included included woody stem hits ($\beta$ = 0.009 ± 0.004), whereas additional variables in the second and third models included water ($\beta$ = -0.006 ± 0.009) and forest cover ($\beta$ = 0.002 ± 0.004), both of which had confidence intervals that overlapped zero (Table 2.9). There were three models with $\Delta AIC_c < 2$ for Magnolia Warbler, but none had strong support (Akaike weights 7.1 to 18.6%); the top-ranked model included only forest cover ($\beta$ = 0.023 ± 0.012), the second-ranked model contained only the intercept, and the third model contained forest cover and trees 8-23 cm ($\beta$ = -0.045 ± 0.088) (Table 2.10).

From transect surveys, the top AIC model for Neotropical migrants contained forest cover ($\beta$ = 0.020 ± 0.005) and woody stem hits ($\beta$ = 0.011 ± 0.005) and held 29.4% of the weight; the second-ranked model (AIC weight 13.2%) contained these terms and trees 8-23 cm ($\beta$ = 0.013 ± 0.015), but confidence interval estimates overlapped zero

(Table 2.11). For temperate migrants the top model contained trees 8-23 cm ($\beta$ = -0.024 ± 0.008), trees > 23 cm ($\beta$ = 0.128 ± 0.011), woody stem hits ($\beta$ = 0.025 ± 0.002) and water ($\beta$ = 0.018 ± 0.005) and held 60.2% of the weight, whereas the second model included these terms and trees 3-8 cm ($\beta$ = 0.003 ± 0.003), holding 26.6% of the Akaike weight (Table 2.12). For forest breeding species the full model held 38.4% of the weight while the second model excluded forest cover and trees 3-8 cm and held 29.7% of the weight (Table 2.13). The top model for early successional breeders contained forest cover ($\beta$ = -0.018 ± 0.004), trees 3-8 cm ($\beta$ = -0.005 ± 0.003), trees 8-23 cm ($\beta$ = -0.037 ± 0.008) and woody stem hits ($\beta$ = 0.024 ± 0.002), holding 41.5% of the Akaike weight (Table 2.14). Obligate frugivores had four top models with weights from 8.5 to 22.4%; all contained woody stem hits ($\beta$ = 0.016 ± 0.003) and water ($\beta$ = 0.033 ± 0.008) (Table 2.15). Transient species diversity estimates had only one top model, containing the terms trees > 23 cm ($\beta$ = 0.026 ± 0.012), woody stem hits ($\beta$ = 0.008 ± 0.002) and water ($\beta$ = 0.016 ± 0.005) and held 27.9% of the Akaike weight (Table 2.16).

After excluding variables with confidence intervals that overlapped zero, variables included in the top models for most species and guilds indicated negative trends for trees 8-23 cm and positive trends for forest cover, woody stem hits and bank fullwidth (Table 2.17). Exceptions to this pattern were the early-successional breeding guild, which was negatively associated with forest cover, and White-throated Sparrow, which had a negative relationship with bank fullwidth. Golden-crowned Kinglet, White-throated Sparrow, Neotropical migrant guild and the forest breeding guild were positively associated with increasing forest cover within 500 m of survey transects, and woody stem hits were positively associated with all guilds as well as White-throated Sparrow and Myrtle Warbler.

There were no differences in detection rates among restoration, mature forest and early successional habitats for most individual species. Golden-crowned Kinglets, however, were least common in restored habitats, whereas White-throated Sparrows and early successional breeders were least common in mature forest habitats (Table 2.18). Although there were no differences in detection rates between restored and remnant habitats for any of the guilds studied, there were consistent trends towards lower diversity in restored habitats, and significantly fewer total birds were detected in restored habitats than in early successional habitats. Differences between restoration and remnant habitats in first-order jackknife estimates of total transient species diversity were not significantly different, although they tended to be higher in early successional habitats.

*Avian mist-net captures*

Over the two fall seasons I captured a total of 2,633 individual birds representing 79 species (Appendix A). Transient species captured (i.e., those not breeding locally) included 30 species of Neotropical migrants and 16 species of temperate migrants. Locally-breeding species captured included 10 species of Neotropical migrants, 11 temperate migrants and 12 year-round residents. The five most common migrant species captured in mist nets included (in decreasing abundance) White-throated Sparrow, Magnolia Warbler, Swainson's Thrush (*Catharus ustulatus*), Golden-crowned Kinglet, and Ruby-crowned Kinglet.

The top-ranked model for White-throated Sparrow capture rates included % Berry ($\beta$ =0.018 ± 0.004 SE), Trees 8-23 cm ($\beta$ = -0.032 ± 0.017 SE) and woody stem hits ($\beta$ = 0.021 ± 0.02), holding 40.7% of the Akaike weight; the second-ranked model was similar, but excluded trees 8-23 cm and held 19.8% of the weight (Table 2.19). There were five models with $\Delta AIC_c < 2$ for Magnolia Warbler, but none had strong support (Akaike weights 5.2 to 12.9%); woody stem hits ($\beta$ = 0.006 ± 0.003) and trees 3-8 cm ($\beta$ = 0.007 ± 0.003) were in 4 of the 5 models (Table 2.20). For Swainson's Thrush, none of the five top models had strong support (Akaike weights 6.5 to 15.0%) and confidence interval estimates for all model parameters contained zero; the top-scoring AIC model contained only the intercept (Table 2.21). The top model for Golden-crowned Kinglet contained % Berry ($\beta$ = 0.005 ± 0.005), trees 3-8 cm ($\beta$ = -0.013 ± 0.005) and water ($\beta$ = 0.004 ± 0.002) and held 39.8% of the Akaike weight; the second-ranked model included these terms in addition to woody stem hits ($\beta$ = 0.006 ± 0.003) and held 15.8% of the Akaike weight (Table 2.22). There were four models with $\Delta AIC_c < 2$ for Ruby-crowned Kinglet, but none had strong support (Akaike weights 9.2 to 19.1%); all models included woody stem hits ($\beta$ = 0.012 ± 0.003), whereas trees > 23 cm ($\beta$ = -0.069 ± 0.052) and water ($\beta$ = -0.010 ± 0.002) were included in the second and third models, respectively (Table 2.23).

There were six models with $\Delta AIC_c < 2$ for Neotropical migrants; no models were strongly supported (Akaike weights 8.7 to 21.7%) but woody stem hits ($\beta = 0.006 \pm 0.001$) and water ($\beta = 0.003 \pm 0.001$) were included in all top models (Table 2.24). Only one top model was apparent for temperate migrants and included terms % Berry ($\beta = 0.008 \pm 0.002$), trees 8-23 cm ($\beta = -0.028 \pm 0.007$) and woody stem hits ($\beta = 0.011 \pm 0.001$), holding 70.7% of the weight (Table 2.25). For forest breeding species the top model contained terms % Berry ($\beta = 0.005 \pm 0.001$), trees 3-8 cm ($\beta = 0.003 \pm 0.001$), trees 8-23 cm ($\beta = -0.016 \pm 0.005$) and woody stem hits ($\beta = 0.008 \pm 0.001$), and held 41.5% of the weight; the second model excluded trees 3-8 cm and held only 16.7% of the weight (Table 2.26). The top model for early successional breeders contained trees 8-23 cm ($\beta = -0.051 \pm 0.007$), trees > 23 cm ($\beta = -0.138 \pm 0.003$), woody stem hits ($\beta = 0.012 \pm 0.001$) and water ($\beta = -0.113 \pm 0.004$) and held 57.5% of the Akaike weight, while the second model included these terms and trees 3-8 cm ($\beta = 0.002 \pm 0.002$) and held 24% of the weight (Table 2.27). Obligate frugivores had three top models with weights from 12 to 25.7%; all contained trees 3-8 cm ($\beta = 0.003 \pm 0.002$), trees 8-23 cm ($\beta = -0.015 \pm 0.005$) and woody stem hits ($\beta = 0.009 \pm 0.001$) (Table 2.28). Transient species diversity estimates had three top models but none had strong support (Akaike weights from 10.1 to 25.4%); all contained trees 3-8 cm ($\beta = 0.026 \pm 0.012$) and woody stem hits ($\beta = 0.008 \pm 0.002$) (Table 2.29).

Woody stem hits were included in at least one top AIC model for all species and guilds (Table 2.30). After excluding variables with confidence intervals that overlapped zero, variables included in the top models for most species and guilds indicated positive trends trends for % Berry, trees 3-8 cm and woody stem hits while trends for trees 8-23 cm and water were negative. Magnolia Warbler and Neotropical migrants however were negatively associated with % Berry, and Golden-crowned Kinglets were negatively associated with trees 3-8 cm. Trees > 23 cm was included in top models for only Ruby-crowned Kinglet and early successional breeding species, and in both cases the relationship was negative.

In almost all cases capture rates were significantly higher in early successional habitats than those in either restored or mature forest habitats (Table 2.31). Exceptions to this pattern included Swainson's Thrush and Golden-crowned Kinglet, for which there was no difference in capture rates between habitat types. There was no difference in capture rates for any species or guild between restored and mature forest habitats.

Overall these results suggest that capture rates for most species and guilds I examined were negatively associated with pole stage trees and bank fullwidth and positively associated with woody stem hits; most species and guilds were positively associated with fruit-bearing shrub cover.

## DISCUSSION

My study suggests that in fall, stopover migrant landbirds respond to habitat attributes at small and large spatial scales. At fine scales, structurally complex habitats with high woody stem and fruiting shrub densities contained the highest numbers of migrants. At a larger spatial scale, increasing amounts of forest cover within 500 m of surveys was associated with higher abundances of several species and guilds, while it was negatively associated with early successional breeding species. Although some of these patterns are consistent with previous studies conducted on stopover habitat use during fall migration, my study is one of very few to examine habitat use on a larger spatial scale and demonstrate this pattern in the context of restored habitats within an agricultural landscape.

Irregardless of guild or species, most migrants were more common in structurally complex habitats with high woody stem densities, and negatively associated with pole-stage (8-23 cm dbh) trees; such sites had low structural heterogeneity (both vertical and horizontal). Most species and guilds were more common in areas containing more fruit-bearing shrub cover. Use of fruit resources in such areas was evidenced by frequent occurrence of grape seeds and dogwood carps in feces of captured Gray Catbirds (*Dumetella carolinensis*), *Catharus* thrushes, Cedar Waxwings (*Bombycilla cedrorum*) and vireos (*Vireo* spp., E. Cashion unpublished data). Overall these results support findings that both structural complexity of habitats in restorations is a strong predictor of migrant abundance (Smiley et al. 2007) and that fruit resources may play a major role in stopover habitat choice (Martin and Karr 1986).

Similar to the breeding bird study conducted by Perkins et al. (2003), responses to forest cover within 500 m depended on the ecological requirements of the species and guilds I studied, with early-successional breeding species having a negative association while interior forest breeding species had a positive association. In this way, my study provides evidence that large-scale features such as amount of mature forest cover surrounding a site may be an important cue for some migrants, similar to other studies conducted during the migratory period (Buler et al. 2007, Packett and Dunning 2009).

I expected restoration habitats to have similar migrant abundances as early successional shrub-sapling habitats at similar stages of succession and thus serve the same ecological function for stopover migrants. Despite being vegetatively similar, capture rates for most species and all guilds were much lower in restoration habitats than in remnant habitats, while transect detection rates tended not to differ. This may be partly because restoration habitats in my study area frequently bordered other habitat types such as fencerows or mature forest edges, where fall stopover migrants tend to congregate (Rodewald and Brittingham 2004) and my transects were often located along such borders. In addition, the wide variance in detection rates in restored habitats may have obscured differences between restored and remnant habitats. Differences in capture rates between restored and remant habitats may have also resulted from mist net placement in areas of thicker shrub density and the fact that we did not sample the forest canopy with mist nets.

It is likely that migrants select habitats using more than vegetation characteristics and resource abundance as selection cues, particularly when that information is not available. For example, migrants may make stopover decisions by using social information in mixed-species flocks to determine food availability and predation risk (Nemeth and Moore 2007). Similarly, mixed species flocks containing parid species may provide cues to stopover migrants about habitat quality and resource abundance (Moore and Aborn 2000, Rodewald and Brittingham 2002). Further, simulated call notes from conspecifics during the post-breeding period were demonstrated to take priority over habitat structure in breeding habitat selection in Black-throated Blue Warblers (*Setophaga caerulescens*, Betts et al. 2008). Mukhin et al. (2008) demonstrated that habitat specialists Eurasian Reed Warblers (*Acrocephalus scirpaceus*) and Sedge Warblers (*Acrocephalus schoenobaenus*) respond strongly to simulated conspecific signals during the settlement period at a shoreline stopover site on the Baltic Sea. During spring migration in Illinois Yellow-breasted Chats (*Icteria virens*) selected suboptimal habitats based on simulated social cues during the pre-settlement period, and there was some evidence that other shrubland breeding species were responding as well (Alessi et al. 2010). Given this evidence it is possible that low occupancy rates in CRP restorations could in part result from of a paucity of conspecific signals in those sites.

Habitat relationships have dominated the focus of migrant landbird conservation efforts, but little has been done to understand habitat selection processes and energetics in restored and remnant habitats within agricultural landscapes during stopover. Previous research has demonstrated that birds will preferentially settle at stopover sites that have an abundance of food resources (reviewed in Moore et al. 1995), so it follows that avian abundance may be an indicator for habitat quality (Rodewald and Brittingham 2004).

However, some of these habitats may not be suitable despite the abundance of birds they contain (Van Horne 1983, Donovan and Thompson 2001), so resource abundance and the energetic condition of captured migrants should be assessed. Data on resource abundance and the energetic condition of migrants were collected for my study, however sample sizes of captured migrants were insufficient to lend power to the analysis. Such research will become increasingly important as remnant habitats are modified for human use.

Although the CRP is often regarded as a panacea for a variety of environmental ailments, many challenges remain, among them a renewed interest in biofuels and the application of appropriate disturbance and management regimes to CRP lands. For example, Grasshopper Sparrows (*Ammodramus savannarum*) in Maryland were absent from CRP grassland plots that were not treated with fire or herbicide within the last 2-3 years (Gill et al. 2006). This complements the finding that species richness of breeding migratory passerines is greater in CRP plots with appropriate disturbance regimes, such as periodic fire and herbicides, suggesting that CRP will need continued management to provide maximum benefit to wildlife (Sladek et al. 2008).

Although early successional shrub-sapling habitats have been viewed as unaesthetic and of little conservation value in comparison to mature woodland (Askins 2001), loss of early successional forests has had negative consequences for birds that breed in these habitats (North American Bird Conservation Initiative 2009). In the lower Midwest, only 4-7% of the landscape is seedling-sapling stage (Trani et al. 2001), suggesting that shrub-sapling habitat could also be a limiting resource for birds migrating through this area. In my study, birds were most abundant in early successional habitats, which supports findings from other migration studies (Rodewald and Brittingham 2004, Smith and Hatch 2008).

If habitat restorations are to effectively provide stopover habitat, they must be actively managed to provide specific attributes favored by migrants. In the short-term, restoration efforts did not often attract large numbers of fall migrants, but this is expected to change with succession. Increases in structural complexity of habitats should improve the ability of restorations to provide stopover habitat for migrants. At present, habitat restorations in the study area may at least serve as the "fire escapes" or "convienience stores" proposed by Mehlman et al. (2005) and within a decade could achieve "full service hotel" status with effective management practices.

Because both local and landscape attributes were important predictors of use by migrating birds, my findings illustrate that effective strategies to provide stopover habitat must work across multiple spatial scales. Although young restoration habitats in the study area were vegetatively similar to many remnant shrub-sapling habitats, capture and detection rates for several guilds and species were lower in restorations. How this decreased abundance is related to vegetative structure within the restorations is unclear, however this study has demonstrated that most guilds and species were more abundant in structurally complex habitats with higher woody stem densities and fruiting shrub cover. Benefits of riparian forest restorations to stopover migrant landbirds can be improved if managers 1) incorporate fruit-bearing shrubs, 2) manage for high woody plant species diversity, and 3) promote strategies to increase forest cover within the matrix surrounding restoration sites, or create restorations within landscapes that have higher percentages of forest cover.

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| Site | Latitude N | Longitude W | Sampling | #Nets/ Transects | Net hours/ minutes | Habitat | Site description |
|---|---|---|---|---|---|---|---|
| 1 | 40.954 | -83.82 | Transects | 1 | 293 | RES | Dave Reese's property, CREP |
| 5 | 40.939 | -83.671 | Transects | 2 | 509 | RES | Riparian buffer strip, CREP (CP22) |
| 7 | 40.904 | -83.608 | Transects | 2 | 476 | RES | 32 ha CRP |
| 21.1 | 41.02 | -83.691 | Transects | 2 | 554 | RES | Oak Woods County Park, non-CRP restoration |
| 33 | 40.869 | -83.593 | Transects | 1 | 199 | RES | Riparian buffer strip, CREP (CP22) |
| 35 | 40.817 | -83.911 | Transects | 3 | 750 | RES | Tree farm adjacent to mature riparian corridor, CRP |
| 36.3 | 40.737 | -83.885 | Transects | 3 | 715 | RES | Tree farm, CRP |
| 41 | 41.125 | -83.609 | Transects | 2 | 509 | RES | Late pole stage, CRP |
| 18S | 41.053 | -83.767 | Transects | 2 | 221 | RES | Litzenburg Memorial Woods (South), CRP |
| 12.2 | 40.952 | -83.549 | Transects | 1 | 303 | ES | Rieck Center, adjacent to riparian corridor |
| 13.1 | 41.032 | -83.556 | Transects | 1 | 303 | ES | Riverbend County Park |
| 32 | 40.871 | -83.679 | Transects | 1 | 261 | ES | So-Han-Co Sportsmen's Club, old field |
| 11A | 40.925 | -83.557 | Transects | 1 | 262 | ES | Remnant shrub-scrub, nearly pole stage |
| 18S | 41.053 | -83.767 | Transects | 2 | 266 | ES | Litzenburg Memorial Woods (South) |
| 25A | 41.133 | -83.639 | Transects | 1 | 293 | ES | Van Buren State Park (West) |

Table 2.1. List of field site coordinates, avian sampling that took place and description of sites. RES sites were habitats restored through the Conservation Reserve Program (CRP) or the Conservation Reserve Enhancement Program (CREP); ES habitats were naturally succeeded (i.e. shrub-sapling) habitats; MF sites were remnant mature forest habitats.

Table 2.1 continued.

| Site | Latitude N | Longitude W | Sampling | #Nets/ Transects | Net hours/ minutes | Habitat | Description |
|---|---|---|---|---|---|---|---|
| 11 | 40.847 | -83.557 | Transects | 1 | 283 | ES | Adjacent to woodlot edge |
| 2 | 40.97 | -83.796 | Transects | 2 | 495 | MF | Located on Ottawa Creek |
| 6 | 40.96 | -83.659 | Transects | 1 | 248 | MF | Camp Berry (Boy Scouts of America property) |
| 12.1 | 40.952 | -83.549 | Transects | 1 | 286 | MF | Rieck Center, inside mature forest edge |
| 13.3 | 41.032 | -83.556 | Transects | 2 | 502 | MF | Riverbend County Park |
| 21.2 | 41.02 | -83.691 | Transects | 2 | 521 | MF | Oak Woods County Park |
| 36.1 | 40.737 | -83.885 | Transects | 1 | 235 | MF | Adjacent to CRP tree farm |
| 42 | 40.864 | -83.682 | Transects | 1 | 248 | MF | Late pole stage forest |
| 18N | 41.063 | -83.763 | Transects | 2 | 449 | MF | Litzenburg Memorial Woods (North), mature |
| 25B | 41.132 | -83.619 | Transects | 3 | 540 | MF | Van Buren State Park (East), mature forest |
| 7 | 40.904 | -83.608 | Mist-netting | 8 | 1014 | RES | 32 ha CRP |
| 36 | 40.737 | -83.885 | Mist-netting | 8 | 1108 | RES | Tree farm, CRP |
| 1 | 40.954 | -83.82 | Mist-netting | 8 | 1112 | ES | Dave Reese's property, pole right-of-way |
| 12 | 40.952 | -83.549 | Mist-netting | 8 | 1192 | ES | Rieck Center |
| 6 | 40.96 | -83.659 | Mist-netting | 8 | 1142 | MF | Camp Berry (Boy Scouts of America property) |
| 25A | 41.133 | -83.639 | Mist-netting | 8 | 1107 | MF | Van Buren State Park (West) |

| | % Forest cover | Canopy height | % Canopy cover | Trees 3-8 cm | Trees 8-23 cm | Trees > 23 cm | % Berry | Woody diversity | Woody stem hits | Forb stem hits | Water |
|---|---|---|---|---|---|---|---|---|---|---|---|
| % Forest cover | 1.00 | 0.70 | 0.47 | -0.13 | 0.24 | 0.52 | 0.14 | 0.11 | -0.10 | -0.46 | 0.51 |
| Canopy height (m) | 0.70 | 1.00 | 0.81 | -0.06 | 0.42 | 0.89 | 0.17 | 0.46 | -0.06 | -0.49 | 0.43 |
| % Canopy cover | 0.47 | 0.81 | 1.00 | 0.10 | 0.65 | 0.87 | 0.05 | 0.53 | -0.04 | -0.45 | 0.30 |
| Trees 3-8 cm dbh | -0.13 | -0.06 | 0.10 | 1.00 | 0.23 | -0.16 | 0.16 | 0.05 | 0.36 | -0.13 | 0.09 |
| Trees 8-23 cm dbh | 0.24 | 0.42 | 0.65 | 0.23 | 1.00 | 0.46 | 0.29 | 0.49 | 0.20 | -0.33 | 0.37 |
| Trees > 23 cm dbh | 0.52 | 0.89 | 0.87 | -0.16 | 0.46 | 1.00 | 0.04 | 0.51 | -0.18 | -0.31 | 0.31 |
| % Berry | 0.14 | 0.17 | 0.05 | 0.16 | 0.29 | 0.04 | 1.00 | 0.49 | 0.73 | -0.19 | 0.27 |
| Woody diversity | 0.11 | 0.46 | 0.53 | 0.05 | 0.49 | 0.51 | 0.49 | 1.00 | 0.28 | -0.34 | 0.34 |
| Woody stem hits | -0.10 | -0.06 | -0.04 | 0.36 | 0.20 | -0.18 | 0.73 | 0.28 | 1.00 | -0.11 | -0.03 |
| Forb stem hits | -0.46 | -0.49 | -0.45 | -0.13 | -0.33 | -0.31 | -0.19 | -0.34 | -0.11 | 1.00 | -0.21 |
| Water | 0.51 | 0.43 | 0.30 | 0.09 | 0.37 | 0.31 | 0.27 | 0.34 | -0.03 | -0.21 | 1.00 |

Table 2.2. Correlation matrix ($r^2$ values) for habitat features associated with transect survey locations in Hancock an Allen counties, northwest Ohio, from late August to late October 2009 and 2010.

| | Canopy height | % Canopy cover | Trees 3-8 cm | Trees 8-23 cm | Trees > 23 cm | % Berry | Woody diversity | Woody stem hits | Forb stem hits | Water |
|---|---|---|---|---|---|---|---|---|---|---|
| Canopy height (m) | 1 | 0.85 | -0.58 | 0.28 | 0.83 | 0.38 | 0.33 | 0.13 | -0.83 | 0.73 |
| % Canopy cover | 0.85 | 1 | -0.51 | 0.4 | 0.77 | 0.36 | 0.39 | 0.17 | -0.77 | 0.79 |
| Trees 3-8 cm dbh | -0.58 | -0.51 | 1 | -0.46 | -0.31 | -0.38 | -0.41 | -0.2 | 0.41 | -0.32 |
| Trees 8-23 cm dbh | 0.28 | 0.4 | -0.46 | 1 | 0.13 | 0.09 | 0.18 | 0.26 | -0.39 | 0.12 |
| Trees > 23 cm dbh | 0.83 | 0.77 | -0.31 | 0.13 | 1 | 0.23 | 0.25 | 0.01 | -0.77 | 0.79 |
| % Berry | 0.38 | 0.36 | -0.38 | 0.09 | 0.23 | 1 | 0.6 | 0.36 | -0.4 | 0.2 |
| Woody diversity | 0.33 | 0.39 | -0.41 | 0.18 | 0.25 | 0.6 | 1 | 0.03 | -0.35 | 0.22 |
| Woody stem hits | 0.13 | 0.17 | -0.2 | 0.26 | 0.01 | 0.36 | 0.03 | 1 | -0.14 | -0.01 |
| Forb stem hits | -0.83 | -0.77 | 0.41 | -0.39 | -0.77 | -0.4 | -0.35 | -0.14 | 1 | -0.62 |
| Water | 0.73 | 0.79 | -0.32 | 0.12 | 0.79 | 0.2 | 0.22 | -0.01 | -0.62 | 1 |

Table 2.3. Correlation matrix ($r^2$ values) for habitat features associated with 6 mist-netting sites in Hancock and Allen counties, northwest Ohio, from late August to late October 2009 and 2010.

| Variable | Early successional | Restoration | Mature forest | F | *p* |
|---|---|---|---|---|---|
| Forest cover | 21.86 (3.44) a* | 9.63 (1.79) b*** | 35.18 (4.35) c* | 15.36 | < 0.001*** |
| Canopy height (m) | 10.05 (1.95) a* | 4.47 (1.25) b*** | 19.22 (1.34) c*** | 26.59 | < 0.001*** |
| % Canopy cover | 54.17 (8.8) | 33.84 (9.02) b*** | 81.25 (7.36) | 8.55 | 0.001*** |
| Trees 3-8 cm dbh | 36.33 (7.89) | 18.88 (4.94) | 14.15 (2.22) c* | 4.81 | 0.017* |
| Trees 8-23 cm dbh | 13.29 (2.18) | 11.01 (2.35) | 13.83 (1.52) | 0.57 | 0.573 |
| Trees > 23 cm dbh | 2.83 (0.99) | 2.06 (0.78) b*** | 7.92 (0.74) c*** | 15.68 | < 0.001*** |
| % Berry | 18.6 (6.17) | 8.59 (3.15) | 6.88 (1.96) | 2.56 | 0.098 |
| Woody stem hits | 37.16 (7.7) | 26.09 (5.23) | 21.03 (3.44) | 2.16 | 0.136 |
| Forb stem hits | 27.08 (11.01) | 26.04 (7.94) | 11.27 (3.53) | 1.38 | 0.271 |
| Water | 15.12 (2.75) a** | 3.3 (1.33) | 10.1 (2.08) | 8.17 | 0.002** |
| Woody diversity | 12.5 (1.59) | 11.9 (2.35) | 13.4 (0.54) | 0.21 | 0.809 |

Table 2.4. Comparison of habitat means and standard errors associated with transect vegetation plots by habitat type. Pairwise comparisons: a = Early successional vs. Restoration, b = Restoration vs. Mature forest, c = Mature forest vs. Early successional; *** = $p < 0.001$, ** = $p < 0.01$, * = $p < 0.05$.

| Variable | Early successional | Restoration | Mature forest | F | *p* |
|---|---|---|---|---|---|
| Canopy height | 5.04 (0.93) | 3.71 (0.33) b*** | 18.6 (0.95) c*** | 10.66 | < 0.001*** |
| % Canopy cover | 54.58 (7.82) | 34.17 (7.46) b*** | 90 (3.25) c** | 16.287 | < 0.001*** |
| Trees 3-8 cm dbh | 49.58 (10.44) | 39.5 (10.34) | 20.8 (3.9) | 2.381 | 0.109 |
| Trees 8-23 cm dbh | 11.42 (1.64) | 13.5 (3.3) | 12.7 (3.61) | 0.138 | 0.872 |
| Trees > 23 cm dbh | 1.0 (0.41) | 0.0 (0.0) b*** | 6.8 (1.34) c*** | 24.159 | < 0.001*** |
| % Berry | 31.92 (9.59) | 11 (2.43) | 30.3 (9.65) | 2.313 | 0.116 |
| Woody stem hits | 74.25 (13.64) | 46.08 (6.98) | 38.6 (4.09) c* | 3.835 | 0.0325* |
| Forb stem hits | 32.5 (10.91) | 26.58 (4.46) | 2.1 (0.74) c* | 4.658 | 0.017* |
| Water | 10.5 (1.36) | 1.58 (0.15) b*** | 103.5 (28.83) c*** | 13.942 | < 0.001*** |
| Woody diversity | 7.25 (1.27) | 6.33 (1.3) | 8.6 (1.71) | 0.623 | 0.543 |

Table 2.5. Comparison of habitat means and standard errors associated with mist-netting vegetation plots by habitat type. Pairwise comparisons: a = Early successional vs. Restoration, b = Restoration vs. Mature forest, c = Mature forest vs. Early successional; *** = $p < 0.001$, ** = $p < 0.01$, * = $p < 0.05$.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 204.48 | 0.00 | 0.66 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 206.36 | 1.87 | 0.26 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 208.84 | 4.35 | 0.08 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 215.94 | 11.46 | 0.00 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 219.55 | 15.07 | 0.00 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 220.98 | 16.49 | 0.00 |
| ForestCover + Trees23 | 3 | 222.16 | 17.68 | 0.00 |
| Trees23 + WoodyStems + Water | 4 | 228.55 | 24.07 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 230.75 | 26.27 | 0.00 |
| Trees23 + WoodyStems | 3 | 233.24 | 28.75 | 0.00 |
| Trees23 | 2 | 234.50 | 30.01 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 235.16 | 30.67 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 235.37 | 30.88 | 0.00 |
| Trees8.23 + Trees23 | 3 | 236.67 | 32.19 | 0.00 |
| ForestCover + Trees8.23 | 3 | 283.65 | 79.17 | 0.00 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 285.23 | 80.74 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 285.56 | 81.08 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 287.28 | 82.80 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 287.59 | 83.11 | 0.00 |
| ForestCover | 2 | 311.48 | 107.00 | 0.00 |
| ForestCover + Trees3.8 | 3 | 312.39 | 107.90 | 0.00 |
| ForestCover + Water | 3 | 313.36 | 108.88 | 0.00 |
| ForestCover + WoodyStems | 3 | 313.74 | 109.26 | 0.00 |
| ForestCover + Trees3.8 + Water | 4 | 314.75 | 110.27 | 0.00 |
| ForestCover + WoodyStems + Water | 4 | 315.71 | 111.23 | 0.00 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 316.99 | 112.50 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 367.63 | 163.15 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 385.51 | 181.02 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 386.31 | 181.82 | 0.00 |
| Trees8.23 | 2 | 386.54 | 182.06 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 402.35 | 197.87 | 0.00 |
| Water | 2 | 405.87 | 201.39 | 0.00 |
| WoodyStems + Water | 3 | 406.50 | 202.01 | 0.00 |
| (Intercept) | 1 | 422.75 | 218.27 | 0.00 |
| WoodyStems | 2 | 423.44 | 218.95 | 0.00 |
| Trees3.8 | 2 | 424.40 | 219.92 | 0.00 |

Table 2.6. AIC models for Golden-crowned Kinglet detection rates. $k$ = the number of parameters in each model; $AIC_c$ = Akaike's information criterion adjusted for small sample sizes; $\Delta AIC_c$ is the difference in $AIC_c$ between the top-ranked model and the model in question; $\omega_i$ is the weight of evidence indicating the relative likelihood of the model.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| ForestCover + WoodyStems + Water | 4 | 274.57 | 0.00 | 0.80 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 279.11 | 4.54 | 0.08 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 281.15 | 6.58 | 0.03 |
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 281.61 | 7.04 | 0.02 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 282.00 | 7.44 | 0.02 |
| ForestCover + WoodyStems | 3 | 282.49 | 7.93 | 0.02 |
| Trees23 + WoodyStems | 3 | 283.05 | 8.49 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 284.48 | 9.91 | 0.01 |
| Trees23 + WoodyStems + Water | 4 | 285.16 | 10.59 | 0.00 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 285.62 | 11.05 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 286.55 | 11.99 | 0.00 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 286.66 | 12.10 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 286.99 | 12.42 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 287.96 | 13.39 | 0.00 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 288.13 | 13.56 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 288.74 | 14.18 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 289.06 | 14.49 | 0.00 |
| WoodyStems | 2 | 289.28 | 14.71 | 0.00 |
| WoodyStems + Water | 3 | 291.13 | 16.56 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 419.58 | 145.01 | 0.00 |
| ForestCover + Trees3.8 + Water | 4 | 430.65 | 156.08 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 431.75 | 157.19 | 0.00 |
| ForestCover + Water | 3 | 434.43 | 159.87 | 0.00 |
| Water | 2 | 436.08 | 161.52 | 0.00 |
| Trees23 | 2 | 436.22 | 161.66 | 0.00 |
| (Intercept) | 1 | 437.27 | 162.71 | 0.00 |
| Trees8.23 | 2 | 442.50 | 167.93 | 0.00 |
| Trees3.8 | 2 | 444.29 | 169.73 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 453.47 | 178.90 | 0.00 |
| ForestCover + Trees23 | 3 | 454.60 | 180.03 | 0.00 |
| Trees8.23 + Trees23 | 3 | 462.79 | 188.22 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 463.11 | 188.54 | 0.00 |
| ForestCover | 2 | 463.14 | 188.57 | 0.00 |
| ForestCover + Trees3.8 | 3 | 463.35 | 188.78 | 0.00 |
| ForestCover + Trees8.23 | 3 | 464.92 | 190.36 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 465.51 | 190.94 | 0.00 |

Table 2.7. AIC models for White-throated Sparrow detection rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| ForestCover + Trees23 | 3 | 101.44 | 0.00 | 0.18 |
| Trees8.23 + Trees23 | 3 | 101.81 | 0.36 | 0.15 |
| Trees23 | 2 | 102.94 | 1.50 | 0.09 |
| ForestCover | 2 | 102.95 | 1.50 | 0.09 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 103.51 | 2.07 | 0.06 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 103.51 | 2.07 | 0.06 |
| Trees8.23 + Trees23 + Water | 4 | 104.10 | 2.66 | 0.05 |
| ForestCover + Trees3.8 | 3 | 104.60 | 3.15 | 0.04 |
| ForestCover + Water | 3 | 104.65 | 3.21 | 0.04 |
| Trees23 + WoodyStems | 3 | 105.00 | 3.55 | 0.03 |
| ForestCover + Trees8.23 | 3 | 105.01 | 3.57 | 0.03 |
| ForestCover + WoodyStems | 3 | 105.18 | 3.73 | 0.03 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 105.32 | 3.88 | 0.03 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 105.65 | 4.21 | 0.02 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 105.75 | 4.30 | 0.02 |
| ForestCover + Trees3.8 + Water | 4 | 106.79 | 5.35 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 106.91 | 5.46 | 0.01 |
| ForestCover + WoodyStems + Water | 4 | 107.00 | 5.56 | 0.01 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 107.30 | 5.86 | 0.01 |
| Trees23 + WoodyStems + Water | 4 | 107.33 | 5.88 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 107.88 | 6.44 | 0.01 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 108.25 | 6.80 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 109.11 | 7.66 | 0.00 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 109.12 | 7.68 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 109.20 | 7.76 | 0.00 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 109.50 | 8.06 | 0.00 |
| (Intercept) | 1 | 112.60 | 11.16 | 0.00 |
| Trees3.8 | 2 | 112.95 | 11.51 | 0.00 |
| Water | 2 | 114.34 | 12.90 | 0.00 |
| WoodyStems | 2 | 114.70 | 13.26 | 0.00 |
| Trees8.23 | 2 | 114.76 | 13.32 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 115.11 | 13.67 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 115.86 | 14.42 | 0.00 |
| WoodyStems + Water | 3 | 116.55 | 15.10 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 117.46 | 16.01 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 118.33 | 16.88 | 0.00 |

Table 2.8. AIC models for Ruby-crowned Kinglet detection rates. See Table 2.6 for column definitions.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| WoodyStems | 2 | 300.10 | 0.00 | 0.25 |
| WoodyStems + Water | 3 | 301.88 | 1.78 | 0.10 |
| ForestCover + WoodyStems | 3 | 302.02 | 1.92 | 0.10 |
| Trees23 + WoodyStems | 3 | 302.31 | 2.21 | 0.08 |
| ForestCover + WoodyStems + Water | 4 | 303.31 | 3.21 | 0.05 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 303.53 | 3.43 | 0.05 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 303.74 | 3.64 | 0.04 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 304.03 | 3.93 | 0.04 |
| Trees23 + WoodyStems + Water | 4 | 304.17 | 4.07 | 0.03 |
| (Intercept) | 1 | 304.18 | 4.08 | 0.03 |
| Trees3.8 + WoodyStems + Water | 4 | 304.22 | 4.12 | 0.03 |
| Water | 2 | 305.49 | 5.39 | 0.02 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 305.73 | 5.63 | 0.02 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 305.76 | 5.66 | 0.02 |
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 305.91 | 5.81 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 305.99 | 5.89 | 0.01 |
| Trees8.23 | 2 | 306.16 | 6.06 | 0.01 |
| Trees3.8 | 2 | 306.25 | 6.15 | 0.01 |
| Trees23 | 2 | 306.31 | 6.22 | 0.01 |
| ForestCover | 2 | 306.31 | 6.22 | 0.01 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 306.35 | 6.25 | 0.01 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 306.64 | 6.54 | 0.01 |
| ForestCover + Water | 3 | 307.29 | 7.19 | 0.01 |
| Trees3.8 + Trees8.23 | 3 | 308.28 | 8.18 | 0.00 |
| ForestCover + Trees8.23 | 3 | 308.33 | 8.23 | 0.00 |
| Trees8.23 + Trees23 | 3 | 308.42 | 8.32 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 308.44 | 8.34 | 0.00 |
| ForestCover + Trees23 | 3 | 308.46 | 8.36 | 0.00 |
| ForestCover + Trees3.8 | 3 | 308.47 | 8.37 | 0.00 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 308.56 | 8.46 | 0.00 |
| ForestCover + Trees3.8 + Water | 4 | 308.72 | 8.62 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 310.07 | 9.97 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 310.43 | 10.34 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 310.51 | 10.42 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 311.01 | 10.91 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 312.99 | 12.89 | 0.00 |

Table 2.9. AIC models for Myrtle Warbler detection rates. See Table 2.6 for column definitions.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| ForestCover | 2 | 84.71 | 0.00 | 0.19 |
| (Intercept) | 1 | 86.19 | 1.47 | 0.09 |
| ForestCover + Trees8.23 | 3 | 86.64 | 1.93 | 0.07 |
| ForestCover + Water | 3 | 86.84 | 2.13 | 0.06 |
| ForestCover + WoodyStems | 3 | 86.95 | 2.24 | 0.06 |
| ForestCover + Trees23 | 3 | 86.95 | 2.24 | 0.06 |
| ForestCover + Trees3.8 | 3 | 86.98 | 2.27 | 0.06 |
| Trees23 | 2 | 87.11 | 2.39 | 0.06 |
| Trees8.23 | 2 | 87.59 | 2.88 | 0.04 |
| Water | 2 | 88.17 | 3.46 | 0.03 |
| Trees3.8 | 2 | 88.29 | 3.58 | 0.03 |
| WoodyStems | 2 | 88.35 | 3.64 | 0.03 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 88.98 | 4.27 | 0.02 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 89.00 | 4.29 | 0.02 |
| ForestCover + Trees3.8 + Water | 4 | 89.16 | 4.44 | 0.02 |
| ForestCover + WoodyStems + Water | 4 | 89.18 | 4.47 | 0.02 |
| Trees8.23 + Trees23 | 3 | 89.20 | 4.48 | 0.02 |
| Trees23 + WoodyStems | 3 | 89.36 | 4.64 | 0.02 |
| Trees3.8 + Trees8.23 | 3 | 89.69 | 4.98 | 0.02 |
| WoodyStems + Water | 3 | 90.43 | 5.72 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 91.27 | 6.55 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 91.43 | 6.72 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 91.44 | 6.72 | 0.01 |
| Trees8.23 + Trees23 + Water | 4 | 91.51 | 6.80 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 91.56 | 6.85 | 0.01 |
| Trees23 + WoodyStems + Water | 4 | 91.62 | 6.91 | 0.01 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 91.63 | 6.91 | 0.01 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 91.63 | 6.92 | 0.01 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 92.06 | 7.34 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 92.57 | 7.86 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 93.98 | 9.26 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 94.01 | 9.29 | 0.00 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 94.10 | 9.38 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 94.34 | 9.62 | 0.00 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 96.54 | 11.82 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 96.55 | 11.84 | 0.00 |

Table 2.10. AIC models for Magnolia Warbler detection rates. See Table 2.6 for column definitions.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| ForestCover + WoodyStems | 3 | 188.93 | 0.00 | 0.29 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 190.52 | 1.59 | 0.13 |
| ForestCover + WoodyStems + Water | 4 | 191.05 | 2.12 | 0.10 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 191.30 | 2.37 | 0.09 |
| ForestCover | 2 | 192.37 | 3.44 | 0.05 |
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 192.96 | 4.03 | 0.04 |
| ForestCover + Trees8.23 | 3 | 193.18 | 4.25 | 0.04 |
| ForestCover + Trees3.8 | 3 | 193.30 | 4.37 | 0.03 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 193.52 | 4.59 | 0.03 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 193.70 | 4.77 | 0.03 |
| Trees23 + WoodyStems + Water | 4 | 193.76 | 4.83 | 0.03 |
| ForestCover + Trees23 | 3 | 193.76 | 4.83 | 0.03 |
| ForestCover + Water | 3 | 194.46 | 5.53 | 0.02 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 194.73 | 5.80 | 0.02 |
| Trees23 + WoodyStems | 3 | 194.96 | 6.03 | 0.01 |
| ForestCover + Trees3.8 + Water | 4 | 195.61 | 6.68 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 195.95 | 7.02 | 0.01 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 196.18 | 7.25 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 196.21 | 7.28 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 196.31 | 7.38 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 197.14 | 8.21 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 197.24 | 8.31 | 0.00 |
| Trees23 | 2 | 198.11 | 9.17 | 0.00 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 198.71 | 9.78 | 0.00 |
| Trees8.23 + Trees23 | 3 | 199.91 | 10.98 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 200.02 | 11.09 | 0.00 |
| WoodyStems + Water | 3 | 200.10 | 11.17 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 201.44 | 12.51 | 0.00 |
| Water | 2 | 201.48 | 12.54 | 0.00 |
| WoodyStems | 2 | 201.72 | 12.79 | 0.00 |
| Trees8.23 | 2 | 201.75 | 12.82 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 202.41 | 13.48 | 0.00 |
| (Intercept) | 1 | 202.78 | 13.85 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 203.75 | 14.82 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 203.79 | 14.86 | 0.00 |
| Trees3.8 | 2 | 204.41 | 15.48 | 0.00 |

Table 2.11. AIC models for Neotropical migrant detection rates. See Table 2.6 for column definitions.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 385.05 | 0.00 | 0.60 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 386.69 | 1.64 | 0.27 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 389.32 | 4.26 | 0.07 |
| Trees23 + WoodyStems + Water | 4 | 391.47 | 6.42 | 0.02 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 392.13 | 7.08 | 0.02 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 393.88 | 8.83 | 0.01 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 393.95 | 8.89 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 395.23 | 10.17 | 0.00 |
| Trees23 + WoodyStems | 3 | 397.51 | 12.46 | 0.00 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 479.61 | 94.56 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 482.82 | 97.77 | 0.00 |
| ForestCover + WoodyStems | 3 | 486.48 | 101.42 | 0.00 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 486.93 | 101.88 | 0.00 |
| ForestCover + WoodyStems + Water | 4 | 488.71 | 103.65 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 491.42 | 106.37 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 494.46 | 109.40 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 511.12 | 126.07 | 0.00 |
| WoodyStems + Water | 3 | 516.18 | 131.13 | 0.00 |
| WoodyStems | 2 | 520.82 | 135.77 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 528.24 | 143.19 | 0.00 |
| Trees23 | 2 | 555.59 | 170.54 | 0.00 |
| ForestCover + Trees23 | 3 | 556.93 | 171.88 | 0.00 |
| Trees8.23 + Trees23 | 3 | 556.93 | 171.88 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 559.26 | 174.21 | 0.00 |
| ForestCover + Trees8.23 | 3 | 618.10 | 233.04 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 619.57 | 234.51 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 620.17 | 235.11 | 0.00 |
| ForestCover | 2 | 623.15 | 238.09 | 0.00 |
| ForestCover + Trees3.8 | 3 | 623.26 | 238.20 | 0.00 |
| ForestCover + Trees3.8 + Water | 4 | 624.16 | 239.10 | 0.00 |
| ForestCover + Water | 3 | 625.32 | 240.26 | 0.00 |
| Trees8.23 | 2 | 627.05 | 242.00 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 629.02 | 243.97 | 0.00 |
| (Intercept) | 1 | 636.36 | 251.30 | 0.00 |
| Water | 2 | 636.53 | 251.47 | 0.00 |
| Trees3.8 | 2 | 637.46 | 252.41 | 0.00 |

Table 2.12. AIC models for temperate migrant detection rates. See Table 2.6 for column definitions.

| | *k* | AIC$_C$ | ΔAIC$_C$ | $\omega_i$ |
|---|---|---|---|---|
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 387.93 | 0.00 | 0.38 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 388.44 | 0.51 | 0.30 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 389.33 | 1.40 | 0.19 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 391.00 | 3.07 | 0.08 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 393.44 | 5.52 | 0.02 |
| Trees23 + WoodyStems + Water | 4 | 394.69 | 6.76 | 0.01 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 395.42 | 7.49 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 409.64 | 21.71 | 0.00 |
| Trees23 + WoodyStems | 3 | 410.67 | 22.74 | 0.00 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 466.09 | 78.16 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 470.33 | 82.40 | 0.00 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 475.92 | 87.99 | 0.00 |
| ForestCover + WoodyStems | 3 | 476.86 | 88.93 | 0.00 |
| ForestCover + WoodyStems + Water | 4 | 478.70 | 90.77 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 503.05 | 115.12 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 509.14 | 121.21 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 510.72 | 122.79 | 0.00 |
| ForestCover + Trees23 | 3 | 526.03 | 138.11 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 530.30 | 142.37 | 0.00 |
| Trees23 | 2 | 530.62 | 142.69 | 0.00 |
| Trees8.23 + Trees23 | 3 | 532.53 | 144.61 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 544.54 | 156.61 | 0.00 |
| WoodyStems + Water | 3 | 548.45 | 160.52 | 0.00 |
| WoodyStems | 2 | 564.03 | 176.10 | 0.00 |
| ForestCover + Trees8.23 | 3 | 576.01 | 188.08 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 578.21 | 190.28 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 580.40 | 192.47 | 0.00 |
| ForestCover | 2 | 583.32 | 195.40 | 0.00 |
| ForestCover + Trees3.8 | 3 | 584.38 | 196.45 | 0.00 |
| ForestCover + Water | 3 | 585.55 | 197.62 | 0.00 |
| ForestCover + Trees3.8 + Water | 4 | 586.56 | 198.64 | 0.00 |
| Trees8.23 | 2 | 623.90 | 235.97 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 625.91 | 237.98 | 0.00 |
| Water | 2 | 631.30 | 243.37 | 0.00 |
| (Intercept) | 1 | 640.97 | 253.04 | 0.00 |
| Trees3.8 | 2 | 643.12 | 255.19 | 0.00 |

Table 2.13. AIC models for forest-breeding species detection rates. See Table 2.6 for column definitions.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 361.15 | 0.00 | 0.42 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 361.78 | 0.63 | 0.30 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 362.46 | 1.32 | 0.22 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 364.81 | 3.67 | 0.07 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 376.27 | 15.12 | 0.00 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 376.42 | 15.27 | 0.00 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 378.43 | 17.29 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 379.47 | 18.32 | 0.00 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 380.30 | 19.16 | 0.00 |
| ForestCover + WoodyStems | 3 | 380.41 | 19.26 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 381.04 | 19.90 | 0.00 |
| ForestCover + WoodyStems + Water | 4 | 381.06 | 19.91 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 381.12 | 19.97 | 0.00 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 386.50 | 25.36 | 0.00 |
| Trees23 + WoodyStems + Water | 4 | 387.97 | 26.82 | 0.00 |
| Trees23 + WoodyStems | 3 | 401.05 | 39.90 | 0.00 |
| WoodyStems + Water | 3 | 404.34 | 43.19 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 406.61 | 45.46 | 0.00 |
| WoodyStems | 2 | 416.77 | 55.62 | 0.00 |
| ForestCover + Trees8.23 | 3 | 457.33 | 96.18 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 459.26 | 98.12 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 459.62 | 98.48 | 0.00 |
| ForestCover + Water | 3 | 461.55 | 100.40 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 461.97 | 100.82 | 0.00 |
| ForestCover + Trees23 | 3 | 462.56 | 101.41 | 0.00 |
| ForestCover | 2 | 463.13 | 101.98 | 0.00 |
| ForestCover + Trees3.8 + Water | 4 | 463.36 | 102.21 | 0.00 |
| ForestCover + Trees3.8 | 3 | 465.39 | 104.25 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 476.04 | 114.90 | 0.00 |
| Trees8.23 + Trees23 | 3 | 491.43 | 130.29 | 0.00 |
| Water | 2 | 495.47 | 134.32 | 0.00 |
| Trees23 | 2 | 496.39 | 135.24 | 0.00 |
| Trees8.23 | 2 | 500.91 | 139.76 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 501.61 | 140.46 | 0.00 |
| (Intercept) | 1 | 523.01 | 161.87 | 0.00 |
| Trees3.8 | 2 | 524.83 | 163.69 | 0.00 |

Table 2.14. AIC models for early-successional breeding species detection rates. See Table 2.6 for column definitions.

| | $k$ | AIC$_C$ | ΔAIC$_C$ | $\omega_i$ |
|---|---|---|---|---|
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 323.82 | 0.00 | 0.22 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 323.85 | 0.04 | 0.22 |
| Trees23 + WoodyStems + Water | 4 | 324.72 | 0.90 | 0.14 |
| WoodyStems + Water | 3 | 325.76 | 1.94 | 0.08 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 325.93 | 2.12 | 0.08 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 326.37 | 2.55 | 0.06 |
| ForestCover + WoodyStems + Water | 4 | 326.62 | 2.81 | 0.06 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 327.05 | 3.23 | 0.04 |
| Trees3.8 + WoodyStems + Water | 4 | 327.29 | 3.47 | 0.04 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 327.39 | 3.57 | 0.04 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 329.69 | 5.88 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 337.64 | 13.82 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 339.91 | 16.10 | 0.00 |
| Trees23 + WoodyStems | 3 | 341.09 | 17.28 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 342.65 | 18.83 | 0.00 |
| WoodyStems | 2 | 342.78 | 18.96 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 344.35 | 20.53 | 0.00 |
| ForestCover + WoodyStems | 3 | 344.48 | 20.66 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 344.87 | 21.06 | 0.00 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 345.33 | 21.51 | 0.00 |
| ForestCover + Water | 3 | 352.47 | 28.65 | 0.00 |
| ForestCover + Trees3.8 + Water | 4 | 353.48 | 29.66 | 0.00 |
| Trees8.23 + Trees23 | 3 | 353.75 | 29.94 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 354.81 | 30.99 | 0.00 |
| Water | 2 | 354.87 | 31.05 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 355.12 | 31.30 | 0.00 |
| Trees3.8 | 2 | 359.53 | 35.72 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 359.85 | 36.03 | 0.00 |
| ForestCover + Trees3.8 | 3 | 361.78 | 37.96 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 362.17 | 38.36 | 0.00 |
| Trees23 | 2 | 363.28 | 39.46 | 0.00 |
| ForestCover + Trees23 | 3 | 363.39 | 39.57 | 0.00 |
| Trees8.23 | 2 | 367.50 | 43.69 | 0.00 |
| (Intercept) | 1 | 368.82 | 45.01 | 0.00 |
| ForestCover + Trees8.23 | 3 | 369.50 | 45.69 | 0.00 |
| ForestCover | 2 | 370.98 | 47.16 | 0.00 |

Table 2.15. AIC models for obligate frugivore detection rates. See Table 2.6 for column definitions.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| Trees23 + WoodyStems + Water | 4 | 228.15 | 0.00 | 0.28 |
| ForestCover + Trees23 + WoodyStems + Water | 5 | 230.22 | 2.07 | 0.10 |
| ForestCover + WoodyStems + Water | 4 | 230.28 | 2.13 | 0.10 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 230.57 | 2.41 | 0.08 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 230.59 | 2.43 | 0.08 |
| WoodyStems + Water | 3 | 230.59 | 2.44 | 0.08 |
| Trees3.8 + WoodyStems + Water | 4 | 231.81 | 3.65 | 0.04 |
| ForestCover + WoodyStems | 3 | 232.24 | 4.09 | 0.04 |
| ForestCover + Trees3.8 + WoodyStems + Water | 5 | 232.39 | 4.23 | 0.03 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 232.69 | 4.53 | 0.03 |
| ForestCover + Trees8.23 + WoodyStems | 4 | 232.73 | 4.58 | 0.03 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 233.07 | 4.92 | 0.02 |
| Trees23 + WoodyStems | 3 | 234.53 | 6.38 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + WoodyStems | 5 | 234.95 | 6.79 | 0.01 |
| Water | 2 | 235.18 | 7.02 | 0.01 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 235.36 | 7.20 | 0.01 |
| ForestCover + Water | 3 | 235.94 | 7.78 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 235.94 | 7.79 | 0.01 |
| Trees8.23 + Trees23 + Water | 4 | 236.35 | 8.19 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 236.57 | 8.41 | 0.00 |
| ForestCover + Trees8.23 | 3 | 237.12 | 8.96 | 0.00 |
| WoodyStems | 2 | 237.65 | 9.49 | 0.00 |
| ForestCover + Trees3.8 | 3 | 237.74 | 9.59 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 | 4 | 237.86 | 9.70 | 0.00 |
| ForestCover + Trees3.8 + Water | 4 | 237.94 | 9.78 | 0.00 |
| ForestCover | 2 | 238.01 | 9.85 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 238.40 | 10.25 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Water | 5 | 238.46 | 10.31 | 0.00 |
| Trees8.23 | 2 | 238.80 | 10.65 | 0.00 |
| ForestCover + Trees23 | 3 | 239.74 | 11.58 | 0.00 |
| ForestCover + Trees3.8 + Trees8.23 + Trees23 | 5 | 239.80 | 11.65 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 240.01 | 11.85 | 0.00 |
| Trees8.23 + Trees23 | 3 | 240.17 | 12.02 | 0.00 |
| Trees23 | 2 | 240.27 | 12.12 | 0.00 |
| (Intercept) | 1 | 241.48 | 13.33 | 0.00 |
| Trees3.8 | 2 | 241.86 | 13.71 | 0.00 |

Table 2.16. AIC models for species diversity estimates of transient species recorded on transects. See Table 2.6 for column definitions.

| **Species** | **Models** | **Forest cover** | **Trees 3-8 cm** | **Trees 8-23 cm** | **Trees > 23 cm** | **Woody stem hits** | **Water** |
|---|---|---|---|---|---|---|---|
| Golden-crowned Kinglet | 2 | 0.019 (0.006) | 0.031 (0.006) | -0.046 (0.022) | 0.296 (0.036) | -0.005 (0.013) | |
| White-throated Sparrow | 1 | -0.025 (0.006) | | | 0.071 (0.023) | 0.039 (0.003) | 0.029 (0.010) |
| Myrtle Warbler | 3 | 0.002 (0.004) | | | | 0.009 (0.004) | -0.006 (0.009) |
| Ruby-crowned Kinglet | 4 | 0.023 (0.012) | | -0.055 (0.030) | 0.145 (0.043) | | |
| Magnolia Warbler | 3 | 0.023 (0.012) | | -0.045 (0.088) | | | |
| **Guild** | | | | | | | |
| Neotropical migrants | 2 | 0.020 (0.005) | | 0.013 (0.015) | | 0.011 (0.005) | |
| Temperate migrants | 2 | | 0.003 (0.003) | -0.024 (0.008) | 0.128 (0.011) | 0.025 (0.002) | 0.018 (0.005) |
| Forest breeders | 3 | 0.006 (0.002) | 0.000 (0.003) | -0.018 (0.008) | 0.110 (0.013) | 0.021 (0.002) | 0.017 (0.006) |
| Early successional breeders | 3 | -0.018 (0.004) | -0.005 (0.003) | -0.037 (0.008) | | 0.024 (0.002) | |
| Obligate frugivores | 4 | | -0.006 (0.004) | 0.015 (0.009) | -0.036 (0.015) | 0.016 (0.003) | 0.033 (0.008) |
| Transient species diversity | 1 | | | | 0.026 (0.012) | 0.008 (0.002) | 0.016 (0.005) |

Table 2.17. Beta coefficients and standard errors for vegetation variables in species and guild models with $\Delta AIC_c < 2$ using transect survey data.

| **Species** | **Early successional** | **Restoration** | **Mature forest** | **F** | ***p*** |
|---|---|---|---|---|---|
| Golden-crowned Kinglet | 4.88 (1.38) a* | 1.5 (0.67) b* | 5 (5.81) | 3.83 | 0.035 |
| White-throated Sparrow | 20.75 (8.11) a* | 5.5 (1.71) | 2.9 (0.67) c* | 5.01 | 0.015 |
| Myrtle Warbler | 6.75 (2.07) | 7.2 (2.57) | 4.9 (1.39) | 0.36 | 0.699 |
| Ruby-crowned Kinglet | 1.88 (0.69) | 0.9 (0.38) | 2.4 (0.58) | 2.06 | 0.149 |
| Magnolia Warbler | 1 (0.27) | 0.9 (0.35) | 1.5 (0.76) | 0.38 | 0.6903 |
| **Guild** | | | | | |
| Neotropical migrants | 6.75 (1.33) | 3.8 (1.18) | 6.7 (3.32) | 0.97 | 0.394 |
| Temperate migrants | 40.38 (12.03) | 18.7 (5.83) | 29.7 (6.2) | 1.79 | 0.187 |
| Forest breeders | 46.38 (11.14) | 22.3 (6.32) | 42.6 (7.04) | 2.64 | 0.091 |
| Early successional breeders | 24.38 (8.14) | 20.3 (5.05) | 4.9 (1.36) c* | 4.01 | 0.031 |
| Obligate frugivores | 22.88 (4.07) | 16.7 (5.11) | 10.9 (1.39) | 2.31 | 0.119 |

Table 2.18. Comparison of means and standard errors of detection rates for analyzed species and guilds by habitat type. Pairwise comparisons: a = Early successional vs. Restored, b = Restored vs. Mature forest, c = Mature forest vs. Early successional. *** = $p < 0.001$, ** = $p < 0.01$, * = $p < 0.05$.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| % Berry + Trees8.23 + WoodyStems | 4 | 129.88 | 0.00 | 0.41 |
| % Berry + WoodyStems | 3 | 131.33 | 1.44 | 0.20 |
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 132.35 | 2.47 | 0.12 |
| % Berry + WoodyStems + Water | 4 | 132.97 | 3.09 | 0.09 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 133.64 | 3.76 | 0.06 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 134.91 | 5.03 | 0.03 |
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 135.38 | 5.50 | 0.03 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 135.69 | 5.80 | 0.02 |
| Trees23 + WoodyStems + Water | 4 | 136.06 | 6.18 | 0.02 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 136.39 | 6.51 | 0.02 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 138.17 | 8.29 | 0.01 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 138.96 | 9.08 | 0.00 |
| WoodyStems + Water | 3 | 143.77 | 13.89 | 0.00 |
| WoodyStems | 2 | 144.96 | 15.07 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 146.11 | 16.23 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 146.43 | 16.55 | 0.00 |
| Trees23 + WoodyStems | 3 | 146.86 | 16.98 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 147.88 | 18.00 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 148.13 | 18.25 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 207.65 | 77.77 | 0.00 |
| % Berry + Trees23 | 3 | 208.07 | 78.19 | 0.00 |
| % Berry + Water | 3 | 208.95 | 79.07 | 0.00 |
| % Berry + Trees8.23 | 3 | 210.05 | 80.17 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 210.84 | 80.96 | 0.00 |
| % Berry + Trees3.8 + Water | 4 | 211.09 | 81.21 | 0.00 |
| % Berry | 2 | 211.50 | 81.62 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 211.92 | 82.04 | 0.00 |
| % Berry + Trees3.8 | 3 | 212.85 | 82.97 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 248.75 | 118.87 | 0.00 |
| Water | 2 | 251.44 | 121.56 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 256.91 | 127.03 | 0.00 |
| Trees8.23 | 2 | 258.99 | 129.11 | 0.00 |
| Trees8.23 + Trees23 | 3 | 259.27 | 129.39 | 0.00 |
| Trees23 | 2 | 262.06 | 132.18 | 0.00 |
| (Intercept) | 1 | 262.57 | 132.69 | 0.00 |

Table 2.19. AIC results for White-throated Sparrow capture rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 192.20 | 0.00 | 0.13 |
| Trees3.8 + WoodyStems + Water | 4 | 192.28 | 0.08 | 0.12 |
| Trees3.8 | 2 | 193.12 | 0.92 | 0.08 |
| % Berry + WoodyStems + Water | 4 | 193.63 | 1.43 | 0.06 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 194.00 | 1.80 | 0.05 |
| % Berry + Trees3.8 + Water | 4 | 194.23 | 2.04 | 0.05 |
| % Berry + Water | 3 | 194.35 | 2.16 | 0.04 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 194.73 | 2.54 | 0.04 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 194.75 | 2.56 | 0.04 |
| Water | 2 | 194.90 | 2.71 | 0.03 |
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 194.91 | 2.72 | 0.03 |
| Trees23 | 2 | 194.99 | 2.79 | 0.03 |
| % Berry + Trees3.8 | 3 | 195.20 | 3.01 | 0.03 |
| Trees3.8 + Trees8.23 | 3 | 195.30 | 3.11 | 0.03 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 195.79 | 3.60 | 0.02 |
| % Berry + Trees23 | 3 | 195.99 | 3.80 | 0.02 |
| WoodyStems + Water | 3 | 196.25 | 4.06 | 0.02 |
| Trees23 + WoodyStems | 3 | 196.28 | 4.09 | 0.02 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 196.53 | 4.33 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 196.67 | 4.48 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 196.70 | 4.51 | 0.01 |
| Trees8.23 + Trees23 | 3 | 196.76 | 4.57 | 0.01 |
| % Berry + WoodyStems | 3 | 196.87 | 4.68 | 0.01 |
| (Intercept) | 1 | 197.10 | 4.91 | 0.01 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 197.32 | 5.13 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 197.46 | 5.27 | 0.01 |
| % Berry | 2 | 197.79 | 5.59 | 0.01 |
| WoodyStems | 2 | 197.90 | 5.71 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 197.99 | 5.79 | 0.01 |
| % Berry + Trees8.23 + WoodyStems | 4 | 198.09 | 5.89 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 198.13 | 5.93 | 0.01 |
| Trees23 + WoodyStems + Water | 4 | 198.23 | 6.04 | 0.01 |
| Trees8.23 | 2 | 198.51 | 6.32 | 0.01 |
| Trees8.23 + Trees23 + Water | 4 | 198.84 | 6.64 | 0.00 |
| % Berry + Trees8.23 | 3 | 199.18 | 6.99 | 0.00 |

Table 2.20. AIC results for Magnolia Warbler capture rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| (Intercept) | 1 | 103.42 | 0.00 | 0.15 |
| WoodyStems | 2 | 104.10 | 0.68 | 0.11 |
| % Berry | 2 | 104.72 | 1.30 | 0.08 |
| Water | 2 | 104.97 | 1.55 | 0.07 |
| Trees3.8 | 2 | 105.09 | 1.68 | 0.06 |
| Trees8.23 | 2 | 105.57 | 2.15 | 0.05 |
| Trees23 | 2 | 105.59 | 2.17 | 0.05 |
| WoodyStems + Water | 3 | 105.94 | 2.52 | 0.04 |
| % Berry + WoodyStems | 3 | 106.09 | 2.67 | 0.04 |
| Trees23 + WoodyStems | 3 | 106.36 | 2.94 | 0.03 |
| % Berry + Water | 3 | 106.64 | 3.22 | 0.03 |
| % Berry + Trees3.8 | 3 | 106.77 | 3.35 | 0.03 |
| % Berry + Trees8.23 | 3 | 106.94 | 3.52 | 0.03 |
| % Berry + Trees23 | 3 | 106.98 | 3.56 | 0.03 |
| Trees23 + WoodyStems + Water | 4 | 107.09 | 3.68 | 0.02 |
| Trees3.8 + Trees8.23 | 3 | 107.35 | 3.93 | 0.02 |
| Trees8.23 + Trees23 | 3 | 107.83 | 4.41 | 0.02 |
| Trees3.8 + WoodyStems + Water | 4 | 107.91 | 4.49 | 0.02 |
| % Berry + WoodyStems + Water | 4 | 108.16 | 4.74 | 0.01 |
| Trees8.23 + Trees23 + Water | 4 | 108.27 | 4.85 | 0.01 |
| % Berry + Trees8.23 + WoodyStems | 4 | 108.45 | 5.03 | 0.01 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 108.50 | 5.08 | 0.01 |
| % Berry + Trees3.8 + Water | 4 | 108.59 | 5.18 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 108.72 | 5.30 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 109.13 | 5.71 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 109.45 | 6.03 | 0.01 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 109.47 | 6.05 | 0.01 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 109.54 | 6.12 | 0.01 |
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 110.33 | 6.91 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 110.39 | 6.97 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 110.79 | 7.37 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 111.05 | 7.63 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 111.55 | 8.13 | 0.00 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 112.00 | 8.58 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 113.38 | 9.96 | 0.00 |

Table 2.21. AIC results for Swainson's Thrush capture rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| % Berry + Trees3.8 + Water | 4 | 130.69 | 0.00 | 0.40 |
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 132.53 | 1.85 | 0.16 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 133.10 | 2.42 | 0.12 |
| Trees3.8 + WoodyStems + Water | 4 | 133.73 | 3.04 | 0.09 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 134.84 | 4.16 | 0.05 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 135.03 | 4.34 | 0.05 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 135.52 | 4.83 | 0.04 |
| % Berry + Water | 3 | 136.09 | 5.40 | 0.03 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 137.22 | 6.53 | 0.02 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 137.28 | 6.60 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 137.42 | 6.73 | 0.01 |
| % Berry + WoodyStems + Water | 4 | 137.78 | 7.09 | 0.01 |
| % Berry + Trees3.8 | 3 | 139.09 | 8.41 | 0.01 |
| % Berry + Trees23 | 3 | 139.53 | 8.85 | 0.00 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 140.14 | 9.46 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 141.43 | 10.75 | 0.00 |
| Trees23 + WoodyStems + Water | 4 | 141.55 | 10.86 | 0.00 |
| % Berry | 2 | 141.59 | 10.90 | 0.00 |
| WoodyStems + Water | 3 | 142.13 | 11.44 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 142.63 | 11.94 | 0.00 |
| % Berry + WoodyStems | 3 | 142.80 | 12.11 | 0.00 |
| Water | 2 | 143.47 | 12.78 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 143.54 | 12.85 | 0.00 |
| % Berry + Trees8.23 | 3 | 143.59 | 12.90 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 144.35 | 13.66 | 0.00 |
| % Berry + Trees8.23 + WoodyStems | 4 | 145.04 | 14.35 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 146.33 | 15.64 | 0.00 |
| Trees3.8 | 2 | 146.46 | 15.77 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 147.69 | 17.00 | 0.00 |
| Trees23 + WoodyStems | 3 | 150.12 | 19.43 | 0.00 |
| WoodyStems | 2 | 150.21 | 19.53 | 0.00 |
| Trees23 | 2 | 152.21 | 21.52 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 152.47 | 21.78 | 0.00 |
| (Intercept) | 1 | 153.18 | 22.50 | 0.00 |
| Trees8.23 + Trees23 | 3 | 154.40 | 23.71 | 0.00 |

Table 2.22. AIC results for Golden-crowned Kinglet capture rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| % Berry + Trees3.8 + Water | 4 | 123.26 | 0.00 | 0.19 |
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 123.46 | 0.20 | 0.17 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 123.86 | 0.60 | 0.14 |
| Trees3.8 + WoodyStems + Water | 4 | 124.72 | 1.46 | 0.09 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 125.77 | 2.51 | 0.05 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 125.78 | 2.52 | 0.05 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 125.82 | 2.57 | 0.05 |
| % Berry + Water | 3 | 125.88 | 2.62 | 0.05 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 127.00 | 3.74 | 0.03 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 127.09 | 3.83 | 0.03 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 127.17 | 3.91 | 0.03 |
| % Berry + WoodyStems + Water | 4 | 127.74 | 4.48 | 0.02 |
| % Berry + Trees3.8 | 3 | 128.06 | 4.80 | 0.02 |
| % Berry + Trees23 | 3 | 128.21 | 4.95 | 0.02 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 128.35 | 5.09 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 128.54 | 5.29 | 0.01 |
| Trees23 + WoodyStems + Water | 4 | 129.44 | 6.18 | 0.01 |
| % Berry | 2 | 130.65 | 7.39 | 0.00 |
| WoodyStems + Water | 3 | 131.23 | 7.97 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 132.07 | 8.81 | 0.00 |
| % Berry + WoodyStems | 3 | 133.56 | 10.30 | 0.00 |
| Water | 2 | 133.98 | 10.72 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 135.07 | 11.81 | 0.00 |
| % Berry + Trees8.23 | 3 | 135.22 | 11.96 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 135.44 | 12.18 | 0.00 |
| % Berry + Trees8.23 + WoodyStems | 4 | 135.82 | 12.56 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 136.05 | 12.79 | 0.00 |
| Trees3.8 | 2 | 136.14 | 12.89 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 136.61 | 13.35 | 0.00 |
| Trees23 + WoodyStems | 3 | 137.32 | 14.06 | 0.00 |
| WoodyStems | 2 | 137.57 | 14.31 | 0.00 |
| Trees23 | 2 | 138.52 | 15.26 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 138.70 | 15.44 | 0.00 |
| (Intercept) | 1 | 138.78 | 15.52 | 0.00 |
| Trees8.23 + Trees23 | 3 | 139.50 | 16.24 | 0.00 |

Table 2.23. AIC results for Ruby-crowned Kinglet capture rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| Trees3.8 + WoodyStems + Water | 4 | 313.03 | 0.00 | 0.22 |
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 314.16 | 1.13 | 0.12 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 314.37 | 1.34 | 0.11 |
| % Berry + WoodyStems + Water | 4 | 314.47 | 1.44 | 0.11 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 314.74 | 1.71 | 0.09 |
| WoodyStems + Water | 3 | 314.86 | 1.83 | 0.09 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 315.48 | 2.45 | 0.06 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 316.17 | 3.14 | 0.05 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 316.51 | 3.48 | 0.04 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 317.01 | 3.98 | 0.03 |
| Trees23 + WoodyStems + Water | 4 | 317.17 | 4.14 | 0.03 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 317.51 | 4.48 | 0.02 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 318.58 | 5.54 | 0.01 |
| Trees23 + WoodyStems | 3 | 319.24 | 6.21 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 320.26 | 7.23 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 320.74 | 7.71 | 0.00 |
| % Berry + Trees8.23 + WoodyStems | 4 | 322.16 | 9.13 | 0.00 |
| WoodyStems | 2 | 323.61 | 10.58 | 0.00 |
| % Berry + WoodyStems | 3 | 324.97 | 11.94 | 0.00 |
| Water | 2 | 325.89 | 12.86 | 0.00 |
| % Berry + Water | 3 | 328.13 | 15.10 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 329.14 | 16.11 | 0.00 |
| % Berry + Trees3.8 + Water | 4 | 329.43 | 16.40 | 0.00 |
| Trees8.23 + Trees23 | 3 | 330.46 | 17.43 | 0.00 |
| Trees23 | 2 | 331.05 | 18.02 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 331.12 | 18.09 | 0.00 |
| % Berry + Trees23 | 3 | 332.73 | 19.69 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 334.48 | 21.45 | 0.00 |
| Trees8.23 | 2 | 336.26 | 23.23 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 337.04 | 24.01 | 0.00 |
| Trees3.8 | 2 | 337.07 | 24.04 | 0.00 |
| % Berry + Trees3.8 | 3 | 337.82 | 24.79 | 0.00 |
| (Intercept) | 1 | 337.88 | 24.85 | 0.00 |
| % Berry + Trees8.23 | 3 | 338.25 | 25.22 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 338.36 | 25.33 | 0.00 |

Table 2.24. AIC results for Neotropical migrant capture rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| % Berry + Trees8.23 + WoodyStems | 4 | 271.36 | 0.00 | 0.71 |
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 273.76 | 2.40 | 0.21 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 276.24 | 4.88 | 0.06 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 278.92 | 7.56 | 0.02 |
| % Berry + WoodyStems | 3 | 285.74 | 14.38 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 286.74 | 15.38 | 0.00 |
| % Berry + WoodyStems + Water | 4 | 288.10 | 16.74 | 0.00 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 288.80 | 17.44 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 288.88 | 17.52 | 0.00 |
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 289.63 | 18.27 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 289.95 | 18.58 | 0.00 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 290.13 | 18.77 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 290.27 | 18.91 | 0.00 |
| Trees23 + WoodyStems + Water | 4 | 298.77 | 27.41 | 0.00 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 300.65 | 29.29 | 0.00 |
| WoodyStems | 2 | 301.88 | 30.52 | 0.00 |
| WoodyStems + Water | 3 | 302.85 | 31.49 | 0.00 |
| Trees23 + WoodyStems | 3 | 303.89 | 32.53 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 305.22 | 33.85 | 0.00 |
| % Berry + Trees8.23 | 3 | 346.10 | 74.74 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 348.10 | 76.74 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 348.43 | 77.07 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 350.84 | 79.48 | 0.00 |
| % Berry + Trees23 | 3 | 355.76 | 84.40 | 0.00 |
| % Berry | 2 | 355.78 | 84.42 | 0.00 |
| % Berry + Water | 3 | 357.35 | 85.99 | 0.00 |
| % Berry + Trees3.8 | 3 | 357.49 | 86.12 | 0.00 |
| % Berry + Trees3.8 + Water | 4 | 359.41 | 88.05 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 395.17 | 123.81 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 400.42 | 129.06 | 0.00 |
| Trees8.23 | 2 | 402.03 | 130.67 | 0.00 |
| Trees8.23 + Trees23 | 3 | 403.70 | 132.34 | 0.00 |
| Water | 2 | 410.26 | 138.90 | 0.00 |
| Trees3.8 | 2 | 415.65 | 144.28 | 0.00 |
| (Intercept) | 1 | 415.65 | 144.29 | 0.00 |
| Trees23 | 2 | 416.44 | 145.08 | 0.00 |

Table 2.25. AIC results for temperate migrant capture rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 410.82 | 0.00 | 0.42 |
| % Berry + Trees8.23 + WoodyStems | 4 | 412.64 | 1.82 | 0.17 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 412.98 | 2.16 | 0.14 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 413.69 | 2.87 | 0.10 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 414.41 | 3.59 | 0.07 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 414.94 | 4.12 | 0.05 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 416.14 | 5.32 | 0.03 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 418.43 | 7.61 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 418.87 | 8.05 | 0.01 |
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 419.52 | 8.70 | 0.01 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 421.35 | 10.53 | 0.00 |
| % Berry + WoodyStems + Water | 4 | 424.37 | 13.55 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 425.32 | 14.50 | 0.00 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 426.07 | 15.25 | 0.00 |
| Trees23 + WoodyStems + Water | 4 | 426.09 | 15.27 | 0.00 |
| WoodyStems + Water | 3 | 427.18 | 16.36 | 0.00 |
| % Berry + WoodyStems | 3 | 428.22 | 17.40 | 0.00 |
| WoodyStems | 2 | 434.32 | 23.50 | 0.00 |
| Trees23 + WoodyStems | 3 | 434.46 | 23.64 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 469.15 | 58.33 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 470.67 | 59.85 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 471.70 | 60.88 | 0.00 |
| % Berry + Trees8.23 | 3 | 472.13 | 61.31 | 0.00 |
| % Berry + Trees3.8 + Water | 4 | 474.98 | 64.16 | 0.00 |
| % Berry + Water | 3 | 476.25 | 65.43 | 0.00 |
| % Berry + Trees23 | 3 | 476.43 | 65.61 | 0.00 |
| % Berry + Trees3.8 | 3 | 477.84 | 67.02 | 0.00 |
| % Berry | 2 | 482.45 | 71.63 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 495.13 | 84.31 | 0.00 |
| Trees8.23 + Trees23 | 3 | 500.98 | 90.16 | 0.00 |
| Water | 2 | 501.45 | 90.63 | 0.00 |
| Trees8.23 | 2 | 503.87 | 93.05 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 505.85 | 95.03 | 0.00 |
| Trees23 | 2 | 512.51 | 101.69 | 0.00 |
| (Intercept) | 1 | 517.30 | 106.48 | 0.00 |

Table 2.26. AIC results for forest-breeding species capture rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 264.23 | 0.00 | 0.57 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 265.98 | 1.75 | 0.24 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 266.84 | 2.61 | 0.16 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 270.91 | 6.68 | 0.02 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 273.14 | 8.91 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 275.13 | 10.90 | 0.00 |
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 302.49 | 38.26 | 0.00 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 302.97 | 38.74 | 0.00 |
| Trees3.8 + WoodyStems + Water | 4 | 310.57 | 46.34 | 0.00 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 315.76 | 51.53 | 0.00 |
| % Berry + WoodyStems + Water | 4 | 317.83 | 53.60 | 0.00 |
| Trees23 + WoodyStems + Water | 4 | 317.93 | 53.70 | 0.00 |
| Trees23 + WoodyStems | 3 | 322.91 | 58.68 | 0.00 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 326.41 | 62.19 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 327.76 | 63.53 | 0.00 |
| WoodyStems + Water | 3 | 333.05 | 68.82 | 0.00 |
| % Berry + Trees8.23 + WoodyStems | 4 | 338.79 | 74.56 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 368.34 | 104.11 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 378.76 | 114.53 | 0.00 |
| Trees8.23 + Trees23 | 3 | 383.07 | 118.84 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 395.99 | 131.76 | 0.00 |
| % Berry + WoodyStems | 3 | 398.99 | 134.76 | 0.00 |
| WoodyStems | 2 | 404.23 | 140.00 | 0.00 |
| % Berry + Trees23 | 3 | 419.51 | 155.28 | 0.00 |
| Trees23 | 2 | 430.99 | 166.76 | 0.00 |
| % Berry + Trees3.8 + Water | 4 | 433.38 | 169.16 | 0.00 |
| Water | 2 | 435.00 | 170.77 | 0.00 |
| % Berry + Water | 3 | 436.98 | 172.75 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 465.97 | 201.74 | 0.00 |
| % Berry + Trees8.23 | 3 | 469.05 | 204.82 | 0.00 |
| Trees8.23 | 2 | 469.67 | 205.45 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 470.04 | 205.81 | 0.00 |
| % Berry + Trees3.8 | 3 | 511.41 | 247.18 | 0.00 |
| Trees3.8 | 2 | 519.74 | 255.51 | 0.00 |
| % Berry | 2 | 529.90 | 265.67 | 0.00 |
| (Intercept) | 1 | 531.37 | 267.14 | 0.00 |

Table 2.27. AIC results for early-successional breeding species capture rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 321.82 | 0.00 | 0.26 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 323.14 | 1.31 | 0.13 |
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 323.35 | 1.52 | 0.12 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 323.90 | 2.07 | 0.09 |
| % Berry + Trees8.23 + WoodyStems | 4 | 323.91 | 2.08 | 0.09 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 324.10 | 2.27 | 0.08 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 324.15 | 2.33 | 0.08 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 324.87 | 3.05 | 0.06 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 325.39 | 3.56 | 0.04 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 327.42 | 5.60 | 0.02 |
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 327.78 | 5.96 | 0.01 |
| Trees3.8 + WoodyStems + Water | 4 | 328.29 | 6.47 | 0.01 |
| WoodyStems + Water | 3 | 331.37 | 9.55 | 0.00 |
| Trees23 + WoodyStems + Water | 4 | 332.50 | 10.67 | 0.00 |
| % Berry + WoodyStems + Water | 4 | 332.51 | 10.69 | 0.00 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 334.54 | 12.71 | 0.00 |
| % Berry + WoodyStems | 3 | 335.04 | 13.22 | 0.00 |
| WoodyStems | 2 | 335.45 | 13.62 | 0.00 |
| Trees23 + WoodyStems | 3 | 335.66 | 13.83 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 380.27 | 58.45 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 382.06 | 60.24 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 382.39 | 60.56 | 0.00 |
| % Berry + Trees8.23 | 3 | 382.73 | 60.90 | 0.00 |
| % Berry + Trees3.8 + Water | 4 | 384.38 | 62.55 | 0.00 |
| % Berry + Trees23 | 3 | 384.61 | 62.78 | 0.00 |
| % Berry + Water | 3 | 385.33 | 63.51 | 0.00 |
| % Berry + Trees3.8 | 3 | 386.14 | 64.32 | 0.00 |
| % Berry | 2 | 389.97 | 68.14 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 396.69 | 74.87 | 0.00 |
| Trees8.23 + Trees23 | 3 | 398.46 | 76.64 | 0.00 |
| Water | 2 | 399.39 | 77.57 | 0.00 |
| Trees8.23 | 2 | 401.30 | 79.48 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 403.55 | 81.73 | 0.00 |
| Trees23 | 2 | 406.23 | 84.40 | 0.00 |
| (Intercept) | 1 | 410.66 | 88.84 | 0.00 |

Table 2.28. AIC results for obligate frugivore capture rates. See Table 2.6 for explanation of column abbreviations used.

| | $k$ | $AIC_C$ | $\Delta AIC_C$ | $\omega_i$ |
|---|---|---|---|---|
| Trees3.8 + Trees8.23 + WoodyStems | 4 | 263.02 | 0.00 | 0.25 |
| Trees3.8 + WoodyStems + Water | 4 | 263.60 | 0.58 | 0.19 |
| % Berry + Trees3.8 + Trees8.23 + WoodyStems | 5 | 264.85 | 1.83 | 0.10 |
| Trees3.8 + Trees8.23 + WoodyStems + Water | 5 | 265.39 | 2.37 | 0.08 |
| % Berry + Trees3.8 + WoodyStems + Water | 5 | 265.62 | 2.60 | 0.07 |
| Trees3.8 + Trees23 + WoodyStems + Water | 5 | 265.88 | 2.86 | 0.06 |
| WoodyStems | 2 | 266.12 | 3.10 | 0.05 |
| Trees23 + WoodyStems | 3 | 266.64 | 3.63 | 0.04 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems | 6 | 267.30 | 4.28 | 0.03 |
| WoodyStems + Water | 3 | 267.73 | 4.72 | 0.02 |
| Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 6 | 267.92 | 4.91 | 0.02 |
| % Berry + WoodyStems | 3 | 268.37 | 5.35 | 0.02 |
| Trees23 + WoodyStems + Water | 4 | 268.75 | 5.73 | 0.01 |
| Trees8.23 + Trees23 + WoodyStems | 4 | 269.00 | 5.99 | 0.01 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 + WoodyStems + Water | 7 | 269.88 | 6.86 | 0.01 |
| % Berry + WoodyStems + Water | 4 | 270.10 | 7.08 | 0.01 |
| % Berry + Trees8.23 + WoodyStems | 4 | 270.73 | 7.72 | 0.01 |
| % Berry + Trees23 + WoodyStems + Water | 5 | 271.01 | 7.99 | 0.00 |
| Trees8.23 + Trees23 + WoodyStems + Water | 5 | 271.22 | 8.20 | 0.00 |
| % Berry + Trees3.8 | 3 | 275.49 | 12.47 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 | 4 | 276.63 | 13.61 | 0.00 |
| % Berry + Trees23 | 3 | 277.53 | 14.52 | 0.00 |
| Trees23 | 2 | 277.72 | 14.70 | 0.00 |
| % Berry + Trees3.8 + Water | 4 | 277.80 | 14.78 | 0.00 |
| Trees3.8 | 2 | 277.84 | 14.82 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Trees23 | 5 | 278.00 | 14.98 | 0.00 |
| % Berry | 2 | 278.37 | 15.35 | 0.00 |
| (Intercept) | 1 | 278.56 | 15.54 | 0.00 |
| % Berry + Trees3.8 + Trees8.23 + Water | 5 | 278.90 | 15.88 | 0.00 |
| Water | 2 | 279.19 | 16.17 | 0.00 |
| Trees3.8 + Trees8.23 | 3 | 279.52 | 16.51 | 0.00 |
| Trees8.23 + Trees23 | 3 | 279.82 | 16.81 | 0.00 |
| % Berry + Water | 3 | 279.84 | 16.82 | 0.00 |
| % Berry + Trees8.23 | 3 | 280.50 | 17.48 | 0.00 |
| Trees8.23 | 2 | 280.66 | 17.65 | 0.00 |
| Trees8.23 + Trees23 + Water | 4 | 282.14 | 19.12 | 0.00 |

Table 2.29. AIC results for transient species diversity estimates (mist netting data). See Table 2.6 for explanation of column abbreviations used.

| **Species** | **Models** | **% Berry** | **Trees 3-8 cm** | **Trees 8-23 cm** | **Trees > 23 cm** | **Woody stem hits** | **Water** |
|---|---|---|---|---|---|---|---|
| White-throated Sparrow | 2 | 0.018 (0.004) | | -0.032 (0.017) | | 0.021 (0.002) | |
| Magnolia Warbler | 5 | -0.009 (0.005) | 0.007 (0.003) | -0.003 (0.013) | | 0.006 (0.003) | -0.004 (0.002) |
| Swainson's Thrush | 5 | 0.005 (0.005) | | | | 0.005 (0.003) | -0.002 (0.002) |
| Golden-crowned Kinglet | 2 | 0.008 (0.004) | -0.013 (0.005) | | | 0.002 (0.003) | -0.009 (0.003) |
| Ruby-crowned Kinglet | 4 | 0.005 (0.005) | | | -0.069 (0.052) | 0.012 (0.003) | -0.010 (0.002) |
| **Guild** | | | | | | | |
| Neotropical migrants | 6 | -0.003 (0.002) | 0.004 (0.002) | | | 0.006 (0.001) | -0.003 (0.001) |
| Temperate migrants | 2 | 0.008 (0.002) | | -0.028 (0.007) | | 0.011 (0.001) | |
| Forest breeders | 2 | 0.005 (0.001) | 0.003 (0.001) | -0.016 (0.005) | | 0.008 (0.001) | |
| Early successional breeders | 2 | | 0.002 (0.002) | -0.051 (0.007) | -0.138 (0.041) | 0.012 (0.001) | -0.113 (0.004) |
| Obligate frugivores | 3 | 0.003 (0.002) | 0.003 (0.002) | -0.015 (0.005) | | 0.009 (0.001) | -0.001 (0.001) |
| Transient species diversity | 3 | -0.001 (0.003) | 0.004 (0.001) | 0.003 (0.004) | | 0.005 (0.001) | -0.0001 (0.001) |

Table 2.30. Beta coefficients and standard errors for vegetation variables in species and guild models with $\Delta AIC_c < 2$ using mist-netting data.

| Species | Early successional | Restoration | Mature forest | F | *p* |
|---|---|---|---|---|---|
| White-throated Sparrow | 6.17 (2.26) a* | 0.33 (0.19) | 0.6 (0.31) c* | 5.73 | 0.008 |
| Magnolia Warbler | 5.25 (1.28)a*** | 0.33 (0.14) | 0.7 (0.21) c** | 12.23 | < 0.001 |
| Swainson's Thrush | 1.67 (0.26) | 1.25 (0.43) | 1.2 (0.36) | 0.53 | 0.596 |
| Golden-crowned Kinglet | 2.83 (0.49) | 2 (0.63) | 2 (0.98) | 0.49 | 0.616 |
| Ruby-crowned Kinglet | 3.5 (0.68) a*** | 1.42 (0.15) | 0.6 (0.27) c*** | 15.45 | < 0.001 |
| **Guild** | | | | | |
| Neotropical migrants | 18.17 (2.83) a*** | 5.33 (1) | 6.2 (0.81) c*** | 15.00 | < 0.001 |
| Temperate migrants | 18.67 (3.63) a*** | 3.25 (0.84) | 7.7 (2.04) c** | 10.54 | < 0.001 |
| Forest breeders | 35.25 (4.31) a*** | 8.08 (1.26) | 14 (2.36) c*** | 23.69 | < 0.001 |
| Early successional breeders | 20.75 (4.32) a* | 8.5 (1.78) | 1.1 (0.5) c*** | 11.71 | < 0.001 |
| Obligate frugivores | 25.42 (2.74) a*** | 6.5 (0.75) | 6 (1.13) c*** | 31.05 | < 0.001 |
| Transient species diversity | 25 (2.04) a*** | 14.08 (2.02) | 15.2 (1.57) c** | 10.05 | < 0.001 |

Table 2.31. Comparison of means and standard errors of capture rates for analyzed species and guilds by habitat type. Pairwise comparisons: a = Early successional vs. Restored, b = Restored vs. Mature forest, c = Mature forest vs. Early successional. *** = $p < 0.001$, ** = $p < 0.01$, * = $p < 0.05$.

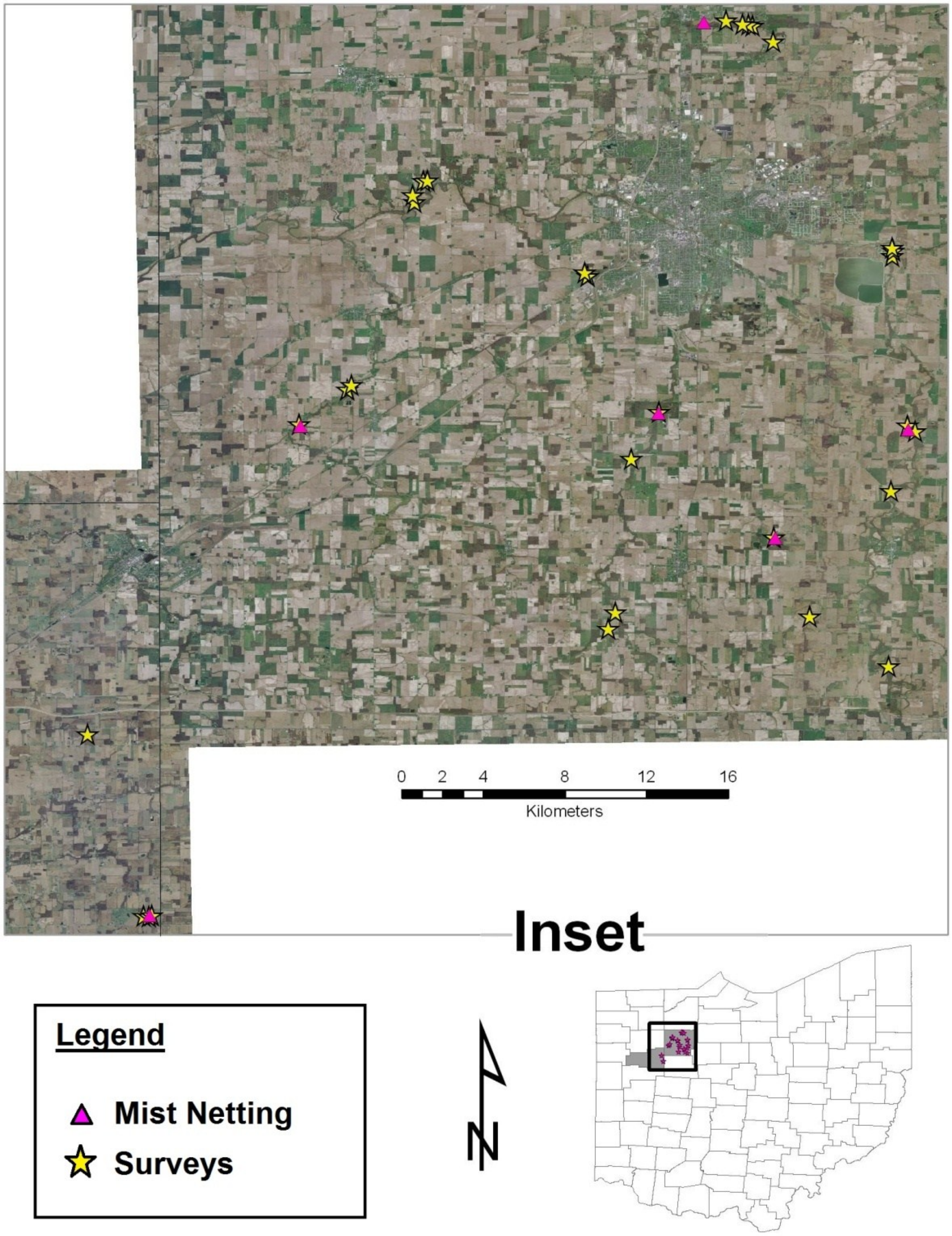


Figure 2.1. Aerial orthophoto of mist netting and transect locations surveyed within Hancock and Allen county, northwest Ohio, late August to late October 2009 and 2010.

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Appendix A: Summary of all species captured in mist nets or detected on transects with scientific names, migratory status and breeding habitat.

| **Species** | **Migratory status** | **Breeding habitat** | **Detections** | **Captures** |
|---|---|---|---|---|
| Great Blue Heron, *Ardea herodias* | TT | OT | 1 | 0 |
| Turkey Vulture, *Cathartes aura* | TT | OT | 1 | 0 |
| Wood Duck, *Aix sponsa* | TT | OT | 53 | 0 |
| Bald Eagle, *Haliaeetus leucocephalus* | TT | OT | 1 | 0 |
| Sharp-shinned Hawk, *Accipiter striatus* | TT | OT | 4 | 3 |
| Cooper's Hawk, *Accipiter cooperii* | TT | OT | 10 | 0 |
| Broad-winged Hawk, *Buteo platypterus* | NT | OT | 1 | 0 |
| Red-tailed Hawk, *Buteo jamaicensis* | TT | OT | 6 | 0 |
| American Woodcock, *Scolopax minor* | TT | OT | 1 | 0 |
| Mourning Dove, *Zenaida macroura* | BT | OP | 29 | 0 |
| Black-billed Cuckoo, *Coccyzus erythropthalmus* | NT | MF | 1 | 2 |
| Yellow-billed Cuckoo, *Coccyzus americanus* | BN | MF | 1 | 2 |
| Eastern Screech Owl, *Megascops asio* | R | OT | 1 | 0 |
| Ruby-throated Hummingbird, *Archilochus colubris* | BN | MF | 18 | 36 |
| Belted Kingfisher, *Megaceryle alcyon* | TT | OT | 4 | 0 |
| Red-headed Woodpecker, *Melanerpes erythrocephalus* | R | MF | 14 | 0 |
| Red-bellied Woodpecker, *Melanerpes carolinus* | R | MF | 171 | 3 |
| Yellow-bellied Sapsucker, *Sphyrapicus varius* | TT | MF | 11 | 0 |
| Downy Woodpecker, *Picoides pubescens* | R | MF | 228 | 20 |
| Hairy Woodpecker, *Picoides villosus* | R | MF | 69 | 0 |
| Northern Flicker, *Colaptes auratus* | R | MF | 97 | 14 |
| Pileated Woodpecker, *Dryocopus pileatus* | R | MF | 3 | 0 |
| Eastern Wood-pewee, *Contopus virens* | BN | MF | 44 | 8 |
| Yellow-bellied Flycatcher, *Empidonax flaviventris* | TN | MF | 0 | 30 |

Appendix A. Migratory status: BT = Breeding temperate, BN = Breeding Neotropical, NT = Neotropical transient, TT = Temperate transient, R = Resident. Breeding habitat: OT = Other, MF = Mature forest, ES = Early successional, OP = Open habitat. * = Obligate frugivore according to Parrish (1997).

Appendix A continued

| Species | Migratory status | Breeding habitat | Detections | Captures |
|---|---|---|---|---|
| Acadian Flycatcher, *Empidonax virescens* | TN | MF | 9 | 3 |
| Traill's Flycatcher, *Empidonax alnorum/traillii* | TN | ES | 3 | 12 |
| Least flycatcher, *Empidonax minimus* | TN | MF | 0 | 10 |
| Eastern phoebe, *Sayornis phoebe** | BT | MF | 9 | 6 |
| Great Crested Flycatcher, *Myiarchus crinitus* | TN | MF | 24 | 0 |
| Yellow throated Vireo, *Vireo flavifrons** | BN | MF | 7 | 0 |
| Blue-headed Vireo, *Vireo solitaris** | TT | MF | 5 | 3 |
| Warbling Vireo, *Vireo gilvus** | BN | MF | 12 | 5 |
| Philadelphia Vireo, *Vireo philadelphicus** | TN | MF | 1 | 0 |
| Red-eyed Vireo, *Vireo olivaceus** | BT | MF | 23 | 13 |
| Blue Jay, *Cyanocitta cristata* | R | MF | 227 | 19 |
| American Crow, *Corvus brachyrhynchos* | R | OP | 14 | 0 |
| Carolina Chickadee, *Poecile carolinensis* | R | MF | 187 | 59 |
| Black-capped Chickadee, *Poecile atricapillus* | R | MF | 88 | 13 |
| Tufted titmouse, *Baeolophus bicolor* | R | MF | 180 | 73 |
| Red-breasted Nuthatch, *Sitta canadensis* | TT | MF | 13 | 4 |
| White-breasted Nuthatch, *Sitta carolinensis* | R | MF | 326 | 5 |
| Brown Creeper, *Certhia Americana* | TT | MF | 93 | 22 |
| Carolina Wren, *Thryothorus ludovicianus* | R | MF | 28 | 12 |
| House Wren, *Troglodytes aedon* | BN | MF | 81 | 19 |
| Winter Wren, *Troglodytes hiemalis* | TT | MF | 42 | 10 |
| Golden-crowned Kinglet, *Regulus satrapa* | TT | MF | 689 | 157 |
| Ruby-crowned Kinglet, *Regulus calendula* | TT | MF | 155 | 105 |
| Blue-gray Gnatcatcher, *Polioptila caerulea* | BN | MF | 24 | 2 |
| Eastern Bluebird, *Sialia sialis* | BT | OP | 21 | 0 |
| Gray-cheeked Thrush, *Catharus minimus** | TN | MF | 10 | 43 |
| Swainson's Thrush, *Catharus ustulatus** | TN | MF | 69 | 124 |
| Veery, *Catharus fuscescens** | TN | MF | 5 | 5 |
| Hermit Thrush, *Catharus guttatus** | TT | MF | 39 | 89 |
| Wood Thrush, *Hylocichla mustelina** | BN | MF | 36 | 6 |
| American Robin, *Turdus migratorius** | BT | OT | 2213 | 142 |
| Gray Catbird, *Dumetella carolinensis** | BT | ES | 346 | 349 |
| Northern Mockingbird, *Mimus polyglottos** | R | OP | 2 | 1 |

Continued

Appendix A continued

| Species | Migratory status | Breeding habitat | Detections | Captures |
|---|---|---|---|---|
| Brown Thrasher, *Toxostoma rufum** | BT | ES | 15 | 15 |
| Eurasian Starling, *Sturnus vulgaris* | R | OP | 97 | 0 |
| Cedar Waxwing, *Bombycilla cedrorum** | BT | OP | 235 | 15 |
| Ovenbird, *Seiurus aurocapillus* | TN | MF | 31 | 66 |
| Northern Waterthrush, *Parkesia noveboracensis* | TN | MF | 6 | 6 |
| Black-and-white Warbler, *Mniotilta varia* | TN | MF | 12 | 18 |
| Prothonotary Warbler, *Protonotaria citrea* | TN | MF | 0 | 1 |
| Tennessee Warbler, *Oreothlypis peregrina* | TN | MF | 6 | 26 |
| Orange-crowned Warbler, *Oreothlypis celata* | TT | ES | 3 | 5 |
| Nashville Warbler, *Oreothlypis ruficapilla* | TN | ES | 18 | 58 |
| Connecticut Warbler, *Oporornis agilis* | TN | MF | 2 | 3 |
| Mourning Warbler, *Geothlypis philadelphia* | TN | ES | 23 | 14 |
| Common Yellowthroat, *Geothlypis trichas* | BN | ES | 100 | 69 |
| American Redstart, *Setophaga ruticilla* | TN | MF | 76 | 51 |
| Cape May Warbler, *Setophaga tigrina* | TN | MF | 1 | 5 |
| Cerulean Warbler, *Setophaga cerulea* | TN | MF | 1 | 0 |
| Northern Parula, *Parula americana* | TN | MF | 6 | 3 |
| Magnolia Warbler, *Setophaga magnolia** | TN | MF | 79 | 157 |
| Bay-breasted Warbler, *Setophaga castanea* | TN | MF | 15 | 2 |
| Blackburnian Warbler, *Setophaga fusca* | TN | MF | 7 | 1 |
| Yellow Warbler, *Setophaga petechia* | BN | ES | 2 | 0 |
| Chestnut-sided warbler, *Setophaga pensylvanica* | TN | ES | 27 | 7 |
| Blackpoll Warbler, *Setophaga striata* | TN | MF | 18 | 14 |
| Black-throated Blue Warbler, *Setophaga caerulescens* | TN | MF | 17 | 14 |
| Western Palm Warbler, *Setophaga palmarum palmarum* | TT | MF | 23 | 14 |
| Myrtle Warbler, *Setophaga coronata coronata** | TT | MF | 677 | 48 |
| Black-throated Green Warbler, *Setophaga virens* | TN | MF | 35 | 6 |
| Canada Warbler, *Cardellina canadensis* | TN | MF | 1 | 1 |
| Wilson's Warbler, *Cardellina pusilla* | TN | ES | 4 | 25 |
| Scarlet Tanager, *Piranga olivacea** | TN | MF | 1 | 1 |
| Northern Cardinal, *Cardinalis cardinalis* | R | ES | 479 | 128 |
| Rose-breasted Grosbeak, *Pheucticus ludovicianus** | TN | MF | 25 | 6 |
| Indigo Bunting, *Passerina cyanea* | BN | ES | 27 | 38 |

Continued

Appendix A continued

| **Species** | **Migratory status** | **Breeding habitat** | **Detections** | **Captures** |
|---|---|---|---|---|
| Eastern Towhee, *Pipilo erythrophthalmus* | BT | ES | 14 | 5 |
| American Tree Sparrow, *Spizella arborea* | TT | ES | 1 | 0 |
| Field Sparrow, *Spizella pusilla* | BT | ES | 111 | 38 |
| Chipping Sparrow, *Spizella passerina* | BT | OP | 13 | 0 |
| Savannah Sparrow, *Passerculus sandwichensis* | TT | OT | 9 | 0 |
| White-throated Sparrow, *Zonotrichia albicollis** | TT | MF | 774 | 184 |
| White-crowned Sparrow, *Zonotrichia leucophrys* | TT | ES | 68 | 12 |
| Fox Sparrow, *Passerella iliaca* | TT | MF | 8 | 7 |
| Song Sparrow, *Melospiza melodia* | BT | ES | 287 | 61 |
| Lincoln's Sparrow, *Melospiza lincolnii* | TT | ES | 36 | 9 |
| Swamp Sparrow, *Melospiza georgiana* | TT | ES | 30 | 10 |
| Slate-colored Junco, *Junco hyemalis hyemalis* | WT | MF | 72 | 12 |
| Eastern Meadowlark, *Sturnella magna* | TT | OT | 2 | 0 |
| Brown-headed Cowbird, *Molothrus ater* | BT | OP | 13 | 0 |
| Red-winged Blackbird, *Agelaius phoeniceus* | BT | OP | 31 | 2 |
| Common Grackle, *Quiscalus quiscula* | BT | OP | 3 | 0 |
| Baltimore Oriole, *Icterus galbula** | BN | MF | 16 | 2 |
| Purple Finch, *Carpodacus purpureus* | TT | MF | 21 | 4 |
| House Finch, *Carpodacus mexicanus* | R | OP | 13 | 5 |
| American Goldfinch, *Spinus tristis* | BT | ES | 392 | 47 |
| House Sparrow, *Passer domesticus* | R | OP | 5 | 5 |

Appendix B: Summary of all species and guilds captured by mist-netting site.

| Species | Site: | 1 (ES) | 6 (MF) | 7 (CRP) | 12 (ES) | 25A (MF) | 36 (CRP) |
|---|---|---|---|---|---|---|---|
| Ruby-throated Hummingbird | | 4 | 8 | 5 | 12 | 2 | 1 |
| Downy Woodpecker | | 4 | 6 | 2 | 3 | 3 | 0 |
| Northern Flicker | | 2 | 1 | 8 | 1 | 0 | 2 |
| Eastern Wood-pewee | | 0 | 4 | 0 | 3 | 0 | 0 |
| Yellow-bellied Flycatcher | | 4 | 0 | 4 | 16 | 0 | 2 |
| Acadian Flycatcher | | 0 | 0 | 3 | 0 | 0 | 0 |
| Traill's Flycatcher | | 2 | 0 | 8 | 0 | 0 | 2 |
| Least Flycatcher | | 4 | 0 | 4 | 2 | 0 | 0 |
| Eastern Phoebe | | 0 | 3 | 2 | 1 | 0 | 0 |
| Warbling Vireo | | 1 | 1 | 2 | 0 | 0 | 1 |
| Red-eyed Vireo | | 2 | 2 | 1 | 4 | 1 | 2 |
| Blue Jay | | 5 | 3 | 6 | 1 | 1 | 5 |
| Carolina Chickadee | | 4 | 14 | 3 | 18 | 3 | 11 |
| Black-capped Chickadee | | 2 | 1 | 0 | 8 | 0 | 0 |
| Tufted Titmouse | | 0 | 17 | 0 | 16 | 5 | 2 |
| White-breasted Nuthatch | | 0 | 1 | 0 | 3 | 0 | 0 |
| Brown Creeper | | 7 | 6 | 1 | 2 | 3 | 1 |
| Carolina Wren | | 3 | 7 | 0 | 0 | 0 | 1 |

Appendix B. ES = Early Successional, CRP = Restored, MF = Mature forest. Species with < 5 captures are not reported:  Sharp-shinned Hawk, Black-billed Cuckoo, Yellow-billed Cuckoo, Red-bellied Woodpecker, Blue-headed Vireo, Red-breasted Nuthatch, Blue-gray Gnatcatcher, Wood Thrush, Northern Mockingbird, Northern Parula, Blackburnian Warbler, Prothonotary Warbler, Connecticut Warbler, Canada Warbler, Scarlet Tanager, Red-winged Blackbird, Baltimore Oriole, Purple Finch. See Appendix A for scientific names and guild classifications.

Appendix B continued

| Species | Site: | 1 (ES) | 6 (MF) | 7 (CRP) | 12 (ES) | 25A (MF) | 36 (CRP) |
|---|---|---|---|---|---|---|---|
| House Wren | | 6 | 3 | 3 | 3 | 0 | 1 |
| Winter Wren | | 2 | 3 | 0 | 2 | 2 | 0 |
| Golden-crowned Kinglet | | 31 | 30 | 1 | 25 | 36 | 10 |
| Ruby-crowned Kinglet | | 37 | 5 | 1 | 28 | 5 | 7 |
| Gray-cheeked Thrush | | 10 | 5 | 0 | 5 | 9 | 3 |
| Swainson's Thrush | | 14 | 19 | 4 | 19 | 23 | 12 |
| Veery | | 3 | 2 | 0 | 0 | 0 | 0 |
| Hermit Thrush | | 24 | 4 | 0 | 24 | 9 | 8 |
| American Robin | | 50 | 32 | 2 | 7 | 33 | 2 |
| Gray Catbird | | 125 | 4 | 54 | 82 | 2 | 6 |
| Brown Thrasher | | 8 | 0 | 2 | 3 | 0 | 0 |
| Cedar Waxwing | | 8 | 3 | 2 | 3 | 0 | 1 |
| Orange-crowned Warbler | | 2 | 1 | 0 | 0 | 0 | 2 |
| Tennessee Warbler | | 3 | 2 | 3 | 12 | 0 | 3 |
| Nashville Warbler | | 11 | 11 | 3 | 20 | 0 | 6 |
| Chestnut-sided Warbler | | 4 | 0 | 0 | 1 | 0 | 32 |
| Magnolia Warbler | | 31 | 7 | 2 | 76 | 5 | 10 |
| Cape May Warbler | | 0 | 0 | 1 | 0 | 0 | 4 |
| Black-throated Blue Warbler | | 3 | 5 | 0 | 2 | 3 | 0 |
| Myrtle Warbler | | 10 | 17 | 1 | 13 | 0 | 1 |
| Black-throated Green Warbler | | 1 | 0 | 0 | 3 | 1 | 1 |
| Western Palm Warbler | | 0 | 0 | 7 | 1 | 0 | 3 |
| Bay-breasted Warbler | | 0 | 1 | 0 | 1 | 0 | 0 |
| Blackpoll Warbler | | 0 | 1 | 1 | 6 | 0 | 5 |
| Black-and-white Warbler | | 3 | 6 | 0 | 5 | 0 | 2 |
| American Redstart | | 13 | 0 | 0 | 23 | 2 | 7 |

Continued

Appendix B continued

| Species | Site: | 1 (ES) | 6 (MF) | 7 (CRP) | 12 (ES) | 25A (MF) | 36 (CRP) |
|---|---|---|---|---|---|---|---|
| Ovenbird | | 13 | 14 | 3 | 16 | 2 | 7 |
| Northern Waterthrush | | 0 | 1 | 0 | 4 | 0 | 0 |
| Mourning Warbler | | 2 | 0 | 1 | 5 | 0 | 3 |
| Common Yellowthroat | | 18 | 1 | 14 | 24 | 0 | 5 |
| Wilson's Warbler | | 1 | 1 | 4 | 3 | 0 | 4 |
| Northern Cardinal | | 42 | 9 | 3 | 22 | 1 | 6 |
| Rose-breasted Grosbeak | | 1 | 0 | 1 | 3 | 0 | 1 |
| Indigo Bunting | | 7 | 3 | 0 | 20 | 0 | 0 |
| Eastern Towhee | | 0 | 0 | 0 | 3 | 0 | 2 |
| Field Sparrow | | 2 | 0 | 17 | 8 | 0 | 8 |
| White-throated Sparrow | | 106 | 12 | 5 | 18 | 4 | 2 |
| White-crowned Sparrow | | 4 | 0 | 6 | 0 | 0 | 0 |
| Fox Sparrow | | 1 | 0 | 0 | 5 | 0 | 0 |
| Song Sparrow | | 19 | 0 | 26 | 7 | 0 | 1 |
| Lincoln's Sparrow | | 7 | 0 | 1 | 0 | 0 | 0 |
| Swamp Sparrow | | 8 | 0 | 1 | 0 | 0 | 0 |
| Slate-colored Junco | | 4 | 2 | 0 | 1 | 1 | 3 |
| House Finch | | 4 | 0 | 0 | 0 | 0 | 0 |
| American Goldfinch | | 5 | 2 | 6 | 26 | 0 | 2 |
| House Sparrow | | 4 | 0 | 0 | 0 | 0 | 0 |
| Neotropical migrants | | 131 | 74 | 41 | 221 | 44 | 103 |
| Temperate migrants | | 238 | 78 | 24 | 117 | 59 | 33 |
| Forest breeders | | 358 | 211 | 74 | 383 | 117 | 115 |
| Early successional breeders | | 276 | 31 | 145 | 224 | 3 | 79 |
| Obligate frugivores | | 394 | 111 | 79 | 259 | 86 | 48 |
| Transient species diversity | | 41 | 26 | 33 | 47 | 20 | 33 |

Appendix C: Comparison of species and guilds detected on transects by habitat type.

| | ES Tot | CRP Tot | MF Tot | ES (SE) | CRP (SE) | MF (SE) | F | *p* |
|---|---|---|---|---|---|---|---|---|
| Wood duck | 0 | 9 | 15 | 0 (0) | 0.9 (0.9) | 1.36 (0.75) | 0.74 | 0.488 |
| Red-bellied woodpecker | 16 | 10 | 29 | 2.29 (0.52) | 1 (0.39) b* | 2.64 (0.43) | 4.06 | 0.029 |
| Downy woodpecker | 16 | 14 | 42 | 2.29 (0.42) | 1.4 (0.4) b** | 3.82 (0.6) | 6.38 | 0.005 |
| Northern flicker | 8 | 5 | 14 | 1.14 (0.4) | 0.5 (0.22) | 1.27 (0.19) | 2.76 | 0.083 |
| Blue jay | 20 | 13 | 31 | 2.86 (0.7) | 1.3 (0.42) | 2.82 (0.69) | 2.13 | 0.139 |
| Carolina chickadee | 17 | 9 | 31 | 2.43 (0.95) | 0.9 (0.35) | 2.82 (0.84) | 2.05 | 0.15 |
| Black-capped chickadee | 5 | 8 | 11 | 0.71 (0.29) | 0.8 (0.33) | 1 (0.4) | 0.16 | 0.852 |
| Tufted Titmouse | 22 | 10 | 25 | 3.14 (0.74) | 1 (0.79) | 2.27 (0.47) | 2.42 | 0.11 |
| White-breasted nuthatch | 20 | 20 | 66 | 2.86 (0.4) | 2 (0.68) b** | 6 (0.95) c* | 7.71 | 0.002 |
| Brown creeper | 5 | 1 | 23 | 0.71 (0.36) | 0.1 (0.1) b** | 2.09 (0.64) | 5.3 | 0.012 |
| House wren | 6 | 8 | 8 | 0.86 (0.34) | 0.8 (0.29) | 0.73 (0.27) | 0.05 | 0.956 |
| Golden-crowned kinglet | 31 | 17 | 156 | 4.88 (1.38) a* | 1.5 (0.67) b* | 5 (5.81) | 3.829 | 0.035 |
| Ruby-crowned kinglet | 14 | 11 | 23 | 1.88 (0.69) | 0.9 (0.38) | 2.4 (0.58) | 2.055 | 0.149 |
| American robin* | 284 | 87 | 279 | 40.57 (19.3) | 8.7 (3.11) | 25.36 (6.71) | 2.48 | 0.103 |

Appendix C. Mean number of birds detected by habitat type with standard errors. Species with < 20 detections were not analyzed (see Appendix B). ES = Early successional, CRP = Restored, MF = Mature forest. Pairwise comparisons: a = Early successional vs. Restored, b = Restored vs. Mature forest, c = Mature forest vs. Early successional; *** = p < 0.001, ** = p < 0.01, * = p < 0.05.

Appendix C continued.

| | ES Tot | CRP Tot | MF Tot | ES (SE) | CRP (SE) | MF (SE) | F | *p* |
|---|---|---|---|---|---|---|---|---|
| Gray catbird* | 64 | 29 | 7 | 9.14 (1.5) a*** | 2.9 (1.06) | 0.64 (0.39) c*** | 18.64 | < 0.001 |
| Cedar waxwing* | 14 | 39 | 8 | 2 (0.72) | 3.9 (1.91) | 0.73 (0.38) | 1.82 | 0.184 |
| White-breasted nuthatch | 20 | 20 | 66 | 2.86 (0.4) | 2 (0.68) b** | 6 (0.95) c* | 7.71 | 0.002 |
| Magnolia warbler* | 8 | 6 | 18 | 1 (0.27) | 0.9 (0.35) | 1.5 (0.76) | 0.376 | 0.6903 |
| Myrtle warbler* | 52 | 74 | 49 | 6.75 (2.07) | 7.2 (2.57) | 4.9 (1.39) | 0.363 | 0.699 |
| American redstart | 6 | 7 | 9 | 0.86 (0.26) | 0.7 (0.33) | 0.82 (0.54) | 0.034 | 0.967 |
| Common yellowthroat | 13 | 14 | 1 | 1.86 (1.55) | 1.4 (0.54) | 0.09 (0.09) | 1.54 | 0.233 |
| Northern cardinal | 75 | 29 | 43 | 10.71 (2.41) a*** | 2.9 (0.64) | 3.91 (0.72) b** | 10.53 | < 0.001 |
| Field sparrow | 7 | 20 | 1 | 1 (0.72) | 2 (0.6) b* | 0.09 (0.09) | 4.35 | 0.024 |
| White-throated sparrow* | 135 | 49 | 66 | 20.75 (8.11) a* | 5.5 (1.71) | 2.9 (0.67) c* | 5.011 | 0.015 |
| White-crowned sparrow | 15 | 10 | 0 | 2.14 (1.67) | 1 (1) | 0 (0) | 1.2 | 0.319 |
| Song Sparrow | 28 | 48 | 3 | 4 (2.09) | 4.8 (1.44) b* | 0.27 (0.19) | 4 | 0.031 |
| American goldfinch | 31 | 54 | 11 | 4.43 (2) | 5.4 (2.07) | 1 (0.33) | 2.45 | 0.107 |

Continued

Appendix C continued.

| | ES Tot | CRP Tot | MF Tot | ES (SE) | CRP (SE) | MF (SE) | F | *p* |
|---|---|---|---|---|---|---|---|---|
| Neotropical migrants | 51 | 35 | 72 | 6.75 ( .33) | 3.8 (1.18) | 6.7 (3.32) | 0.9678 | 0.394 |
| Temperate migrants | 280 | 187 | 337 | 40.38 (12.03) | 18.7 (5.83) | 29.7 (6.2) | 1.795 | 0.187 |
| Forest breeders | 469 | 323 | 698 | 46.38 (11.14) | 22.3 (6.32) | 42.6 (7.04) | 2.64 | 0.091 |
| Early successional breeders | 260 | 228 | 81 | 24.38 (8.14) | 20.3 (5.05) | 4.9 (1.36) c* | 4.006 | 0.031 |
| Obligate frugivores | 176 | 167 | 116 | 22.88 (4.07) | 16.7 (5.11) | 10.9 (1.39) | 2.313 | 0.119 |
| Transient species diversity | 45 | 39 | 39 | 23.75 (2.72) | 16.2 (3.42) | 19.9 (0.87) | 2.068 | 0.147 |

Appendix D: Comparison of species and guilds captured by habitat type.

| | ES Total | CRP Total | MF Total | ES (SE) | CRP (SE) | MF (SE) | F | *p* |
|---|---|---|---|---|---|---|---|---|
| Ruby-throated Hummingbird | 16 | 6 | 10 | 0.9 (0.31) | 0.25 (0.18) | 0.33 (0.14) | 2.601 | 0.09 |
| Yellow-bellied Flycatcher | 20 | 6 | 0 | 1.1 (0.28) a* | 0.33 (0.14) | 0.08 (0.08) c*** | 8.852 | < 0.001 |
| Carolina Chickadee | 22 | 14 | 17 | 1.5 (0.67 | 0.67 (0.22) | 0.5 (0.26) | 1.683 | 0.202 |
| Tufted Titmouse | 16 | 2 | 22 | 1.5 (0.52) | 0.17 (0.11) | 1.58 (0.53) | 3.63 | 0.038 |
| Golden-crowned Kinglet | 56 | 11 | 66 | 2.7 (0.58) a* | 0.67 (0.36) b** | 3.67 (0.73) | 7.393 | 0.002 |
| Ruby-crowned Kinglet | 65 | 8 | 10 | 3.5 (0.85) a*** | 0.33 (0.14) | 0.75 (0.22) c*** | 13.262 | < 0.001 |
| Gray-cheeked Thrush* | 15 | 3 | 14 | 0.9 (0.1) | 0.25 (0.13) a* | 0.92 (0.26) | 4.335 | 0.022 |
| Swainson's Thrush* | 33 | 16 | 42 | 1.5 (0.27) | 0.83 (0.24) | 1.83 (0.44) | 2.411 | 0.106 |
| Hermit Thrush* | 48 | 8 | 13 | 2.7 (0.68) a** | 0.58 (0.19) | 0.92 (0.26) c* | 7.623 | 0.002 |
| American Robin* | 57 | 4 | 65 | 1.8 (0.61) | 0.08 (0.08) b** | 5 (1.6) | 6.163 | 0.006 |
| Gray Catbird* | 207 | 60 | 6 | 11.3 (2.51) a*** | 2.75 (0.87) | 1 (0.6) c*** | 13.965 | < 0.001 |
| Nashville Warbler | 31 | 9 | 11 | 1.7 (0.37) a* | 0.5 (0.19) | 0.58 (0.29) c* | 5.263 | 0.011 |
| Chestnut-sided Warbler | 5 | 32 | 0 | 0.2 (0.13) | 0.08 (0.08) | 0 (0) | 1.347 | 0.275 |
| Magnolia Warbler* | 107 | 12 | 12 | 6 (1.42) a*** | 0.5 (0.19) | 0.67 (0.19) c*** | 16.762 | < 0.001 |

Appendix D. Mean number of birds captured by habitat type with standard errors. Species with < 30 captures were not analyzed (see Appendix B). See Appendix C for habitat abbreviations and key to pairwise comparisons.

Appendix D continued.

| | ES Total | CRP Total | MF Total | ES (SE) | CRP (SE) | MF (SE) | F | *p* |
|---|---|---|---|---|---|---|---|---|
| Myrtle Warbler* | 23 | 2 | 17 | 1.5 (0.37) | 0.08 (0.08) | 1.25 (0.66) | 2.827 | 0.075 |
| American Redstart | 36 | 7 | 2 | 2.1 (0.89) | 0.5 (0.34) | 0.17 (0.11) c* | 4.018 | 0.028 |
| Ovenbird | 29 | 10 | 16 | 1.8 (0.36) a* | 0.58 (0.19) | 0.75 (0.28) c* | 5.347 | 0.01 |
| Common Yellowthroat | 42 | 19 | 1 | 2 (0.42) a** | 0.67 (0.26) | 0 (0) c** | 13.997 | < 0.001 |
| Northern Cardinal | 64 | 9 | 10 | 4.3 (1.11) a** | 0.58 (0.26) | 1.25 (0.51) c** | 8.441 | 0.001 |
| Indigo Bunting | 27 | 0 | 3 | 1 (0.39) a** | 0 (0) | 0 (0) c** | 7.815 | 0.001 |
| Field Sparrow | 10 | 25 | 0 | 0.7 (0.26) | 1.5 (0.48) b** | 0 (0) | 5.648 | 0.008 |
| White-throated Sparrow* | 124 | 7 | 16 | 6.2 (2.76) a* | 0.33 (0.19) | 1.33 (0.61) | 4.402 | 0.021 |
| Song Sparrow | 26 | 27 | 0 | 1.7 (0.73) | 1.33 (0.41) | 0.08 (0.08) c* | 3.528 | 0.042 |
| American Goldfinch | 31 | 8 | 2 | 1.8 (0.59) a* | 0.58 (0.19) | 0.08 (0.08) c*** | 6.909 | 0.003 |
| Neotropical migrants | 352 | 144 | 118 | 19.4 (3.27) a*** | 5.33 (1) | 7.17 (0.94) c*** | 15.885 | < 0.001 |
| Temperate migrants | 355 | 57 | 137 | 19.4 (4.34) a*** | 3.25 (0.84) | 8.92 (1.89) c * | 9.891 | < 0.001 |
| Forest breeders | 741 | 189 | 328 | 37 (5.02) a*** | 8.08 (1.26) | 16.08 (2.4) c*** | 22.761 | < 0.001 |
| Early successional breeders | 500 | 224 | 34 | 23.1 (4.83) a** | 8.5 (1.78) | 12.75 (1.1) c*** | 14.028 | < 0.001 |
| Obligate frugivores | 653 | 127 | 197 | 28.4 (3.4) a*** | 7.33 (0.87) | 12.75 (2.55) c*** | 19.789 | < 0.001 |
| Transient species diversity | 48 | 33 | 40 | 26 (2.31) a*** | 14.08 (2.02) | 4.94 (1.42) c** | 10.475 | < 0.001 |

Appendix E: Mean and standard error of habitat features by banding site. See appendix C for habitat abbreviations.

| **Variable** **Site:** | **1 (± SE), ES** | **12 (± SE), ES** | **6 (± SE), MF** | **25 (± SE), MF** | **7 (± SE), RES** | **36 (± SE), RES** |
|---|---|---|---|---|---|---|
| Canopy height | 7.17 (0.38) | 2.92 (0.33) | 18 (1.14) | 19.2 (1.59) | 2.75 (0.11) | 4.39 (0.39) |
| % Canopy cover | 72.5 (11.53) | 36.67 (2.79) | 83 (3.74) | 97 (3) | 10 (1.58) | 51.43 (7.38) |
| Trees 3-8 cm | 33.5 (3.7) | 65.67 (19.03) | 19 (6.34) | 22.6 (5.16) | 78 (7.02) | 12 (3.19) |
| Trees 8-23 cm | 13.33 (2.62) | 9.5 (1.88) | 7 (2.26) | 18.4 (6.12) | 3.2 (0.8) | 20.86 (3.51) |
| Trees > 23 cm | 1.5 (0.62) | 0.5 (0.5) | 4 (1.05) | 9.6 (1.75) | 0 (0) | 0 (0) |
| % Berry | 58.5 (10.51) | 5.33 (3.38) | 54.4 (11.04) | 6.2 (2.56) | 3.4 (2.71) | 16.43 (1.81) |
| Woody stem hits | 104.83 (18.68) | 43.67 (9.78) | 30.6 (3.83) | 46.6 (5.34) | 29.6 (7.48) | 57.86 (8.44) |
| Forb hits | 11 (4.19) | 54 (17.93) | 3.8 (0.97) | 0.4 (0.24) | 37 (4.83) | 19.14 (5.41) |
| Water | 9 (2.68) | 12 (0) | 17 (0) | 190 (0) | 1 (0) | 2 (0) |
| Total woody species diversity | 26 | 12 | 15 | 19 | 9 | 12 |