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Bird Communities of Isolated Cypress Wetlands Along an Urban Gradient in Hillsborough County, Florida

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by

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ABSTRACT

Migratory bird communities are sensitive to landscape alteration. Urban development significantly impacts bird communities on breeding grounds, as well as en-route during migration. One current theory is that Neotropical migratory birds are not limited by breeding or wintering habitat constraints but by food and habitat availability along major migration routes. The eastern flyway is the route taken by neotropical land-birds through eastern North America that follows coastal areas denoted by intense urban development. Coastal areas funnel birds to major departure points along the northern coast of the Gulf of Mexico and the western coast of Florida.

Birds were monitored for 12 consecutive months along a decadal time gradient of urban development. Cypress domes are present through a broad scale of urban development in Hillsborough County creating ideal natural sampling units for long term monitoring of wetland bird communities in urban areas. Residential non-migratory bird communities were least influenced by development and did not change significantly with urban development. Neotropical and short-distance migratory birds, however, declined significantly for both richness and bird abundance with increased urban land cover. Migratory birds positively correlated with forested area at a spatial scale of 500 meters surrounding sites. Wintering migrants hit a critical point in development between 10 and

20 years of age, after which they disappeared. Neotropical migrants were most sensitive to declines significantly at sites classified as heavily degraded by the UMAM (Uniform Mitigation Assessment Method) a ‘wetland integrity index’.

Introduction

Freshwater wetlands are common landscape features of southwest Florida and have important environmental functions. They filter water and sequester contaminants, reducing watershed pollution and eutrophication (Gopal 1999). Wetland habitats promote floral and faunal diversity in general (Ewel and Odum 1984), and many birds utilize them for nesting, roosting, and feeding. Forested wetlands are important foraging and roosting areas for many residential and migratory bird species. En-route neotropical migratory birds rely on these habitats key stopover points as resting and foraging areas during spring and winter migrations (Bryce and Hughes 2002). Forested wetlands also serve as wintering grounds for short-distance migratory birds who wintering in the southeastern United States (Buler et al. 2007).

Wetlands declined by 53% in the continental United States from 1780 to 1980 (Dahl 1990). Florida suffered the greatest loss of any state (3.8 million hectares) (Dahl 1990) in the past two decades Florida lost 44,500 hectares of forested wetlands, 59% attributed to urbanization (Kautz et al. 2007). Urban development is a significant ongoing threat to wetlands of the United States, especially Florida, because of disproportional human populations (54%) living in coastal areas (Crossett et al. 2004).

The Tampa/St. Petersburg metropolitan area in southwest Florida includes parts of Hernando, Hillsborough, Polk, and Pinellas counties and has experienced a 26% population increase in the past two decades contributing to its current population of approximately 4 million (Claritas 2008). In 1986, 23% of land area was covered by freshwater wetlands (Haag et al. 2005), 90% of which were considered isolated, lacking

direct surface water outlets downstream under normal flow conditions (Leibowitz 2003). Of these, 79% were less than 2 hectares in size. Such small isolated wetlands mostly consist of either open freshwater marshes or cypress domes.

The study area in Hillsborough County was originally covered primarily by pine flatwoods, palmetto prairie, and forested wetlands (Xian et al. 2007, Friesen et al. 1995). As urban development increased, cypress domes were ‘preserved’ disproportionately within the urban landscape, compared to upland forest, and have become increasingly isolated from one another and other natural surroundings. Most of the surrounding upland forest and scrub has been converted to urban development isolating cypress domes and removing natural wildlife corridors.

Urban development effects bird community altering species richness, abundance and diversity within incorporated wetlands (Whited et al. 2000). Sensitivity of birds to urbanization varies greatly, and while density and nesting rates can be higher in urban areas, species richness is often greatly reduced (Gravereski 1976, Bessinger and Osborn 1982, Chace and Walsh 2006). Urbanization favors both synanthropic non-migratory species and exotic species such as house sparrows, rock pigeons, and European starlings (Garaffa et al. 2009), while excluding many species sensitive to human disturbance such as neotropical migratory wood warblers (Parulidae) (Rottenborn 1999, Allen and O’Conner 2000, Whited et al. 2000). Bird guilds most tolerant to urban environments are mainly granivores, omnivores, and areal insectivores that can better utilize the urban ecosystem (Allen and O’Conner 2000, Chace and Walsh 2006).

Urban development and surrounding land use have far greater impacts on migratory bird species opposed to residential communities. Neotropical migratory songbirds are most affected by urban development, while short distance wintering migrants such as American robin (*Turdus migratorius*) and yellow-rumped warbler (*Dendroica coronata*) are to a lesser degree (Whitcomb et al. 1981, Flather and Sauer 1996). En-route neotropical migratory birds are particularly susceptible to changes in land use and decrease drastically with increased urbanization. On the southeastern Atlantic and Gulf coast of North America, this poses a problem because neotropical migratory species utilize coastal areas such as the Tampa/St. Petersburg metropolitan area for rest stops and foraging before open ocean flights. The time spent at stopover sites greatly exceeds flight time and determines the duration of total migration time (Alerstam 1981, Buler et al. 2007). Neotropical migrants must build fat reserves during fall migration before embarking on long open ocean flights through the Gulf of Mexico and Caribbean (Moore and Kerlinger 1987). During spring, these same wetlands are used to replenish fat reserves depleted during the return flight. Habitat quality and food availability during stopovers determine the rate at which migrants replenish lost fat reserves. Birds that can recover faster and build reserves are able to spend more time on the breeding grounds, increasing reproductive potential (Moore and Kerlinger 1987, Kuenzi and Moore 1991).

Neotropical migratory birds that breed in northeastern North America and winter in the Caribbean, Central, and South America have declined in recent decades (Robbins et al. 1989, Askins et al. 1990, Peterjohn et al. 1995). Both local and national studies using data from the North American Breeding Bird Surveys indicate declines in many

neotropical migratory species (Finch and Stangle 1993, Robbins et al. 1989). From 1980 to 1991, 35 species of Class A neotropical land-birds declined, excluding swimming and most wading birds, while only 15 species have increased in numbers (Peterjohn et al. 1995). Several species of wood warblers, which comprise the greatest proportion of neotropical migrants, have declined in recent decades (Moore and Kerlinger 1987). Significant declines in Cerulean Warbler (*Dendroica cerulea*) and Prairie Warbler (*Dendroica pinus*) populations have been observed but definitive reasons for these are not yet established (Robbins et al. 1992). Declines are speculated to involve habitat destruction of breeding territory in North American and wintering territory in the Caribbean and South America. However, habitat utilization along complete migratory routes is an important factor for bird migrations that is not yet well understood (Moore and Kerlinger 1987, Newton 2006).

Migration is a time of extremely high bird mortality, in black throated blue warblers' mortality rates during migration range from 27% to 33% accounting for 85% of total annual mortality (Sillett and Holmes 2002). Major reasons for high mortality rates during migration include unfavorable weather conditions, predation, and food availability. Variation in food availability at stopover sites is often density dependent leading to longer stopover times as migration intensity increases (Newton 2006). In woodland sites along the Gulf of Mexico, passerine migrants rapidly depleted insect abundance by up to 67% during periods of high migratory intensity (Moore and Young 1991).

Urban growth results in conversion of natural habitats into managed urbanized systems. Native forested wetlands are reduced and fragmented into heterogeneous disjunct patches or islands (Alig and Healy 1987, and Garaffa et al. 2009). Neotropical migratory birds take visual cues for habitat quality based on various factors including patch size, vegetative structure, adjoining upland tree cover, distance to roads, and urban intensity (Askins et al. 1990, Mills et al. 1991, Whited et al. 2000, Marzluff and Ewing 2001). Bird species richness has been decreases with increasing urban land cover, while diversity increases with increasing tree development, even in urban settings (Donnelly and Marzluff 2006, Pennington et al. 2008).

The purpose of this study was to determine the effects of urban development on bird communities utilizing isolated cypress domes in Hillsborough County, Florida. Since bird guilds respond differently to disturbance the effects were measured for each migratory bird guild by computing total abundance and species richness. A gradient of urban intensity was used to explore how land cover and local site features including vegetation and wetland area influence site selection by residential, wintering short-distance migrants, and en-route neotropical migratory birds. Finally bird species richness was compared to a wetland integrity index (UMAM) to determine its value as an estimator for bird utilization.

Methods

Study Area

This study was conducted on cypress dome wetlands within the Tampa Bay metropolitan area of northern Hillsborough County, Florida, an area of expanding urban development in a landscape rich in small, forested cypress wetlands (Haag et al. 2005). Cypress domes make up a substantial portion of remaining forested (88%) and wetland (52%) areas in urbanized northern Hillsborough County (FGDL 2006).

Cypress domes are depressional, shallow forested wetlands with longer hydroperiods than freshwater marshes (Ewel and Odum 1984). They are typically hydrologically isolated from other wetland and riverine features of the landscape and are dominated by pond cypress (*Taxodium ascendens*) and bald cypress (*Taxodium distichum*), with swamp tupelo (*Nyssa sylvatica* var. *biflora*) often co-dominant. Other common canopy and sub-canopy tree species include red maple (*Acer rubrum*), dahoon holly (*Ilex cassine*), and swamp bay (*Persea palustris*).

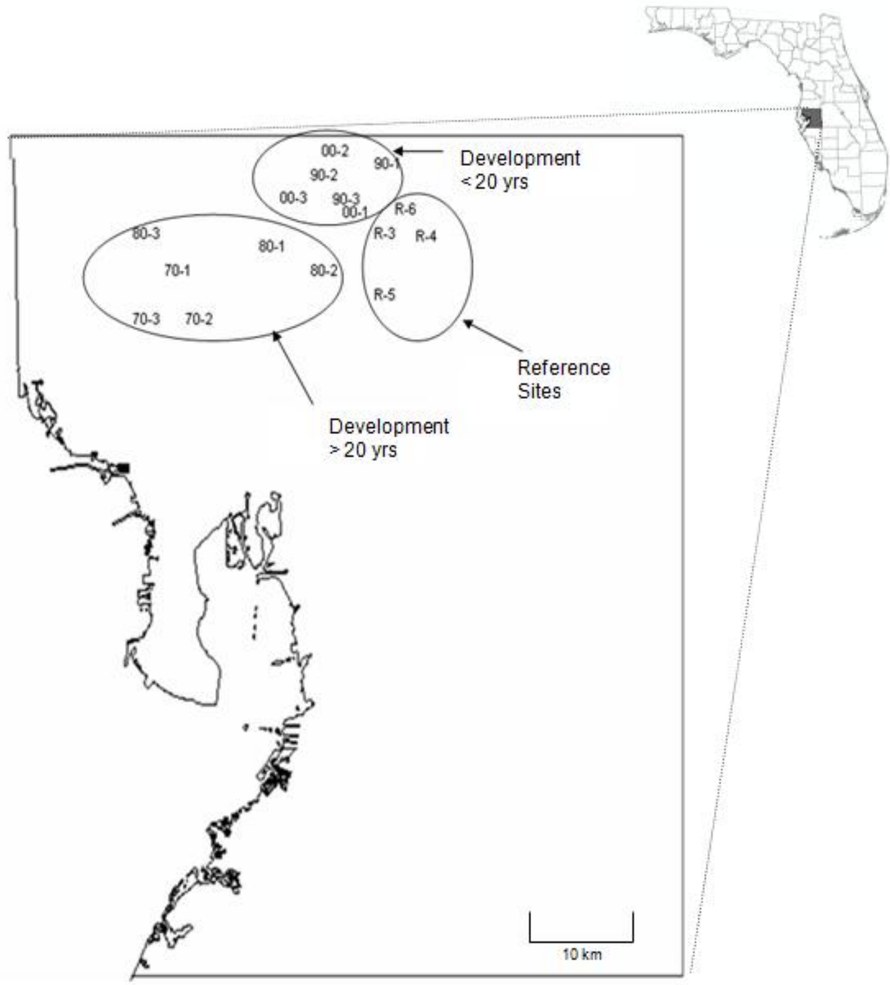


Figure 1. Locations of Study Sites in Hillsborough County FL.

Study areas were analyzed for impacts to bird communities related to urban development. The sixteen sites which ranged between 0.5 and 4.0 hectares were selected along an urban gradient since initial development (1970's – 2000's). Wetland age was based on the decade initial development began within a 200 meter radius of the site. Four reference sites were located in the Lower Hillsborough Flood Detention Area, conservation land owned by Southwest Florida Water Management District, which is devoid of major urban development and/or agricultural activity (FGDL 2006).

Water Quality

Wetland areas were determined using Arc GIS 9.2 and Florida Wetland Inventory Map data. Monitoring stations were established at the deepest point of each wetland where water levels and quality were surveyed monthly during periods of inundation. Water quality data (pH, conductivity, and dissolved oxygen) were collected using a YSI 6920 V2 multi-meter sonde.

Vegetation Analysis

Wetland tree community parameters included canopy cover, species composition, basal area. Canopy composition and mortality were assessed using cross sectional belt transects radiating from the wetland center in the four primary directions. The wetland center was established using Florida Wetland Inventory polygons and a hand held Garmin Etrex GPS was used to find data points in the field. Fallen trees were counted only if the base originated within or intersected the transect line. Belt transects ran from the center to the wetland edge and were 5 meters wide divided into 10 meter long sub-plots arranged linearly along the transect. All trees within transects with diameter greater than 2 cm were identified to species and measured for diameter at breast height. The latter were used to estimate average basal area for each wetland as a measure of forest density. Ten density measurements were made randomly along transect lines for each wetland using a hand held densiometer to calculate average canopy density.

Wetland Habitat Assessment

Habitat assessments have been widely used to estimate wetland integrity and value to wildlife (Lonard et al. 1981). Bird species richness correlates with wetland

habitat assessment values and has been used as an indicator group in environmental indices such as the Habitat Assessment Technique, HAT (Cable et al 1989).

The wetlands of this study were evaluated using the Florida Department of Environmental Protection's (FDEP) Uniform Mitigation Assessment Method, part II (UMAM). This method is used for by state agencies to determine the "ecological value" of wetlands likely to be impacted by development. It evaluates wetlands based on three indicators of wetland function: location and landscape support, "wetland water environment", and community structure (FDEP 2005) to produce a score between 1 and 0, with 1 considered a pristine wetland and scores near 0 for heavily degraded sites with low functional value. For this study, UMAM scores were used to determine relative status of wetlands and degree of impact to the wetland and surrounding area.

Indicators of location and landscape support include: support of surrounding habitat, invasive and exotic plant species presence in proximity to the wetland, wildlife access (presence or absence of barrier), adverse impact of landscape on wildlife, and hydrological connectivity. Wetland water environmental indicators include: water quantity, timing of inundation, frequency, duration, depth, and saturation of soils. Finally, community structure indicators include: wetland plant cover, proportional presence of invasive and exotic species, health of plant community (stress or increased mortality) and recruitment.

Land use

Digital land use and cover data from the Florida Geographic Data Library (FGDL) and the U. S. Geological Survey were used to analyze surrounding land use.

Hillsborough County Property Appraiser maps and FGDL parcel maps were used to determine the average age of the earliest 25% of urban development within 500 meters of each study wetland. Land use within the 500 meter buffer was determined using the 2006 Florida Land Use, Land Cover Classification System (FLUCCS) map layer, and assigned to three categories of land use for analysis: urban, forested and open water/open land.

Percent tree coverage was determined using high-altitude area photographs projected onto digital orthophoto quarter quadrangle (DOQQ) maps. A 1 km^2 fishnet grid containing 1600 individual pixels, each 25m x 25m, was overlain on top of DOQQ map layers and centered on each study area. Sites were analyzed for percent tree cover and impervious surface area (Donnelly and Marzluff 2006, Botsford 2000). Tree cover categories included: forested $\geq$ 70% tree cover and $<$20% impervious surfaces, urban forest $\geq$ 20% tree cover and 20 to 60% impervious structure, and open land (urban)$\geq$ tree cover and $\geq$ 60% impervious surfaces (Botsford 2000).

Birds

Avian surveys were conducted monthly from September 2008 to August 2009. Fixed radius point counts with a maximum radius of 40 meters (Whitcomb et al. 1981), established using a Nikon 550 handheld laser range finder. The area surveyed was approximately 0.50 hectares, which approximates the size of the smallest wetlands in the study. A single bird sampling station was selected at a midpoint between wetland edge and center for each wetland. Birds were surveyed using auditory and visual census techniques for species and individual counts via standard point count techniques (Bibby et al. 2000). Only birds utilizing wetlands were counted in surveys, and both flyovers

and those detected beyond the wetland boundary were excluded. Each survey was conducted for at least 10 minutes within the first 3 hours of daylight, coinciding with the period of greatest avian activity (Bibby et al. 2000). Point counts began following a 5 minute rest period after arrival at each site so birds could re-acclimate after disturbance. All bird surveys were conducted by a single observer for consistency.

Data Analysis

Bird data were summarized for each site by individual abundance and species richness, and then grouped by feeding strategy and migratory guild. Migratory bird guilds were classified as residential non-migratory, neotropical migrants, and short-distance wintering migrants according to Whitcomb et al. (1981) and the American Ornithologists' Union (1998). Species richness was calculated using total species present for each wetland and compared with predicted species richness extrapolated using jackknifing methods described by Zahl (1977) to determine community representation. Birds were also assigned a feeding guild (insectivorous, omnivorous, granivorous, carnivorous, or piscivorous) for further analysis. Species richness and abundance present were selected as dependent variables for data analysis. Diversity was determined using both Shannon's and Simpson's diversity indices for relative comparison among sites. Bird communities were compared cumulatively and monthly between urban and rural sites. Independent variables corresponded to age of urban development, intensity of urban development (percent urban landcover), wetland size, tree cover in the surrounding landscape, and UMAM wetland integrity score. Forested area spatial distribution within 500 meters of sample sites was determined using the T-Square distance method (Diggle 1976). Spearman's rank correlations were used to evaluate correlations between

landscape variables and wetland index parameters. Multiple regression models were constructed to evaluate relationships between bird migratory groups and land cover variables using PASW 18 (SPSS) statistical software significance was determined using standard 95% confidence intervals.

Results

Water Quality

Due to drought conditions during the study, water quality data were only collected from August 2008 through October 2008 when most study wetlands were inundated. Both pH and conductivity were significantly greater in urban than reference wetlands. Wetland pH quickly increased with onset of development (reference mean pH 4.8, urban mean pH 6.3), then remained relatively constant between 6.1 and 6.9 among urban wetlands with no significant differences regardless of development age (Figure 2a).

Conductivity also significantly increased immediately with initiation of urban development from a mean of 70 $\mu S/cm^2 \pm 7.6$ in reference wetlands to a mean of 222 $\mu S/cm^2 \pm 118$ for all urban classes combined (Figures 2b). Wetland 2000-2 had greatly elevated conductivity (566 $\mu S/cm^2 \pm 387$) coinciding with ongoing construction on surrounding property. It was the only site near active construction and had significantly higher conductivity than all other urban sites ($\geq 269\ \mu S/cm^2$) in spite of silt fences designed to reduce runoff and erosion around its perimeter.

Urbanization leads to multiple non-point and point source impacts on wetland water quality. It increases overland flow associated with impervious surfaces leading to deposition of contaminants in wetlands and elevated conductivity (Lee et al. 2009). Use of groundwater for lawn maintenance also can contribute to increased pH in wetlands and lakes (Martin et al. 1976).

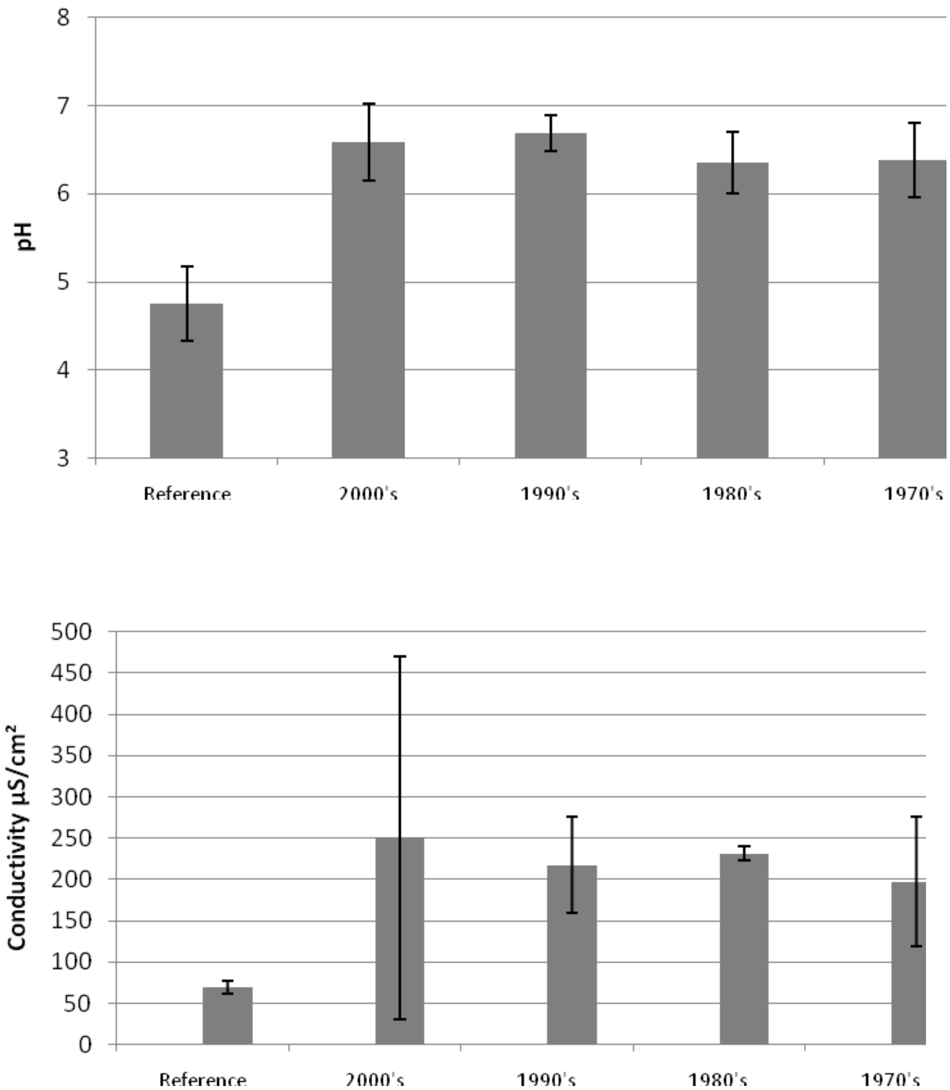


Figure 2. Wetland pH and Conductivity. *standard deviations

Wetland Vegetation

Pond cypress was the dominant tree in all wetlands, contributing 81% ± 9 of total basal area; however, the proportionality of recruit size-class cypress (dbh < 8 cm) declined with urban development. Both facultative and facultative-wet trees species increased proportionately for the recruit size-class of the oldest two urban development periods (1970's and 1980's). Pond cypress contributed 90% of the recruit size-class for reference wetlands and 83% for both the 1990's and 2000's. It continued to decline in

1980's wetlands (52%) of recruit size-class trees, while subdominant trees increased including swamp tupelo (35%) and red maple (13%) These two species, along with dahoon holly and swamp bay, were also present in reference, 2000's, and 1990's wetlands, only in lower concentrations.

The 1970's wetlands suffered the greatest decline in cypress regeneration to 7% of the recruit size-class. Most recruit trees in 1970's wetlands were red maple (62%), Chinese tallow *Sapidium sebiferum* (20%), and Brazilian pepper *Schinus terebenthifolius* (11%). The latter two are invasive exotic species and were found in only in the oldest two wetland classes (1970's - 1980's).   Exotic invasions have been attributed to urban habitat isolation, degraded water quality, and altered hydrology in other studies of Tampa area wetlands (Haag et al. 2005, Rochow 1994).

The decline in dominance of pond cypress associated with geographic isolation within a mosaic of urban development, combined with increased exotic species, are indicative of losses to cypress dome functionality (Jubinsky and Anderson 1996). The shift from cypress dome to mixed wetland hardwoods in developed areas older than 20 years suggest long-term alteration to wetland hydrology typically associated with a decrease in annual hydroperiod. This is a problem that commonly takes decades to appear and is not easily corrected (Ewel and Odum 1984). Urban isolation of forested wetlands often leads to local extirpation of native flora and facilitates invasion by early successional and exotic species (Ehrnfeld 2000).

Table 1. Wetland characteristics and land use by age of surrounding urban development.

Site	Class	Time of Initial Develop-ment	Wetland Area (ha²)	UMAM Site Quality Index	Tree Basal Area (cm²)	Wetland Canopy Cover (Percent)	Surrounding Wetland Area (ha²)*	Urban Land area* (ha)	Forest Tree Cover≥70%* (ha²)	Low Density Forest 70% > Tree Cover > 20% * (ha²)	Open Land Tree Cover ≤20%* (ha²)
R-3	Ref.	-	0.95	0.96	47	91	17.25	0	86.19	11.5	2.31
R-4		-	2.26	0.99	58	95	33.38	0	90.19	8.5	1.31
R-5		-	5.56	0.96	78	90	33.95	0	87	8.5	4.5
R-6		-	3.61	0.94	69	88	39.05	0	87.31	11.63	1.06
2000-1	2000's	2000	1.79	0.74	55	91	19.05	62.77	30.25	16.31	53.44
2000-2		2005	4.31	0.82	45	73	21.93	66.62	40	14.19	47.06
2000-3		2000	2.53	0.81	68	80	29.75	38.87	42.5	20.88	36.63
1990-1	1990's	1990	1.15	0.78	51	84	15.81	70.60	19.44	19.13	61.44
1990-2		1997	2.51	0.76	36	40	29.49	54.88	26.44	15.75	57.81
1990-3		1997	0.56	0.77	41	90	15.84	77.28	20.38	12.44	67.19
1980-1	1980's	1987	1.68	0.76	67	91	30.27	54.22	36.5	21.94	41.56
1980-2		1989	0.5	0.77	72	93	30.28	78.75	44.75	17.06	41.31
1980-3		1984	0.75	0.59	52	93	15.19	57.48	17.38	41.38	43.75
1970-1	1970's	1974	2.75	0.46	76	82	7.77	83.88	23.94	11.44	36.13
1970-2		1978	1.9	0.35	55	92	9.78	86.33	18.19	32.88	48.94
1970-3		1978	2.58	0.44	72	84	6.56	89.98	16.63	41.56	41.81

* Within 500m radius of study wetland

Land Use

The density of urban development within 500 meters of study wetlands significantly correlated with the time of development using Spearman's rank correlation ($\rho = 0.62$, $n = 12$, $p = 0.03$). Total urban area classified as medium to high density urban by the Florida Land Use Cover Classification System (FLUCCS) (FGDL 2004) was used to quantify urban land cover. Urban areas of Tampa Bay are characterized by low levels of forested tree cover (tree cover ≥ 70%) and moderate levels to high of low density forest (70% > tree cover > 20%) and open land (tree cover < 20%). The 1970's class had the highest level of urban land cover within 500 meters, while recent classes (1990's and 2000's) showed moderate levels. The 1980's, however, had extremely high variance with wetlands 80-1 and 80-2 having less open land and more forested area than expected due to their close proximity to the Hillsborough River and its extensive riparian forest.

Landscapes surrounding urban sites differed from those of reference sites by the presence of expansive open land among the former. Landscape surrounding 1970's wetlands had 84% to 90% open land area (within 500 m), recent and intermediate development classes (2000-1980) showed moderately less open land between (39-79%), while reference sites had < 1% open land within the surrounding landscape.

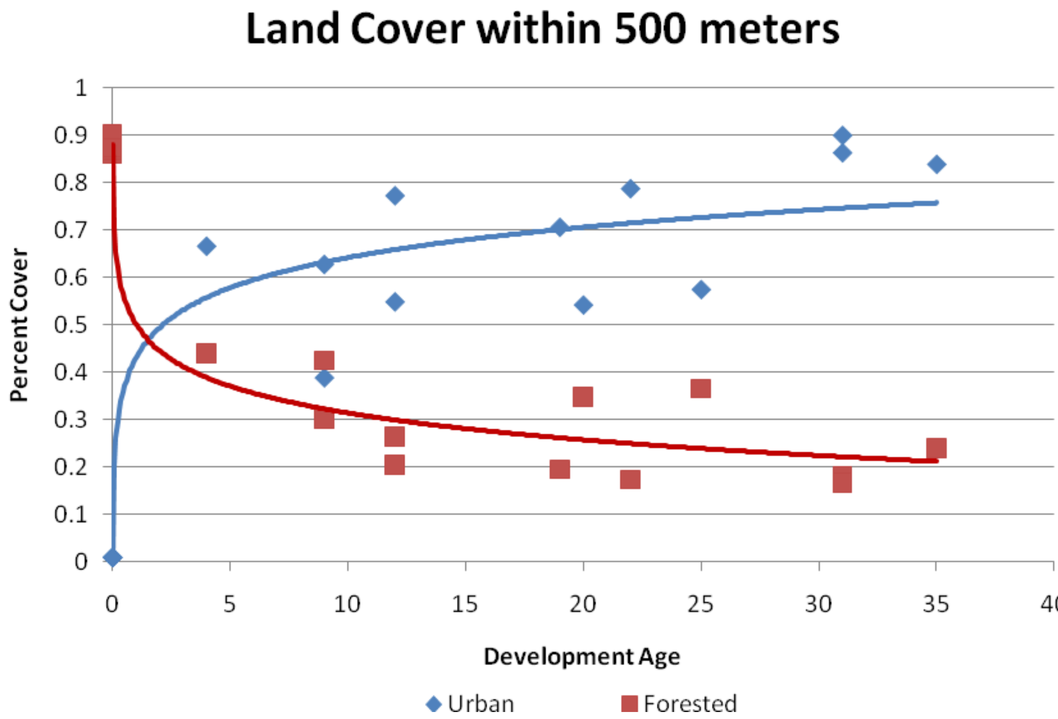


Figure 3. Percent Land Cover Within 500 Meters of Wetlands Along a Gradient of Increasing Age of Urban Development.

Total forested area (tree cover ≥ 70%) within the landscape surrounding urban sites decreased over time ($\rho = 0.915$, $n = 16$, $p > 0.001$) with an increase in urban land area. The 1970's urban sites had 52% less forested area than 2000's sites and 77% less than reference sites (Table 1). Urban land cover increased logarithmically converting 56% ± 15 of the landscape to urban within the first 10 years after development (Figure 3). Urban area significantly correlated with forest using Spearman's rank correlation ($\rho = -0.721$, $n = 16$, $p = 0.002$). The remaining tree cover was either clumped into isolated patches, or transformed into low density urban forest (tree cover ≤ 70%), which is not suitable habitat for many forest dwelling migratory bird species (Marzluff and Ewing 2001). Spatial landscape analysis, using the T-square index of spatial pattern (C), within

500 meters of wetlands showed that forested was the dominant habitat in rural reference areas and was randomly distributed (C = 0.433, $p = 0.20$), but forest area was clumped in urban environments (C = 0.666, $p < 0.05$) into remnant isolated patches.

The expansion of urban development in Hillsborough County in recent decades (Xian et al. 2007) has led to significant losses in forested area (Figure 3) while urban impervious surfaces have increased within the watershed. Garcia-Fresca (2005) found that, in urban areas, during rain events impervious surface area increased overland flow, erosion, and sediment deposition into local water bodies, and decreased ground water recharge.

Wetland Condition Index

Significant negative correlations were found between UMAM wetland index values and ages of development ($R^2 = -0.85$, s = 0.14, $p < 0.001$) (Figure 4). Reference wetlands had the highest scores (UMAM = 0.96 ± 0.02), indicating little to no impact to wetland condition. The 2000's and 1990's wetlands showed light to moderate disturbance (UMAM = 0.78 ± 0.03); the most extensive impacts were decreased water quality and urban isolation. The 1980's wetlands had moderate disturbance (UMAM = 0.70 ± 0.10) with degraded sites structure, water quality, and moderate habitat isolation. The 1970's had the lowest scores (UMAM = 0.41 ± 0.06), showing significant losses to system integrity, associated with habitat isolation, poor water quality, and abundant invasive and exotic plants.

In 1984, rules were revised by state agencies to include general wetland permitting. Wetland resource permitting was further streamlined by the Florida

Department of Environmental Regulation in 1992 to jointly permit storm-water systems and protect wetlands concurrently in response to the Warren Henderson Wetlands Protection Act (SWFWMD 2008). Cluster analysis of wetlands UMAM scores grouped wetlands into two urban groups, urban prior to 1984 (UMAM = 0.42 ±0.06, n = 3) and after 1984 (UMAM = 0.75 ± 0.06, n = 6) and one rural group (UMAM = 0.96 ± 0.02). Though these scores did coincided with state regulatory dates for general wetland permitting they also strongly fit a linear regression (R^2 = -0.85), and it is beyond the scope of this project to suggest any impacts of these policies.

Urban development can negatively impact cypress dome and their bird communities by altering water chemistry, vegetative structure, hydroperiod and surrounding land use (Lonard et al. 1981), which decreases wetland functional values to their landscapes. Migratory birds, both en-route neotropical migrants and wintering short-distance migrants, are extremely sensitive to alterations in forested wetland and riparian structure and decline rapidly with this type of urban development (Rodewald and Matthews 2005).

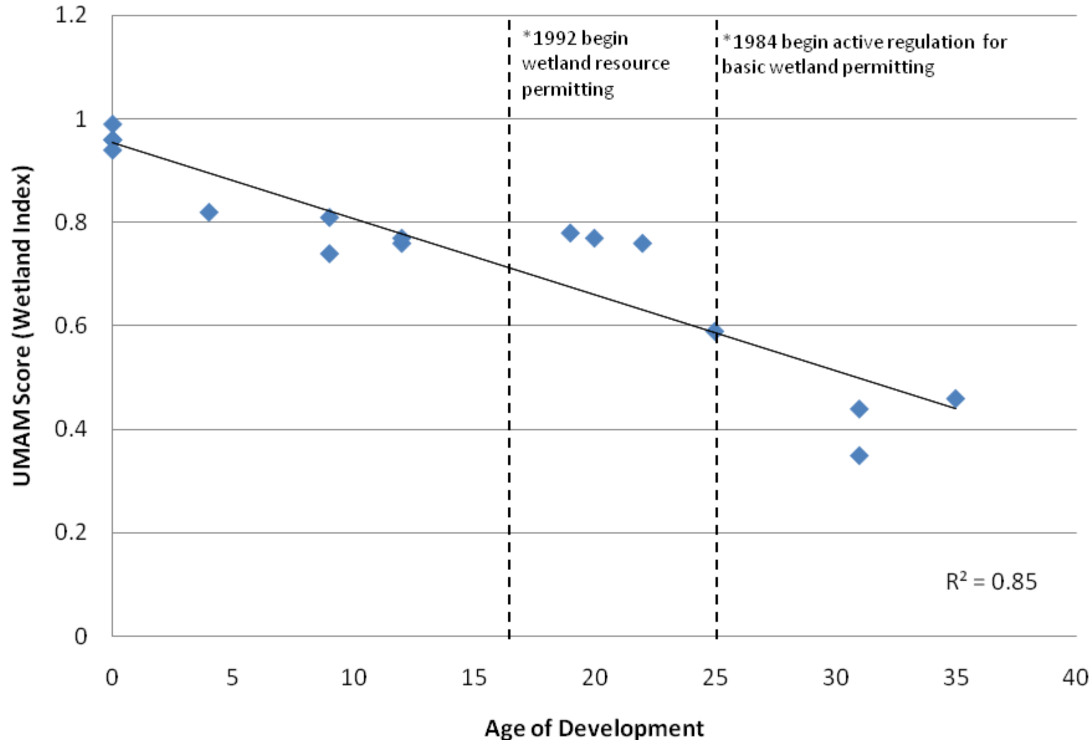


Figure 4. Wetland UMAM Values and Age of Urban Development.

Bird Community

A total of 2528 individuals representing 79 bird species were recorded over the 12 month sampling period at the 16 study sites (Appendix A). Non-migratory, permanent residents made up the majority of bird detections (68% abundance, 36 species), wintering migrants were second (21%, 13 species), and en-route neotropical migrants comprised 11% of observations from 30 species. Six common resident species were found at all sites and made up 53% of total observations. In decreasing order of occurrence, they were: Carolina wren (Thryothorus ludovicianus), blue-gray gnatcatcher (*Polioptila caerulea*), northern cardinal (*Cardinalis cardinalis*), tufted titmouse (*Baeolophus bicolor*), blue jay (*Cyanocitta cristata*), and red-bellied woodpecker (*Melanerpes carolinus*). Eleven permanent residents were present at 75% of sites. Only 3 species of

migratory birds were present at 50% of urban sites: gray catbird (*Dumetella carolinensis*), black-and-white warbler (*Mniotilta varia*), and northern parula (*Parula americana*).

Community Statistics

Bird communities declined with conversion of forest to open land and urban low density urban forest. Species richness negatively correlated with open land ($R^2 = -0.61$, $p = 0.01$) and urban forest area ($R^2 = -0.65$, $p = 0.01$) and positively correlated with forest area ($R^2 = 0.74$, $p = 0.01$). Mean species richness declined over time in wetlands after initial development, from reference wetlands ($S = 33 \pm 2.2$), to recent development (2000's and 1990's) ($S = 25 \pm 4.6$), to older development (1970's and 1980's) ($S = 19 \pm 3.7$). Observed species richness for each wetland was extrapolated using the jackknifing technique (Zahl 1977) to estimate predicted species. The latter did not differ significantly from the observed (chi square $X^2 = 3.57$, $p = 0.31$, $df = 3$) indicating adequate representation among development groups. Monthly species richness (per sampling event) also significantly declined with onset of urban development, and was also highest at rural sites ($S = 10 \pm 3.1$) intermediate at recent development (2000's-1990's, $S = 6 \pm 2.2$) and lowest at older development sites (1980's-1970's, $S = 5 \pm 2.0$).

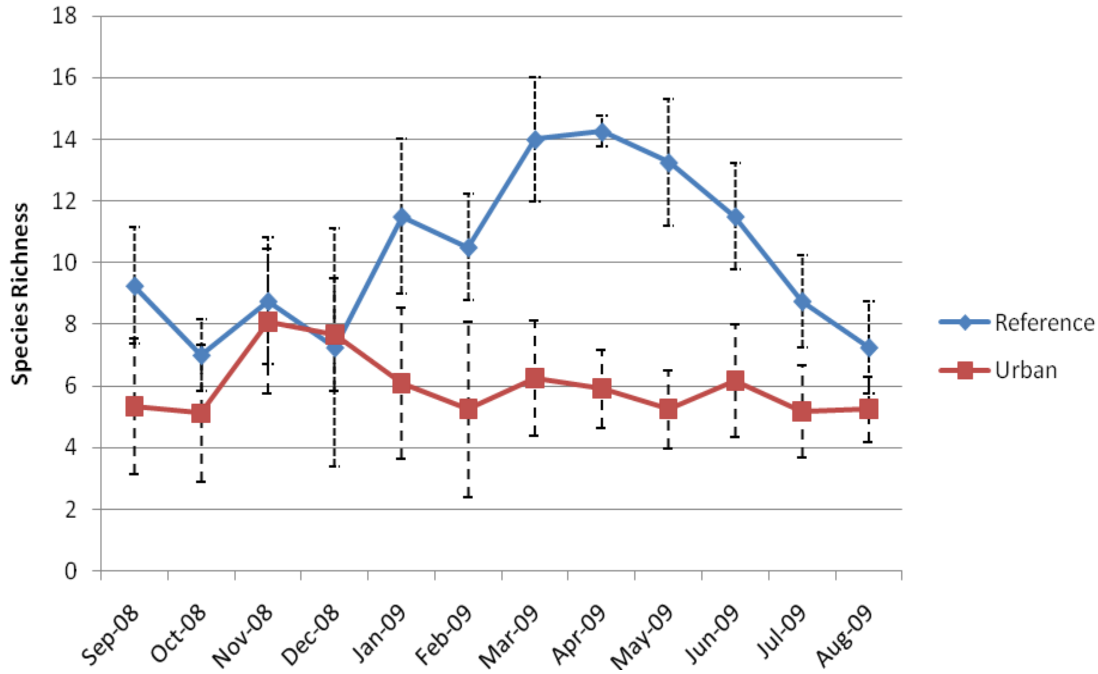


Figure 5. Monthly Species Richness of Urban and Rural sites.

Monthly species richness showed distinctly different trends for combined urban versus rural sites (Figure 5). Reference wetlands had significantly greater species richness during spring and summer coinciding with migration and breeding seasons of many birds, especially migrants that showed stronger preferences for rural sites. Spring and summer are times of intensive calling by territorial males, which increases detection potential for reclusive species and upper canopy birds (Emlen 1971, Farnsworth et al 2002). Common nesting migrants included black-and-white warbler, northern parula, palm warbler, prothonotary warbler (*Protonotaria citrea)*, yellow-throated vireo (*vireo flavifrons*), and red-eyed vireo (*Vireo olivaceus*) that, with the exception of the northern parula, showed strong selective preferences for reference sites.

Neotropical and wintering migrant species richness were significantly lower in urban wetlands during peak migration (Figure 6). Neotropical richness increased in rural

wetlands from April to June coinciding with spring migration. Neotropical migrant species richness over the entire year for urban wetlands showed a max species richness of 0.83 ± 1.1 species compared to rural wetlands with 3.7 ± 1.7 species. Urban migrant richness remained low during the entire study opposed to reference migrants.

Total abundance followed a similar trend as richness. Birds declined as landscapes surrounding wetlands transitioned from rural to urban, similar to results by Donnelly and Marzluff (2006), and Bryce et al. (2002). Abundance declined steadily with urban age through 2000's (n = 144 ± 6) and (1990's, n = 139 ±7) before leveling off in 1980's (n = 97 ± 4) and slightly increasing in 1970's (n = 107 ± 4). Both richness and abundance fell into two distinct urban groups, new development (1990's and 2000's) and old development (1970's and 1980's) that also differed in community composition.

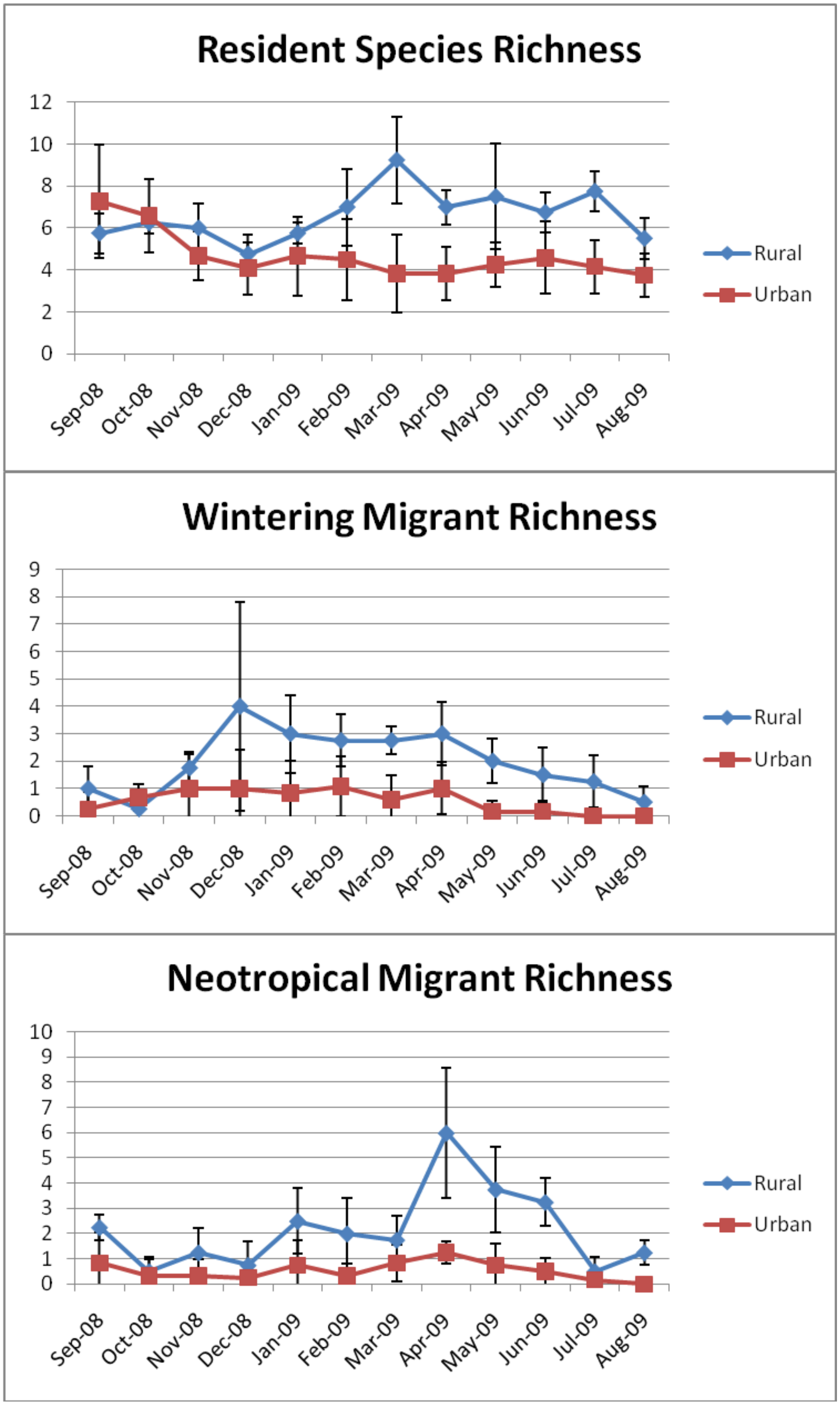


Figure 6. Monthly Species Richness by Migratory Guild.

Old development had proportionately greater resident bird populations, while new had better winter migrant representation. New development was more representative of reference sites while old development resembled urban bird communities described in both the Mid-Atlantic and Pacific Northwest (Cam et al. 2000, Donnelly and Marzluff 2004, and Dowd 1992). The significant differences in bird abundance between urban and rural areas observed here have been reputed in fewer than half of greater than 100 studies published on the topic (Marzluff et al. 2001). Most reputed increased abundance or only slight declines; however, they indicate substantial declines in species richness with urbanization.

Table 2. Mean Species Richness and Individual Abundance per Development Group.

			Migrants	
Site Class	*Totals*	*Residents*	*Wintering*	*Neotropical*
		Species Richness		
1970's	19.3	15.7	1	2.7
1980's	19.3	13.3	1.7	4.3
1990's	22.7	14	5	3.7
2000's	26.3	14.7	5.3	6.3
Reference	32.5	12.7	7.7	12
		Individuals Abundance		
1970's	106.7	97.3	2	7.3
1980's	97	79	7.3	10.7
1990's	138.7	91.7	41.7	5.3
2000's	144	99.3	34.7	10
Reference	268.25	153.75	69	45.5

Shannon's diversity index (H') displayed a similar pattern as species richness (Figure 7). Rural wetlands had the highest diversity (H' = 2.86 ± 0.09) followed by new development (H' = 2.6 ± 0.15), and it was the lowest in old development (H' = 2.42 ± 0.15). Shannon's index indicated bird communities declined with times since development ($R^2 = -0.536$, $p = 0.001$) and increased with forest area within the landscape ($R^2 = 0.57$, $p = 0.001$). Diversity was significantly lower in cypress dome wetlands than in similar wetland deciduous forests of northeastern United States (H'= 4.07 ± 0.16) (Tramer 1969), possibly due to fewer migratory species breeding in southwestern Florida. Simpsons index was used to assess dominance among species observed and did not significantly vary among wetland classes ($\lambda = 0.91 \pm 0.02$).

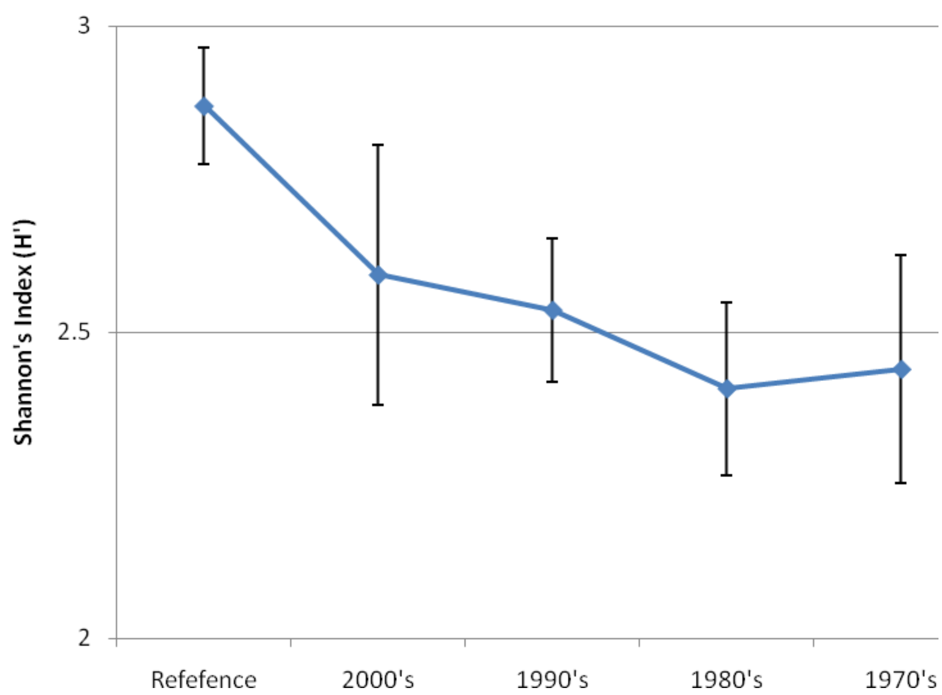


Figure 7. Bird Diversity of Wetland Classes.

Feeding Strategy

Feeding strategies within bird communities changed markedly for residential and migratory bird populations along the rural to urban gradient. Residential communities were composed primarily of omnivores (56% richness, 54% abundance), followed by insectivores (14% richness, 42% abundance), piscivorous wading birds (19% richness, 2% abundance), and carnivores (11% richness, 2% abundance).

Both en-route neotropical and short-distance migrants wintering in Florida were primarily insectivorous passerines (92% and 83%, respectively) and accounted for 84% of total insectivores. They spend the majority of the year on wintering grounds and en-route (Robbins et al. 1989). Insectivorous passerine migrants included tyrant flycatchers, vireos, warblers and their allies that migrate to breeding grounds in northern latitudes during spring (Sherry and Holmes 1995) coinciding with insect emergences and longer days for improved foraging potentials (Alerstam 2001 and Marra et al. 2005). Insectivorous migrants travel south in autumn and winter to avoid freezing temperatures and insufficient insect availability (Alerstam 2001, Keunzi et al. 1991).

Table 3. Migratory Bird Guild Relationships to Land Cover.

Migratory Guild	Area	Forest	Urban Forest	Open Land	adj. R^2
			Species Richness		
en-route migrants	-0.18	0.90**	-0.68**	-0.80**	0.80**
Wintering migrants	-0.24	0.70**	-0.86**	-0.45*	0.75**
Residents	-0.17	-0.39	0.37	0.30	0.07
Total Species	**-0.30**	**0.74****	**-0.65****	**-0.61****	**0.60****
			Abundance		
en-route migrants	-0.04	0.91**	-0.53*	-0.88**	0.78**
Wintering migrants	-0.16	0.72**	-0.79**	-0.51*	0.58**
Residents	-0.17	0.80**	-0.45	-0.77**	0.54*
Total Abundance	**-0.16**	**0.90****	**-0.65***	**-0.77****	**0.76****

*. Significant at p = 0.05 level. **Significant at p = 0.01 level

Resident Birds

The abundance of resident birds positively correlated with forest area ($R^2 = 0.80$, $p = 0.01$) and negatively with urban development ($R^2 = -0.77$, $p = 0.05$) within 500 m of study wetlands. Species richness, however, was not correlated with any land use parameter ($R^2 = 0.07$). There were no significant differences in residential bird populations among urban classes, suggesting they are more adept at living in urban environments (Garaffa et al. 2009, Pennington et al. 2008).

Residential bird species richness did not change significantly with onset of urban development (Table 3), but their representation within the communities changed considerably with changes in urban land use. Both species richness and abundance were proportionally greatest in the two oldest development classes (Individuals 88%, Species 76%) (Figure 8), attributed to drastic declines in migrants and increased abundance of synanthitropic species associated with urban areas, as noted by (Lancaster and Rees 1979, Stratford and Robinson 2005). Wetlands in recent development (1990's and 2000's) were intermediate (Individuals 68%, Species 58%) and reference wetlands had the lowest

proportional representation of resident birds (Individuals 58%, Species 41%) though number of individuals did not decline.

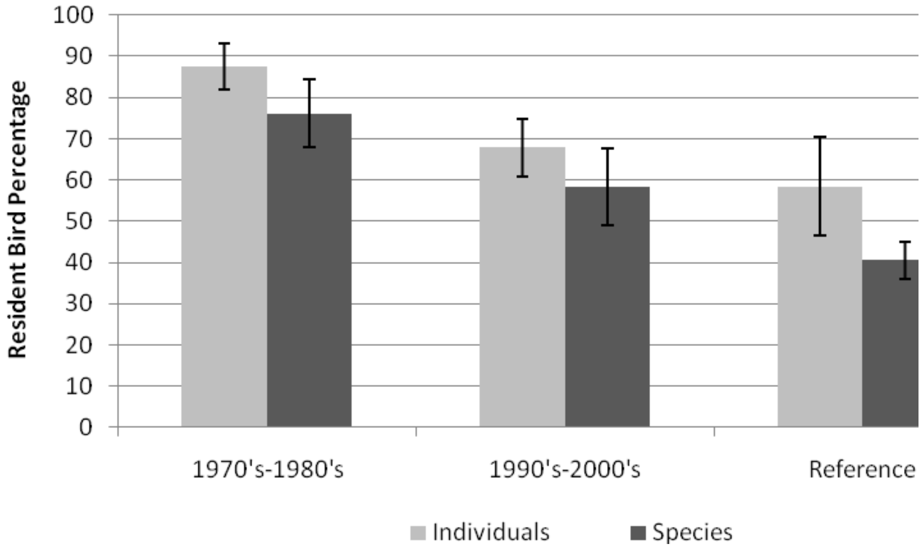


Figure 8. Proportional species richness and abundance of resident bird.

Urban areas were characterized by expansive open land that differed from upland forests surrounding reference wetlands. Such open areas are preferred habitats for many residential omnivores including European starling (*Sturnus vulgaris*) (Whitehead et al. 1995), northern mocking bird (*Mimus polyglottos*), red-winged blackbird (*Agelaius phoeniceus*), common grackle (*Quiscalus quiscula*) and boat–tailed grackle (*Quiscalus major*) (Melles et al. 2003). Dense edge habitats along forested wetlands in urban areas are preferred habitat of brown thrashers (*Toxostoma rufum*), a common edge species not found in interior forests (Aldefer 2006).

Wading birds were represented by 6 resident species including one federally endangered species, the wood stork (*Mycteria americana*), and 3 listed as ‘species of special concern’ by the State of Florida: limpkin (*Aramus guarauna*), little blue heron

(*Egretta caerulea*), and white ibis (*Eudocimus albus*). Wading birds were found only in wetlands with water levels > 10 cm depth, which occurred during only 29% of sampling events, and they showed no response to any other parameter.

Wintering Migrants

Wintering migrants (short-distance migrants) include species wintering exclusively in the southeastern United States and neotropical migrants with substantial populations wintering southern Florida, as well as the Caribbean, Central and South America. They were predominantly insectivores (S = 11, 84% abundance) but did include two omnivores, gray catbird (*Dumetella carolinensis*), an edge species that does very well in urban wetland habitats, and American robin (*Turdus migratorius*), whose preferred foraging habitats include residential and open forested lands (Aldefer 2006). With these exceptions, the remaining 11 species (86% abundance) of wintering migrants were sensitive to urban alteration, including 5 species of warblers that accounted for 80% of wintering migrant observations: prairie warbler (*Dendroica discolor*), palm warbler (*Dendroica palmarum*), pine warbler (*Dendroica pinus*), yellow throated warbler (*Dendroica dominica*), and yellow-rumped warbler (*Dendroica coronata*). Wintering migrants showed sensitivity to landscape alteration by urban development, declining in species richness with increasing low density urban forest ($R^2 = -0.86$, $p = 0.01$), consistent with previous studies (Pennington et al. 2008, Stratford and Robinson 2005). They were positively correlated with forested area ($R^2 = 0.70$, $p = 0.05$) preferring reference sites (n = 276, sites = 4) and recent development (1990's and 2000's) (n = 230, sites = 6) over old development (n = 29, sites = 6). The gray catbird accounted for the majority of old urban observations (79%). It preferred urban sites to rural, possibly

because of thick edge habitats surrounding urban wetlands lacking at rural wetlands. Wintering migrants showed a critical change in site utilization between the 1990's and 1980's classes were, except for gray catbird, they disappeared at older sites (Figure 9). Wintering migrant species richness did not respond to land use, but did positively correlate with UMAM habitat index scores using Spearman's rank correlation ($\rho = 0.810$, $n = 16$, $p = 0.01$).

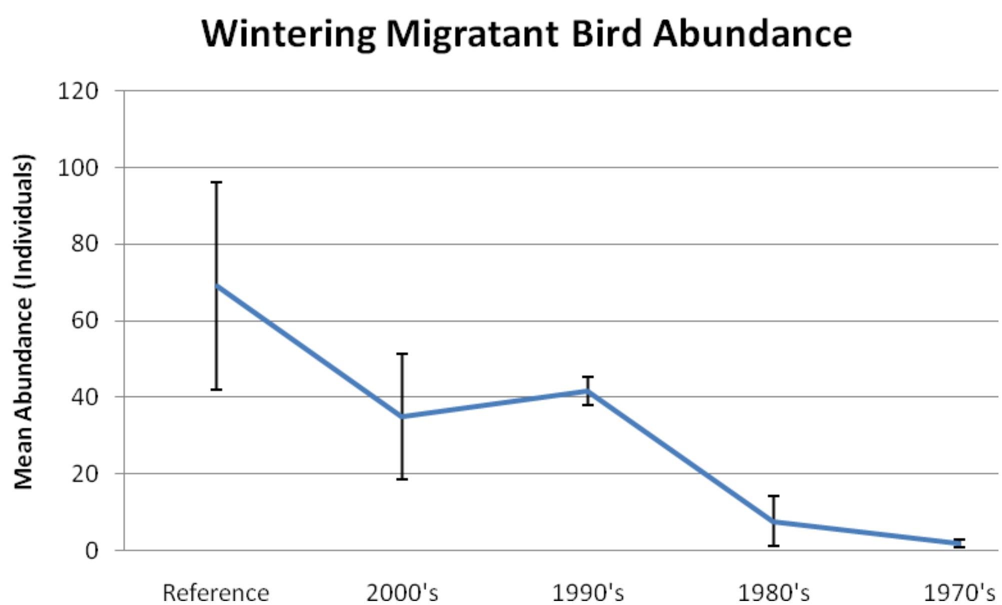
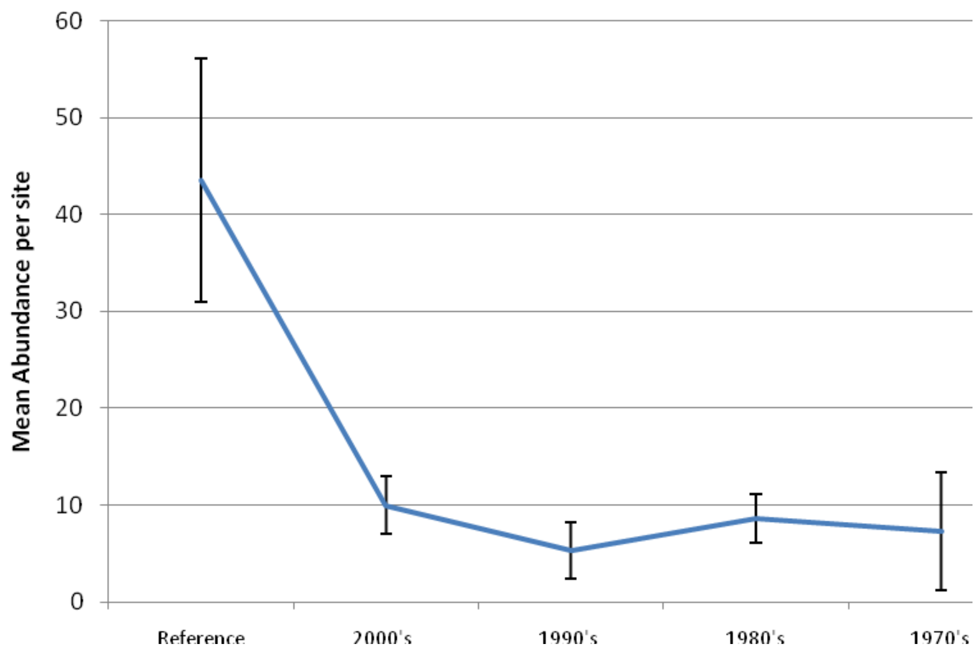


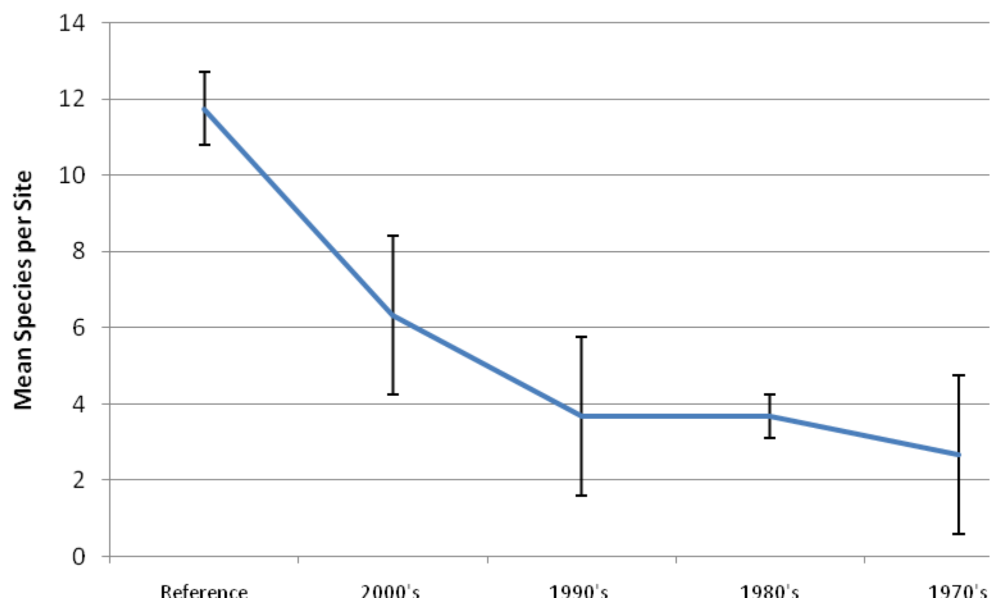
Figure 9. Wintering Bird Abundance.

Neotropical Migrants

Neotropical migrants strongly correlated with land use within 500 meters of wetlands. Positive relationships were found for both species richness ($R^2 = 0.91$, $p = 0.001$) and abundance ($R^2 = 0.90$, $p = 0.005$) versus forest area, and negatively correlated with increased urban area ($R^2 = -0.88$ $p = 0.005$ and $R^2 = -0.80$ $p = 0.005$ respectively) were noted. Neotropical migrant abundance declined from a mean of 43±12 birds per

site in rural wetlands to 7±3 in the 1970's wetlands (Figure 10). Species richness followed a similar pattern, declining from a mean of 12±1 species in rural wetlands to 3±2 in the 1970's sites.





Figures 10. Neotropical Migrant Abundance and Richness by Wetland Class.

Migratory birds did not significantly respond to wetland size (abundance $R^2 = -0.04$ $p = 0.89$, species richness $R^2 = -0.18$ $p = 0.49$), which is consistent with a study of woodlot patch size in urban and rural areas of 4 to 20 hectares (Freisen et al. 1995). Neotropical species richness ($\rho = 0.76$, $n = 16$, $p = 0.01$) and abundance ($\rho = 0.671$, $n = 16$, $p = 0.01$) did positively correlate with wetland integrity index (UMAM) scores for individual wetlands .

Wood warblers (Parulidae) accounted for 70% of neotropical bird observations and 15 of 30 neotropical migrant species observed. Only one warbler, northern parula, appeared insensitive to urban development and was observed at all urban sites of older classes. It was the most abundant, comprising 30% of neotropical migrant observations and 50% from 1970's and 1980's class wetlands. The remaining warbler species declined from 20 observations per site at reference sites to just 2 at old urban sites.

Wood warblers are a group of concern because they are highly migratory and sensitive to habitat alteration not only on their breeding and wintering grounds but along migratory routes (Minor and Urban 2010). Some warblers have declined significantly in recent decades including cerulean and pine warblers (Peterjohn et al. 1995), along with the endangered Kirtland's warbler (*Dendroica kirtlandii*) and the presumed extinct Bachman's warbler (*Vermivora bachmanii*).

Summary and Conclusions

Many neotropical migratory birds use Tampa Bay as a staging area before embarking on open water fall migrations to the Caribbean, Central and South America. It is also an important recovery area upon returning from wintering grounds during early

spring en route to breeding grounds in the northern temperate zone (Moore et al. 1995). These stops just before and right after open ocean flights are vital to bird survival and are also important for recovery of lost fat reserves before continuing to their migratory endpoints (Kuenzi et al. 1991). During migration, birds expend great amounts of energy traveling through unfamiliar terrain with variable food availability. Stopover habitats are vital during migration, a time of extremely high mortality for birds accounting up to 80% of annual mortality (Sillett and Holmes 2002). Habitats along the migration route can act as nutrient bottlenecks, limiting available resources for late migrants (Alerstam and Hedenstrom 1998). Low habitat quality at stopover sites along migratory route greatly extend stopover time and decrease possible time spent on wintering and breeding ranges (Moore and Kerlinger 1987). Delayed arrival on migratory grounds can have additional negative consequences for migratory birds including lower body weight and poor nest site selection. Reduced nesting conditions due to late arrival further result in reduced broods per year, increased nest predation and nest parasitism, and reduced overall nesting success (Rodewald and Shustack 2008).

In recent years, it has been proposed that en-route migration may actually be the factor limiting migratory bird populations (Alerstam and Hedenstrom 1998, Newton 2004). Many migratory birds migrate along the eastern coast of North American (Moore and Kerlinger 1987) in coastal areas that have had seen the greatest national population growth in the United States in recent decades (Dahl 1990). For this reason, preservation of forested wetlands for conservation of migratory bird populations is very important for maintaining current populations and is an issue of concern for migrant species with significant declines in recent decades like the pine warbler that utilizes forested areas

near coasts (Robbins et al. 1989). For cypress domes in Tampa, forest area and habitat quality are important for maintaining migratory bird population. Land managers need to preserve sufficient amounts of wetland and upland forested area in new development and maintain connection to rural or exurban systems in order to provide adequate wildlife corridors to support native bird populations and other biota.

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Appendices

Appendix A. Bird distribution among sites

				Sample Site Species Presence															
Common Name	Scientific Name	Migratory Status	Feeding Strategy	1970-1	1970-2	1970-3	1980-1	1980-2	1980-3	1990-1	1990-2	1990-3	2000-1	2000-2	2000-3	Ref-3	Ref-4	Ref-5	Ref-6
Acadian Flycatcher	Empidonax virescens	M	I	X									X						
American Crow	Corvus brachyrhynchos	R	O							X	X	X		X					
American Goldfinch	Carduelis tristis	M	G					X	X						X			X	
American Robin	Turdus migratorius	W	O					X					X			X			
American Redstart	Setophaga ruticilla	M	I										X						
Anhinga	Anhinga anhinga	R	P									X							
Barred Owl	Strix varia	R	C	X								X			X				
Blackburnian Warbler	Dendroica fusca	M	I											X		X			
Blue-grey Gnatcatcher	Polioptila caerulea	R	I	X	X	X	X	X	X	X	X	X	x	X	X	X	X	X	X
Brown-headed Nuthatch	Sitta pusilla	R	I									X							
Blue-headed Vireo	Vireo solitarius	M	I													X	X		
Blue Jay	Cyanocitta cristata	R	O	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Black Vulture	Coragyps atratus	R	C			X							X						
Blackpoll Warbler	Dendroica striatta	M	I															X	
Brown Creeper	Certhia americana	R	I														X		
Brown Thrasher	Toxostoma rufum	R	O			X	X	X				X	X						
Boat-tailed Grackle	Quiscalus major	R	O			X	X			X	X	X	X					X	
Black-throated Green Warbler	Dendroica virens	M	I														X		
Black and White Warbler	Mniotila varia	M	I				X	X	X	X			X	X	X	X	X	X	X

R = residents, W = wintering migrants, M = Neotropical migrants. I = insectivore, O = omnivore, G = granivore, C = carnivore, P = piscivore.

Appendix A. Bird distribution among sites (Continued)

Common Name	Scientific Name	Migratory Status	Feeding Strategy	Sample Site Species Presence															
				1970-1	1970-2	1970-3	1980-1	1980-2	1980-3	1990-1	1990-2	1990-3	2000-1	2000-2	2000-3	Ref-3	Ref-4	Ref-5	Ref-6
Carolina Chickadee	Poecile carolinensis	R	O	X	X	X	X	X	X							X	X	X	X
Carolina Wren	Thryothorus ludovicianus	R	I	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Cedar Waxwing	Bombycilla cedrorum	R	G							X			X	X			X		
Cape May Warbler	Dendroica tigrina	M	I													X			
Common Grackle	Quiscalus quiscula	R	O							X		X	X		X				
Cooper's Hawk	Accipiter cooperii	R	C	X															
Common Nighthawk	Chordeiles minor	M	I															X	
Common Yellowthroat	Geothlypis trichas	M	I	X							X			X			X		X
Chestnut-sided Warbler	Dendroica pensylvanica	M	I								X		X				X		
Downy Woodpecker	Picoides pubenscens	R	I	X	X		X	X		X	X	X	X	X	X	X	X	X	X
Eastern Phoebe	Sayornis phoebe	M	I	X		X	X				X		X	X		X	X	X	
Eastern Kingbird	Tyrannus tyrannus	W	I													X		X	
Eastern Towhee	Pipilo erythrophthalmus	R	I																X
European Starling	Sturnus vulgaris	R	O					X											
Fish Crow	Corvus ossifragus	R	O	X	X	X		X	X		X			X		X		X	
Great Blue Heron	Ardea herodias	R	P							X				X	X				X
Grey Catbird	Dumetella carolinensis	W	O		X	X	X	X	X	X	X	X	X	X	X				X
Great Egret	Ardea alba	R	P	X								X	X	X			X		
Gray Kingbird	Tyrannus dominicensis	M	I													X			
Hairy Woodpecker	Picoides villosus	R	O			X									X		X	X	

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Appendix A.   Bird distribution among sites (Continued)

Common Name	Scientific Name	Migratory Status	Feeding Strategy	Sample Site Species Presence															
				19 70-1	19 70-2	19 70-3	19 80-1	19 80-2	19 80-3	19 90-1	19 90-2	19 90-3	20 00-1	20 00-2	20 00-3	Ref -3	Ref -4	Ref-5	Ref-6
Hermit Thrush	Cartharus mimus	M	O	X				X					X						X
Little Blue Heron	Egretta caerulea	R	P											X					
Limpkin	Aramus guarauna	R	P	X															
Louisiana Waterthrush	Seiurus motacilla	M	I															X	X
Mourning Dove	Zenaida macroura	R	G													X	X		
Magnolia Warbler	Dendroica magnolia	M	I																X
Northern Cardinal	Cardinalis cardinalis	R	O	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Northern Flicker	Colaptes auratus	R	O	X	X	X	X		X			X					X	X	
Northern Mockingbird	Mimus polyglottos	R	O					X	X	X	X	X	X		X				
Northern Parula	Parula americana	M	I	X	X	X	X	X	X	X	X	X		X	X	X	X	X	X
Northern Waterthrush	Seirus noveboracensis	M	I													X			
Osprey	Pandion haliaetus	R	P							X									
Ovenbird	Seiurus aurocapilla	M	I									X		X		X	X	X	X
Palm Warbler	Dendroica palmarum	W	I	X							X	X	X	X	X	X	X	X	X
Philadelphia Vireo	Vireo philadelphicus	M	I													X			
Pine Warbler	Dentroica pinus	W	I													X		X	X
Pileated Woodpecker	Dryocopus pileatus	R	O	X	X	X	X	X	X	X	X		X	X	X	X	X	X	X
Prothonotary Warbler	Protonotaria citrea	M	I								X			X		X	X	X	X
Prairie Warbler	Dendroica discolor	W	I					X			X	X	X	X		X	X	X	X
Red-breasted Nuthatch	Sitta canadensis	W	I														X		

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Appendix A. Bird distribution among site (Continued)

Common Name	Scientific Name	Migratory Status	Feeding Strategy	Sample Site Species Presence															
				1970-1	1970-2	1970-3	1980-1	1980-2	1980-3	1990-1	1990-2	1990-3	2000-1	2000-2	2000-3	Ref-3	Ref-4	Ref-5	Ref-6
Red-bellied Woodpecker	Melanerpes carolinus	R	O	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
Ruby-crowned Kinglet	Regulus calendula	W	I											X		X	X		
Red-eyed Vireo	Vireo olivaceus	M	I	X	X		X									X	X	X	X
Red-shouldered Hawk	Buteo lineatus	R	C	X	X	X	X	X	X		X	X	X	X		X	X	X	X
Red-winged Blackbird	Agelaius phoeniceus	M	O											X					
Scarlet Tanager	Piranga olivacea	M	I													X			X
Sora	Porzana carolina	W	O	X															
Swamp Sparrow	Melospiza georgiana	M	I																X
Tufted Titmouse	Baeolophus bicolor	W	O	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X
White-eyed Vireo	Vireo griseus	W	I					X		X		X		X		X	X	X	X
White Ibis	Bubulcus ibis	R	O		X							X	X						
Wood Duck	Aix sponsa	R	O									X							
Wood Stork	Mycteria americana	R	O			X													
White-throated Sparrow	Zonotrichia albicollis	R	O						X										
Yellow-breasted Chat	Icteria virens	M	I													X		X	
Yellow-billed Cuckoo	Coccyzus americanus	M	O													X			
Yellow-bellied Sapsucker	Sphyrapicus varius	W	O									X	X	X		X			
Yellow-rumped Warbler	Dendroica coronata	W	I							X	X	X		X	X	X	X	X	X
Yellow-throated Vireo	Vireo flavifrons	M	I	X											X	X	X	X	X
Yellow-throated Warbler	Dendroica dominica	W	I							X	X		X	X		X	X	X	X

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