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SPATIOTEMPORAL VARIATION OF AVIAN POPULATIONS WITHIN
GEOGRAPHICALLY ISOLATED FRESHWATER MARSHES

by

BRIAN LEE RODENBECK
B.S. University of Central Florida, 2001

A thesis submitted in partial fulfillment of the requirements
for the degree of Master of Science
in the Department of Biology
in the College of Sciences
at the University of Central Florida
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ABSTRACT

Metacommunity connectivity, i.e., multi-species dispersal events, is vital to metapopulation persistence in patchy landscapes. Assessments of metacommunity connectivity are not trivial. However, a relationship between trophic rank and the species-area relationship has been found in previous studies, allowing for the use of the predator species-area relationship to act as a surrogate measure of actual metacommunity connectivity of prey species in some systems. For this study, avian species were selected as they are generalist top predators within the study system. Predator species richness within geographically isolated freshwater marshes is influenced by a number of factors. I explore the relative roles of patch area, seasonality, hydroperiod, isolation, and vegetation structure on habitat use in the isolated freshwater marshes embedded within the dry prairie ecosystem of Central Florida. Predator species richness was surveyed in 50 sites for three seasons: fall 2005, winter 2005/06, and spring 2006 and the observed avian assemblage measures were subdivided into foraging guilds for analysis. Wading guild (e.g., egrets, herons, bitterns) species richness was correlated with hydroperiod and vegetation structural variables while perching guild (e.g., blackbirds, sparrows, meadowlarks) species richness was correlated with isolation, hydroperiod, and area annually. Overall predator and all guild species richness measures were also correlated with patch area for all seasons. These results suggest that while a complex mixture of patch area, hydroperiod and isolation influence habitat utilization that varies by season and at the community, guild and individual species level, the underlying predictors that define habitat use in wetlands annually includes hydroperiod, and is not exclusively patch area. Additionally, seasonal differences in predator species richness were found to be significant in some cases indicating that future avian population studies may benefit by sampling outside of the normally studied spring breeding

season. Results of this study support the use of predator species richness as a suitable assay of metacommunity connectivity of prey species. Applications and implications of this approach toward future conservation efforts are discussed.

*For David,
I am a better person for having known you*

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LIST OF ABBREVIATIONS

Abbreviation	Scientific Name	Common Name
AMBI	<i>Botaurus lentiginosus</i>	American Bittern
AMCR	<i>Corvus brachyrhynchos</i>	American Crow
AMRO	<i>Turdus migratorius</i>	American Robin
BAEA	<i>Haliaeetus leucocephalus</i>	Bald Eagle
BEKI	<i>Ceryle alcyon</i>	Belted Kingfisher
BLVU	<i>Coragyps atratus</i>	Black Vulture
BTGR	<i>Quiscalus major</i>	Boat-tailed Grackle
BWTE	<i>Anas discors</i>	Blue-winged Teal
CAWR	<i>Thryothorus ludovicianus</i>	Carolina Wren
CGDO	<i>Columbina passerine</i>	Common Ground-dove
CONI	<i>Chordeiles minor</i>	Common Nighthawk
COYE	<i>Geothlypis trichas</i>	Common Yellowthroat
CRCA	<i>Caracara plancus</i>	Crested Caracara
EAME	<i>Sturnella magna</i>	Eastern Meadowlark
EAPH	<i>Sayornis phoebe</i>	Eastern Phoebe
EATO	<i>Pipilo erythrophthalmus</i>	Eastern Towhee
GBHE	<i>Ardea herodias</i>	Great Blue Heron
GRCA	<i>Dumetella carolinensis</i>	Gray Catbird
GREG	<i>Ardea alba</i>	Great Egret
HOME	<i>Lophodytes cucullatus</i>	Hooded Merganser
KILL	<i>Charadrius vociferous</i>	Killdeer
KIRA	<i>Rallus elegans</i>	King Rail
LBHE	<i>Egretta caerulea</i>	Little Blue Heron
LEBI	<i>Ixobrychus exilis</i>	Least Bittern
LESC	<i>Aythya affinis</i>	Lesser Scaup
LOSH	<i>Lanius ludovicianus</i>	Loggerhead Shrike
MAWR	<i>Cistothorus palustris</i>	Marsh Wren
MODO	<i>Zenaida macroura</i>	Mourning Dove
MODU	<i>Anas fulvigula</i>	Mottled Duck
NOBO	<i>Colinus virginianus</i>	Northern Bobwhite
NOCA	<i>Cardinalis cardinalis</i>	Northern Cardinal
NOMO	<i>Mimus polyglottos</i>	Northern Mockingbird
OSPR	<i>Pandion haliaetus</i>	Osprey
PUGA	<i>Porphyryula martinica</i>	Purple Gallinule

RBWO	<i>Melanerpes carolinus</i>	Red-bellied Woodpecker
RSHA	<i>Buteo lineatus</i>	Red-shouldered Hawk
RWBL	<i>Agelaius phoeniceus</i>	Red-winged Blackbird
SACR	<i>Grus canadensis</i>	Sandhill Crane
STHA	<i>Buteo brachyurus</i>	Short-tailed Hawk
SWSP	<i>Melospiza georgiana</i>	Swamp Sparrow
TRSW	<i>Tachycineta bicolor</i>	Tree Swallow
TUVU	<i>Cathartes aura</i>	Turkey Vulture
WHIB	<i>Eudocimus albus</i>	White Ibis
WISN	<i>Gallinago gallinago</i>	Wilson's Snipe
WOST	<i>Mycteria americana</i>	Wood Stork

CHAPTER ONE: INTRODUCTION

One of the larger issues that face ecological researchers this century is answering the broad scale questions posed by the relatively young sub-discipline of conservation biology. What are the effects of anthropogenic changes, e.g. habitat loss and fragmentation, on ecosystem biodiversity and what approaches best serve to preserve biodiversity in reserve design? There is no simple answer to these questions. Earlier efforts in developing conservation policy relied heavily on the theory of island biogeography (MacArthur & Wilson 1967), resulting in larger and more continuous patches being preferred over smaller patches for preserving biodiversity. Conflicting models and studies contesting the concept of large mainland populations that are resistant to extinction, a cornerstone of the theory, and a lack of explicit spatial context within the theory led to a paradigm shift in the early nineties (for a good review, see Hanski & Gilpin 1997). Another theory was put forward as a more effective model, the metapopulation concept (Levins 1969). The theory, which states that populations in fragmented landscapes exist as local populations within patches that are connected through some level of migration to form a metapopulation, allows for the inclusion of habitat spatial context within the landscape and a single species level perspective in reserve design formerly not possible in island biogeography based studies. The widespread applicability of this concept has led to an explosion in its citation within the scientific literature in the past decade (Hanski & Gaggiotti 2004, Kritzer & Sale 2006). A couple of the foci that have developed in its application to reserve design is the concept of landscape connectivity (Tischendorf & Fahrig 2000) and the applications of corridors (Noss 1993; Noss & Daly 2006). While much has been published on these topics, there is little agreement on how to most effectively measure connectivity (Moilanen & Nieminen 2002, Calabrese & Fagan 2004) and the usefulness of corridors for conserving biodiversity (Beier

1998, Hannon & Schmiegelow 2002, Levey et al. 2005). A primary problem is that these concepts are innately species specific in both how they are interpreted and applied. A potential solution is to look beyond these concepts as metapopulation scale measures and focus on a metacommunity-based approach.

Metacommunities are defined as a set of local communities, linked by dispersal of multiple species that all have the potential to interact with one another (Wilson 1992). Possibly the greatest weakness of the metapopulation concept is its single species focus. Past studies of metapopulations examining the effects of landscape fragmentation have led to sometimes unexpected observations in the responses of individual species and biodiversity (Lindenmayer & Franklin 2002; Debinski & Holt 2000). Metapopulation models do not account for the interactions of many species existing at multiple trophic levels, interactions that can have significant impacts on metapopulation persistence (Harrison & Taylor 1997). To address this trophic complexity, the development of a framework for study of community ecology at large spatial scales is a necessity (for a good review, see Holyoak et al. 2005). Unfortunately, implementation of this framework into empirical studies has been difficult and not widely attempted. Few examples, i.e. studies that are explicitly defined as metacommunity studies, can be found in the literature.

One approach to exploring metacommunity structure has been through food web dynamics. Studies of food webs and metacommunities have largely been separated until recently (Holt & Hoopes 2005), with metacommunities contextualized by spatial characterizations through island biogeography theory and species dispersal (Volkov et al. 2003; Lobel et al. 2006; Richter-Boix et al. 2007) and food web dynamics focused on trophic structure and predator-prey interactions within the community, without concern for spatial influences (Polis et al. 1997;

Finlay et al. 2002; Polis et al. 2004). A full discussion of the integration of these two fields of study is beyond the scope and focus of this paper, but recent work has led to the synthesis of what has been termed “trophic island biogeography” by Holt & Hoopes (2005). The relationship of trophic rank to the species-area relationship is described in greater detail in an earlier paper by Holt et al. (1999), in which the species richness of predators are found to be correlated with the species-area relationship of their prey. The strength of this correlation varies, dependent on the whether systems are dominated by specialist or generalist predators (Figure 1). In this study, species richness of generalist predators will be investigated in habitat patches subject to cyclic stochastic extinctions of local populations for most prey species. Presence/absence of prey species resources within patches is indicative of dispersal success or failure. Given the established qualitative relationship found in previous studies (Figure 1), the observed predator species-area relationship will be tested as a surrogate measure of metacommunity connectivity for prey populations.

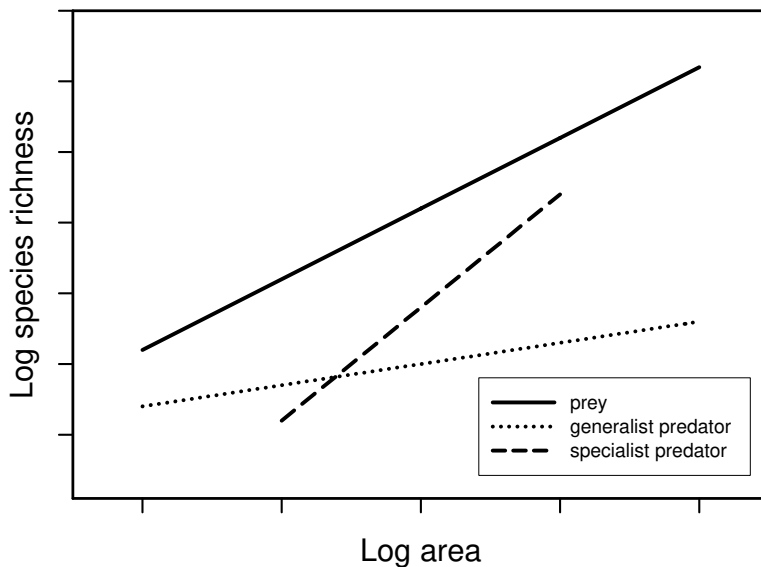


Figure 1. Species richness-area relationship as a function of trophic rank and level of specialization (adapted from Holt & Hoopes 2005)

Birds are a diverse, highly mobile and widespread taxon. They can be found in a variety of ecosystems, spanning the globe from tropical rainforests along the equator to the vast expanses of tundra above the Arctic circle. Some are heavily specialized and adapted to specific environments (Dugger et al. 2005, Koenig 2005, Breininger et al. 2006, Forcada et al. 2006), while others are habitat generalists that travel great distances and utilize numerous distinct ecosystems along the way (Griffis-Kyle & Beier 2005, Jones et al. 2005). Due to their dispersal ability, they are largely exempt from factors that influence habitat selection for other species, i.e. many of their prey species, in patchy and fragmented landscapes. Their presence in a habitat patch is related to the resources that are available and not a result of the presence or absence of dispersal barriers between patches. In the selected system of study, they form an assemblage of generalist predators at the top of the food chain, making them ideal for study (Appendix A).

The selected system of study, geographically isolated freshwater marshes, was chosen for three reasons. First, defined as wetlands that are completely surrounded by upland systems (Tiner 2003), these ecosystems may have hydrologic connections through ground water flows and periodic sheet flow, making them essentially wetland habitat islands within an upland matrix that form a naturally patchy landscape. A naturally patchy habitat type is vital to avoid including habitat loss as a factor that may influence metacommunity connectivity, confounding results. Second, these wetlands are often settings of high biodiversity (Scheffers et al. 2006) and endemism (Leibowitz 2003), and previous studies have shown isolated wetlands to support diverse communities of invertebrates, amphibians, birds and mammals (Gibbs 1993, Naugle et al. 1999, Brooks & Doyle 2001, Bradford et al. 2003, Gibbons et al. 2006, Schooley & Branch 2006, Zamudio & Wiczorek 2007). Additionally, in recent years, wetlands have come to be recognized for the importance of the various ecosystem services and vital habitat they provide

(Costanza et al. 1997, Postel & Carpenter 1997, Jackson et al. 2001, Hansson et al. 2005). These characteristics make this system exceedingly relevant for study in the development of a new measure with such heavy implications for assessing habitat conservation values. Finally, for this study to work, local communities, i.e. most of the prey species, must undergo cyclic stochastic extinctions. Without regular extinctions, the existence of metacommunity connectivity (multi-species dispersal events) cannot be confirmed to occur, as local communities may persist. The ephemeral nature of the study system assures that only populations of more sessile organisms persist, i.e. plant species, with the majority of prey species being reliant on water as a limiting factor and subject to cyclic stochastic extinctions.

Ultimately, the outcome of this study is three fold. The first objective is to investigate the role of area along with other multi-scale factors in establishing metacommunity dynamics within isolated freshwater marshes. Focus is placed on measuring multi-scale variables that may contribute to foraging habitat use and include in addition to patch area: seasonality, isolation, hydroperiod, water depth, percent cover, percent patch edge, and edge stem density. While the role of patch area, has long been associated with measures of species richness across a wide spectrum of taxa, other variables may also prove useful as predictors of species richness and are worthy of investigation. The island biogeography model (MacArthur & Wilson 1967) has served as such a dominating paradigm for various ecosystem studies and its application beyond oceanic islands and archipelagoes to terrestrial systems has become ubiquitous in ecological literature (Dunn & Loehle 1988, Bender et al. 1998, Krauss et al. 2003, Peintinger et al. 2003, Paracuellos 2006a, Skorka et al. 2006). However, while the species-area relationship is well-established (Arrenius 1921, Simberloff 1976, Triantis et al. 2003, Watson et al. 2005, Paracuellos 2006a) and is a focus of this study in establishing a surrogate measure of metacommunity connectivity,

it may also create a miasma that conceals the importance of other variables such as scale, trophic interactions, levels of isolation, and ephemeral characters such as hydroperiod (the length of inundation) that can also greatly influence habitat occupation. For example, recent studies have tied hydroperiod to species richness for a number of prey organisms, ranging from plants and zooplankton to amphibians and mammals (Brooks and Doyle 2001, Weyrauch and Grubb 2004, Serrano & Fahd 2005, Tavernini et al. 2005, Battaglia and Collins 2006). The implication being that in wetland systems, hydroperiod may be a more accurate predictor of prey species richness and possibly an alternative to patch area in assessing metacommunity connectivity worth exploration.

The second objective is to measure the strength and validity of this approach toward measuring metacommunity connectivity. Patch area may serve as a suitable broad scale predictor for overall predator species richness, whereas other variables like hydroperiod, isolation and vegetation structure tend to act at another level, applicable toward finer scale measures including foraging guilds and presence/absence data of individual species (Mortberg 2001, Bradford et al. 2003, Paracuellos 2006b). This change of scale makes these measures important for studying populations in greater detail but tend to be more difficult to measure and usually require moderate to extensive field work. Thus, they are less attractive for use in both short and long term studies. The purpose of their inclusion in this study is to determine if results of finer scale measures coincide with the findings of the broader scale predator species-area relationship that is the focus of investigation.

Finally, the third objective is to form an argument for the use of this measure, i.e. predator/ higher trophic level species richness, in assessing metacommunity connectivity levels for forming conservation policy, for expanding the protections granted geographically isolated

wetlands, and for the use of predator species richness as an indicator of ecosystem function. Habitat loss is occurring at an unprecedented rate. The need for quick, effective measures to assess habitat conservation value is clear. While the potential level of information associated with this measure is clearly not as great as those found in metapopulation based approaches, it does allow for a simple assessment of multiple target conservation areas to direct further, more specific and finer scale assessment measures. In the end, expediting the identification of potential habitat patches for inclusion in reserve design is the goal of this study.

This study tested two null hypotheses concerning the relationship of predator species richness with patch area, isolation, hydroperiod and measures of vegetation structure in geographically isolated freshwater marshes.

H_{01} : Predator species richness is independent of patch area.

H_{02} : Predator species richness is independent habitat patch specific measures including of isolation, hydroperiod, and measures of vegetation structure.

CHAPTER TWO: METHODS

Study area

Conservation area description and history

Two conservation areas were selected for this study, Three Lakes Wildlife Management Area (TLWMA) and Kissimmee Prairie Preserve State Park (KPPSP) (Figure 2). Both areas consist of large tracts of dry prairie dotted with widespread isolated freshwater marshes. Dry prairie is an ecosystem that is endemic to Florida, distinguished from other Florida habitats by its vast treeless expanses and grass-covered nature (Taylor 1998). The ecosystem is thought to have a natural fire frequency of 1-3 years, but now is subjected primarily to prescribed burns during the winter months (Watts et al. 2004). This shrub-grassland landscape is dominated by wiregrass (*Aristida stricta*), saw palmetto (*Serenoa repens*), running oak (*Quercus pumila* & *Q. minima*) and a variety of clonal re-sprouting shrubs including gallberry (*Ilex glabra*), staggerbush (*Lyonia fruticosa*) and shiny blueberry (*Vaccinium myrsinites*) (Orzel & Bridges 2004). The term “dry” prairie can be misleading as this ecosystem can flood during periods of heavy rainfall and be subject to overland sheet flow, however the water table is usually well below ground level leading to near xeric conditions throughout most of the year (Abrahamson & Hartnett 1990). Both conservation areas have been subjected to significant levels of hydrologic manipulation that have affected the natural drainage of water across the landscape, but multiple efforts to restore hydrology of the dry prairie to a more natural state, i.e. allowing for greater sheet flow, are presently underway or planned (Florida Ecological Restoration Inventory, personal communications).

The more northern conservation area, TLWMA, was established in 1974 to protect and manage the wet prairies and freshwater marshes within the dry prairie ecosystem that provide for

natural flood storage. Formally known as the Three Lakes Ranch, it was purchased under the Environmentally Endangered Lands Program and is administered by the Florida Fish and Wildlife Conservation Commission (www.floridaconservation.org). The area allows for a number of recreation opportunities, including more intensive activities such as hunting and ATV use. Unfortunately, illegal off-road ATV and other vehicle use have led to some habitat degradation during the hunt seasons (personal observation).

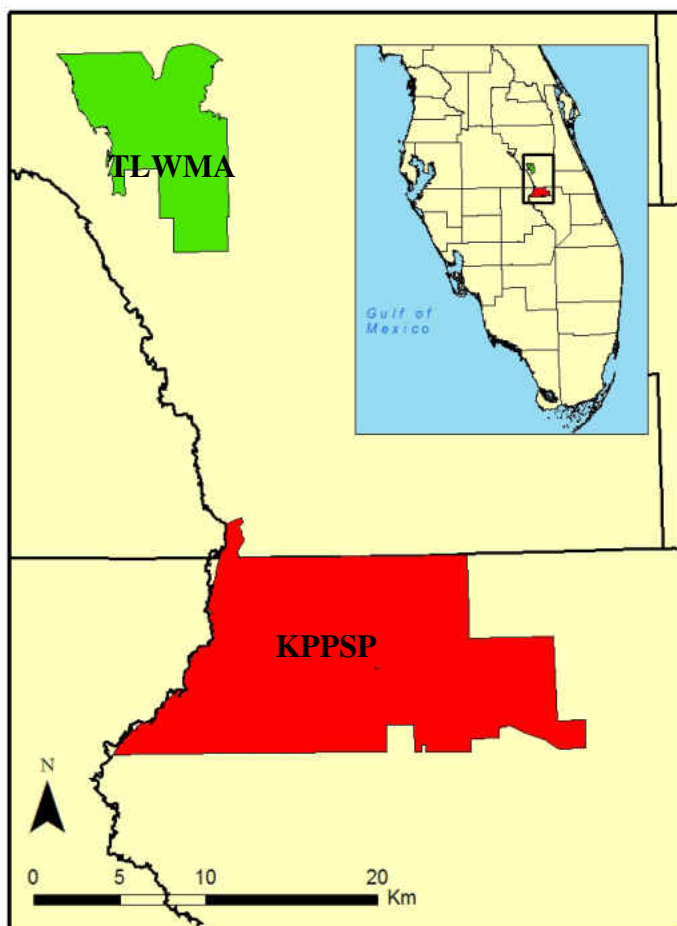


Figure 2. Located approximately 45 km south of Orlando, FL, Three Lakes Wildlife Management Area (TLWMA) and Kissimmee Prairie Preserve State Park (KPPSP) contain two of the largest tracts of remaining dry prairie ecosystem in Florida

The other study area, KPPSP, does not allow for hunting and strictly bans ATV use, with exceptions made for land managers and researchers, thus it has virtually no ongoing habitat degradation. This 18,500 ha preserve park was established 1998 and lies approximately 24 km to the south of TLWMA. While KPPSP policies do allow for a greater level of protection, this relatively new conservation area has been subjected to more recent cattle grazing, has over 2,500 ha of improved pasture lands, and more extensive hydrologic manipulations than TLWMA. However, habitat restorations are ongoing here with efforts focused on filling artificial ditches and controlling invasive species.

Wetland patch selection

National Wetlands Inventory (NWI) maps were the primary source of data used in the site selection process (U.S. Fish and Wildlife Service 2006). The first step in site selection was to determine the number and distribution of prospective freshwater marsh sites by patch area. Given the dominance of smaller wetland patches in the landscape (Figure 3), sorting by patch area was done to insure there was sampling of a continuum of wetland patches of various sizes. Site selection involved modification of NWI maps using ArcGIS 9.1 software (ESRI 2006). An altered version of a second GIS map, the Florida Natural Areas Inventory Management Areas (FNAIMA) map, which included only the two study conservation areas (Florida Natural Areas Inventory 2006), was used to modify the NWI maps so they only included wetland habitat within the targeted study areas. Next, a new attribute field was added to the modified NWI map, based on whether a wetland was either geographically isolated or connected. Geographically isolated wetlands were defined as a wetland patches that were entirely surrounded by upland systems. Any remaining wetlands not classified as isolated were considered geographically connected either by association with a body of water or being adjacent to a wetland categorized as

connected. High resolution aerials (Land Boundary Information System 2005) were overlaid with the modified NWI maps to confirm classification decisions.

The wetland patch area field attribute was used to generate a histogram of all isolated emergent wetlands within the conservation areas. This histogram was divided into five size classes with equal numbers of wetland patches (Figure 3). From each size class three groups of ten wetlands were randomly selected. One group of ten served as a sampling from the size class, while the other groups served as a pool of alternate sites should a selected wetland patch prove to be inaccessible for logistical reasons or highly disturbed, i.e. subjected to ditching, ATV damage,

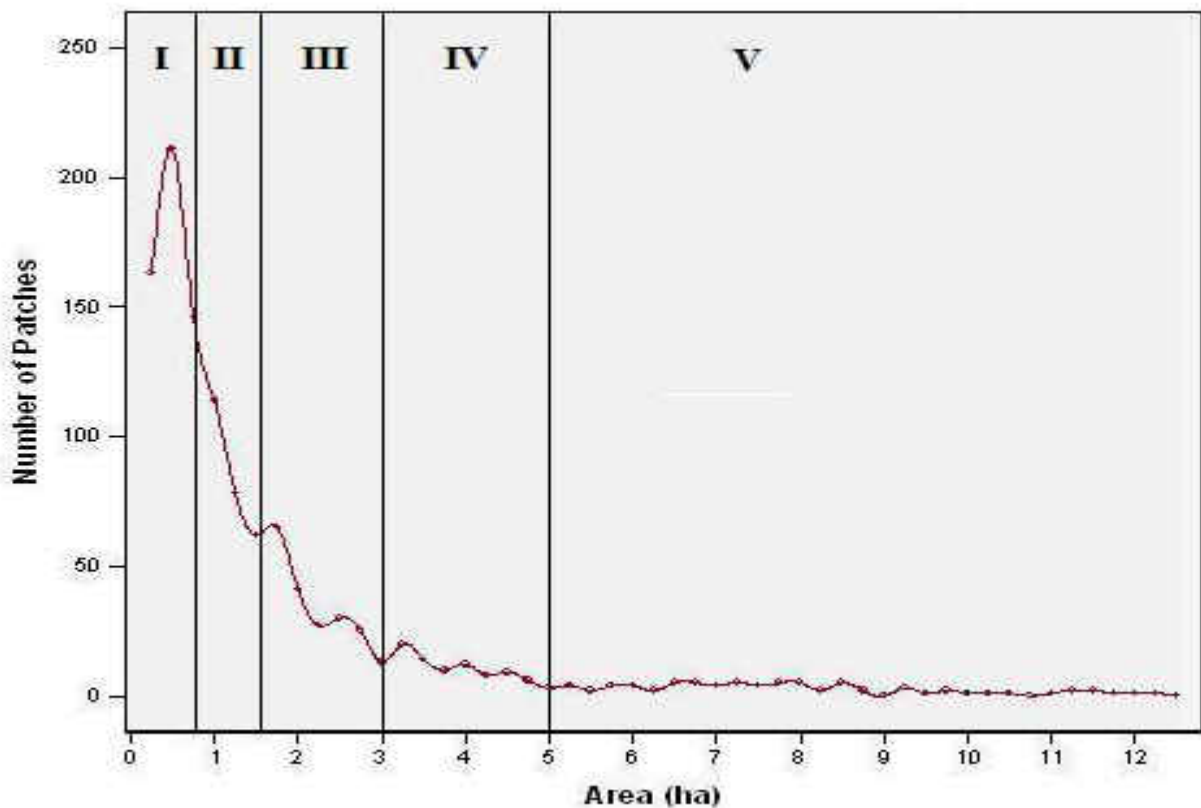


Figure 3. Five size classes for site selection determined from histogram results. Class I is comprised of all patches less than .75 ha in size, class II sites ranged between .75 and 1.5 ha, class III sites ranged between 1.5 and 3 ha, class IV sites ranged between 3 and 5 ha, class V was comprised of all remaining sites larger than 5 ha

adjacent to an improved road, or transected by a road. All sites were at least 250 m apart. Ultimately, 50 freshwater marsh sites were selected that ranged in size from 0.28 to 9.75 ha, with twenty-eight of these sites located at TLWMA and the remaining twenty-two sites at KPPSP (figures 4 & 5 and Appendix A). As site selection was completed, the final sites were numbered based on a two digit classification system composed of size class (1-5) (Figure 3) and order of confirmation (0-9). Sites were numbered as they were accepted, so while this classification system allows for a quick assessment of site size based on its number, sites are not ranked within classes by size from smallest to largest, i.e. site 18 is not larger than site 10, it was merely confirmed later in the selection process (Appendix A).

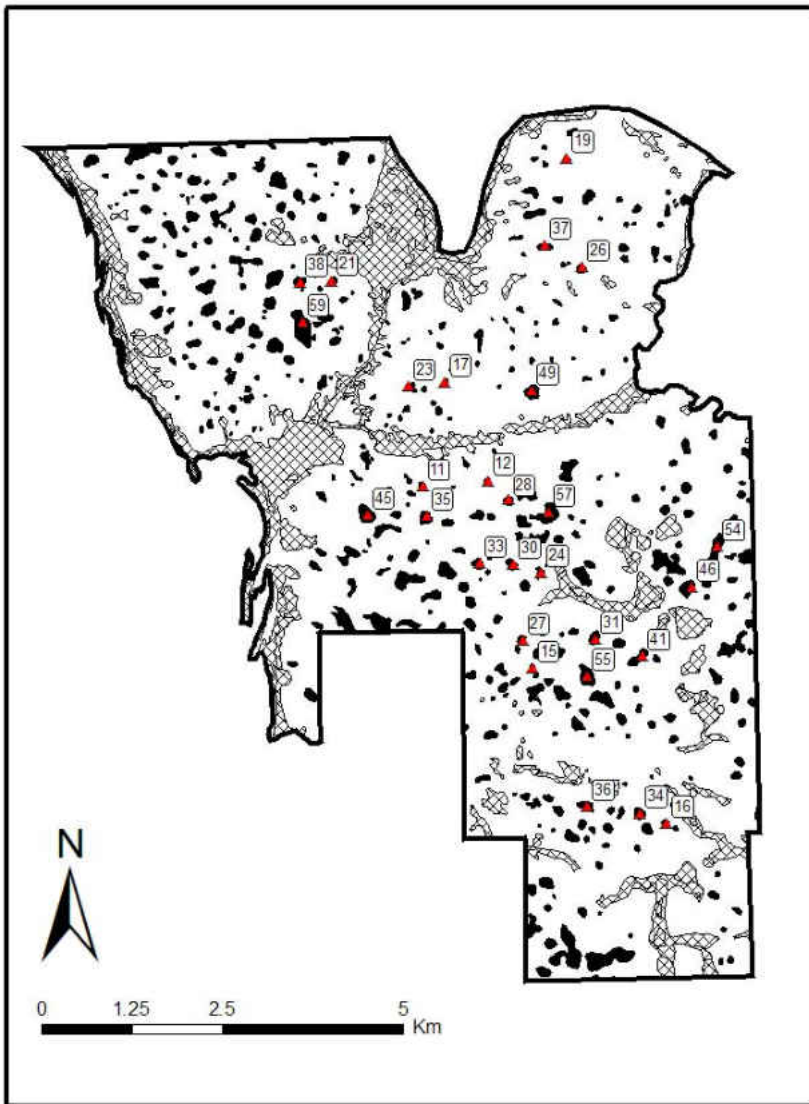


Figure 4. Geographically isolated wetlands (solid black polygons) and connected wetlands (cross-hatched polygons) at Three Lakes Wildlife Management Area. Red triangles and associated numbers indicate where sampling sites were located within the conservation area.

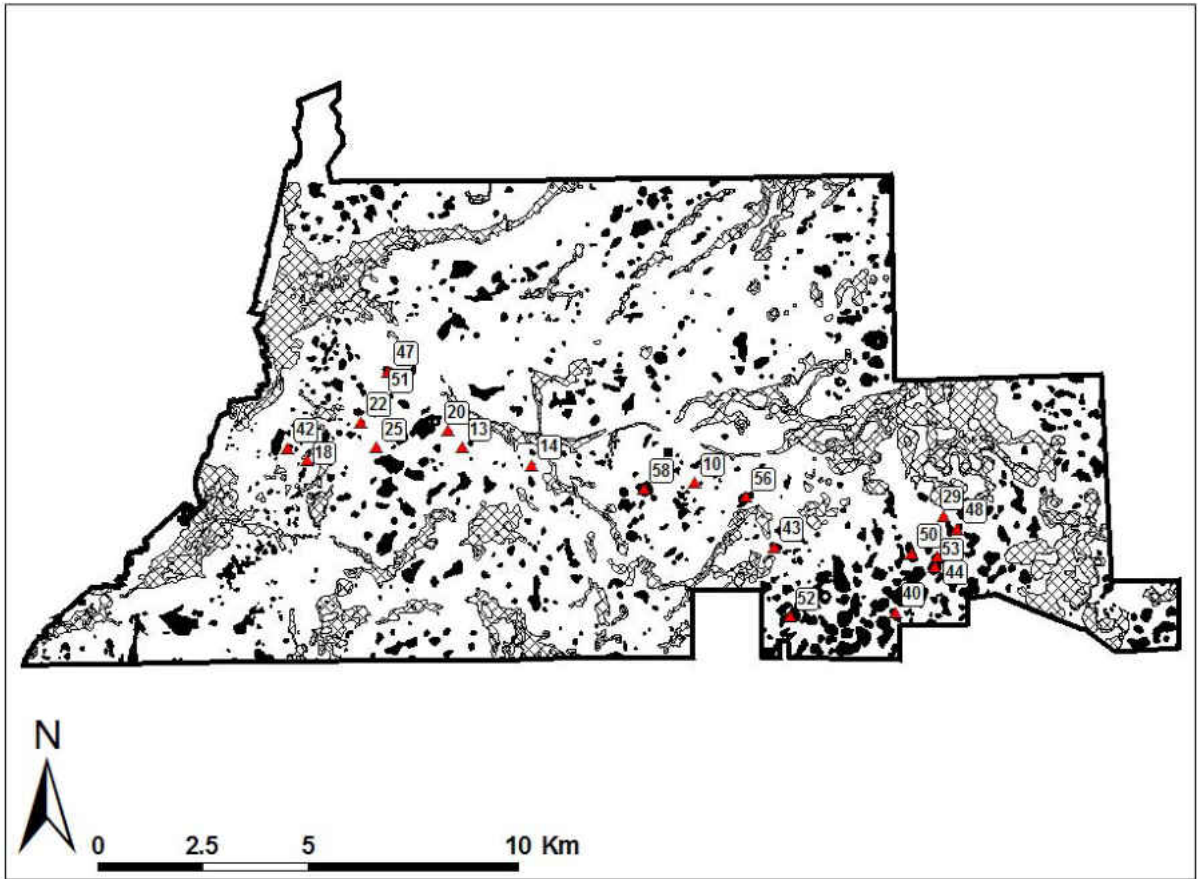


Figure 5. Geographically isolated wetlands (solid black polygons) and connected wetlands (cross-hatched polygons) at Kissimmee Prairie Preserve State Park . Red triangles and associated numbers indicate where sampling sites were located within the conservation area.

Finally, given the 1992 publication of the NWI maps, the information in these maps required verification. NWI maps have been shown in previous studies to be accurate (Kudray & Gale 2000, Stolt & Baker 1995). However, as an additional check for this study, published NWI data of wetland area and associated edge were verified by generating a digitized map from 2005 high resolution aerials and collecting GPS field measurements using a Trimble Geo XT for each selected site. These three measures of area and edge were then compared and tested for significant differences (Appendix B).

Bird survey methodology and preliminary assessments of accuracy

Area counts were conducted at each site and sampling design was primarily modeled after the methodology described by Bibby et al. (2000). Three visits were made to each site per season, usually within a four-week period. This was done for two reasons, to allow for detection of possible changes in site species composition through the season and due to study logistics. This was the shortest period possible in which all sites could be sampled to create a “snapshot” of the seasonal species richness. Each visit to a sampling site was at least seven to ten days apart and if multiple count stations were located at a site, then all stations, at that location, were visited on the same day.

Prior to starting the first season of sampling, count stations which would act as regular stopping points during counts were established with semi-permanent markers at each site. To compensate for differences in size of habitat patches, some sites had multiple count stations. Sites that were less than 2 ha had one count station, sites having an area between 2 and 5 ha had two stations, and sites that were greater than 5 ha in size had three stations (Huff et al. 2000). Each station marker was set at least 150 m from any other stations placed at a single site and stations were placed to allow for ease of access, most efficient travel between adjacent stations and changes in the order in which stations were visited. Finally, count stations were established at positions along the edge of the freshwater marsh sites that allowed for the greatest level of detection of bird species within or at the edge of the habitat patch.

Preliminary counts were conducted during the spring and summer seasons of 2005. The purpose of these preliminary counts were threefold: to familiarize myself with the study area, to familiarize myself with the avian community present within the dry prairie, and to ferret out possible weaknesses in the censusing protocols. Both study areas are vast expanses of land, with

site locations being widespread. Time was a limiting factor in effectively assessing species composition seasonally, knowledge of the exact locations of all sites and how to reach them as quickly as possible was necessary. This period served to allow the observer to get the “lay of the land” so to speak. This time also allowed for practicing counts and development of skills to quickly and accurately identify species. Finally, observations allowed for habitat specific count constraints to be determined and logistic problems in the study design to be discovered. As a result, several previously accepted sites were relocated due to logistical time constraints and an earlier decision to assess the bird community during the summer season was removed from the study design. Due to the size of the study area and the shorter observation period resulting from rapid temperature rise in the morning hours, summer counts were simply not feasible.

Sampling was spread across three seasons: fall 2005, winter 2005/06 and spring 2006, with the summer season being excluded from this study. Fall season counts were made between October 27th and November 30th, winter counts were made between January 21st and February 22nd, and spring counts were made between March 24th and April 29th. Observations were made starting at dawn and completed within four hours. Detections of birds within the site habitat patch were made by both visual and auditory confirmation. Counts were taken on days with no or light winds (<8 km/h) and in no or light fog when conditions for bird detection were considered acceptable. When conditions included heavy wind, rain, and/or temperatures greater than 29°C, counts were not conducted. The decision to stop counts if temperatures exceeded 29°C was based on observed drops in avian activity above this temperature. This may be a result of a similar drop observed in activity of certain prey species (amphibians, flying insects) within sites. While it is unclear exactly why avian activity in isolated wetlands does drop off at 29°C, it was clear early in this study that little information on species composition would be gained with

observations made at higher temperatures. Finally, weather conditions and time of day of all censuses were recorded following Hanowski & Niemi (1995).

Each site visit consisted of a five minute count period split into two recording intervals, a three minute period spent at the count station, followed by a minimal two minute search period. An additional one minute period was included prior to the start of the count period to allow for a settling down time after reaching the count station (Gibbons & Gregory 2006). If any bird was detected while approaching the count station or flushed upon arrival, it was included in first three minutes of the count based on its approximate location within the site. During the count period, any species detected was recorded in one of five categories: flushed during count, within 50 m of station, greater than 50 m from station, aerial and associated with wetland site, or aerial and independent of site. For birds detected by auditory means, distance and location from observer was estimated. Additional time was included at the end of the 5-minute count period(s) to allow for a short walk through the wetland to flush and/or detect some of the more secretive birds, i.e. bitterns, snipes, etc., that hide in the vegetation and may not otherwise be counted. If a site had more than one count station, then the path between stations was slowly and quietly walked with any observations being split between adjacent stations. Any bird detected in this manner was counted in the flushed category. Throughout the count period within a single habitat patch, great effort was made to avoid double counting of individuals. Observed birds were tracked to the best of the observer's ability and if a bird was suspected of already being counted for any reason, it was not included. For the purposes of data analysis, birds counted as aerial and habitat independent were not included in species richness estimates and species presence/absence data. Finally, species richness measurements were subdivided by foraging guild and season for data analysis purposes. The three foraging guilds were based on the primary foraging characteristic

utilized by the species and consisted of swimmers, waders, and perchers (classification of specific species into guilds can be found in Appendix A).

Landscape variables

Measures of wetland patch area (m²) and wetland edge (m) for each site were taken from published NWI maps. Isolation indices were determined by utilizing the distance tool in ArcGIS 9.1 to determine minimum Euclidean distance between two wetland patches. Three measures of isolation were calculated, adapted from previous studies that utilized patch isolation indices in studies of other habitat types (Lynch & Whigham 1984, Opdam et al. 1985, Brown & Dinsmore 1986). The first measure, ISO_a was the distance in meters from a selected site to the nearest isolated wetland. The second measure, ISO_b was the distance in meters from a selected site to the nearest connected wetland. The final measure, ISO_c was the average distance to the nearest five isolated wetlands for each site (Figure 6). All isolation measures were between the selected site and wetlands having similar vegetation types (i.e. if a forest wetland was closest to a selected freshwater marsh, then it was not counted as it is likely to play host to different avifauna).

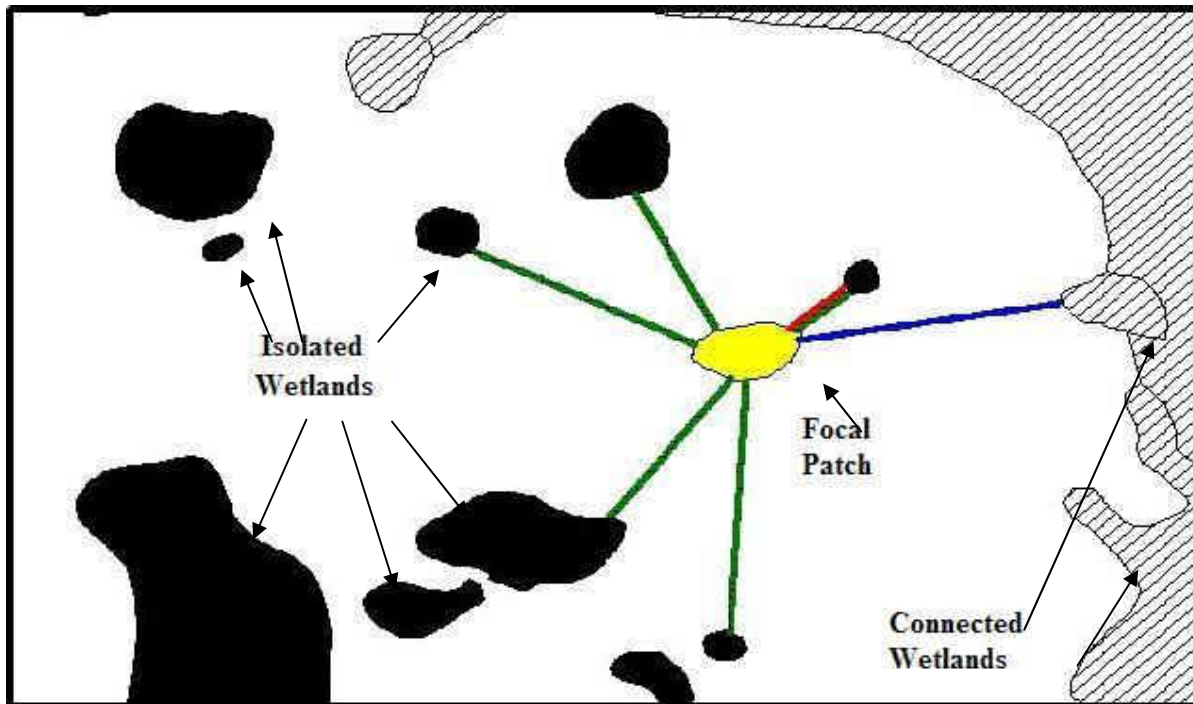


Figure 6. ISO_a is represented here by the red line, ISO_b is represented by the blue line, and ISO_c is the average distance of the five green lines.

Hydrological variables

Water depth measures were based on the water depth readings for maidencane (*Panicum hemitomen*) as this ecotope was present and usually the most dominant of all ecotopes at each site. Field measures were made with a 1.2 m staff gauge by wading into the wetland 2 m past the outer edge of any given ecotope. Initial depth measures were taken for each ecotope present (refer to vegetation variables section for an in depth description of ecotope structure in the studied freshwater marshes). However, after the first five measurements, only the depth of the maidencane ecotope was monitored for the remainder of the study. Maidencane generally was the inner most ecotope for each site, so initial measures of the associated outer ecotopes acted as a baseline used to calculate water depth during latter visits and estimates of the hydroperiod for

other ecotopes. Three measurements of the maidencane ecotope were taken during seasonal bird counts and were averaged to generate a seasonal depth measure for each site.

Measurement of hydroperiod, the total number of days per year of inundation, was based on yearlong measures of water depth for 26 sites at TLWMA taken at ten day intervals. Estimates for KPPSP sites were calculated as regular measures at these sites were not logistically possible. If a site dried out completely between observations, the date of this event was estimated. In conjunction with data on daily rainfall totals and weather conditions, estimates of hydroperiod were made for the remaining sites not regularly monitored. While these estimates of hydroperiod could not be used to predict water depth, they were found to be accurate through field observations during seasonal bird counts.

Vegetation structure

Percent cover measures were based on both high resolution aerials and field observations. Using a printed outline of the wetland patch perimeter, estimates of percent cover were made in the field by drawing the coverage of dominant ecotopes. Most wetland patches have well defined ecotopes that form in predictable patterns with sand cordgrass (*Spartina bakeri*) and wiregrass (*Aristida stricta*) taking peripheral positions as disjoint patches along the wetland edge. St. John's Wort (*Hypericum flasviculum*) usually forms a large, well-defined outer ecotope interspersed with beakrush species (*Rynchosporia spp.*). Finally, maidencane (*Panicum hemitomen*) and other panic grasses usually form the largest inner ecotope, with pickerelweed

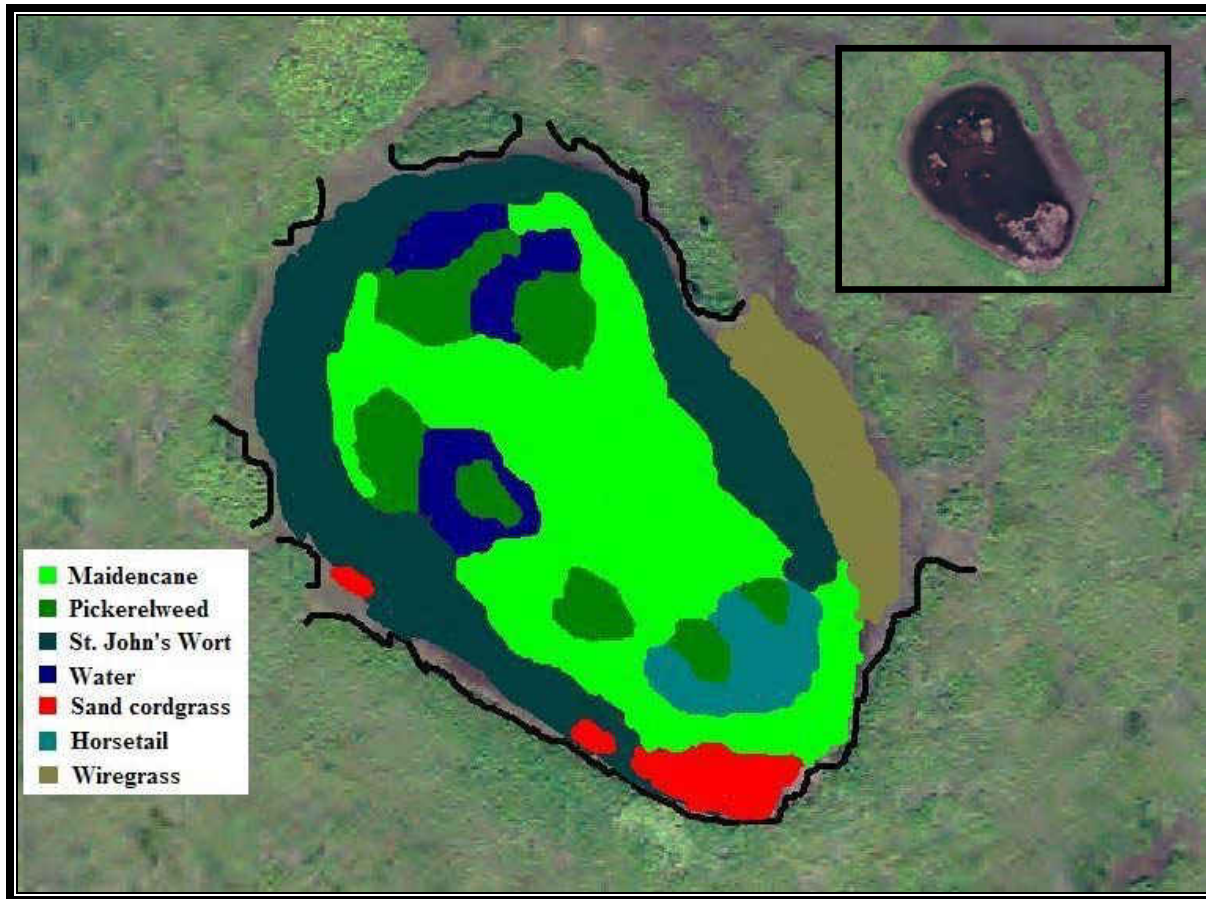


Figure 7. The finished percent cover map of site 47 at Kissimmee Prairie Preserve State Park. Ecotopes are mapped on the large, central map by distinct colors for each vegetation type. The inset provides a reference for comparison between the field map and what can be distinguished in the high resolution aerial.

(*Pontederia cordata*) forming either a much smaller continuous or many patchy zones within the maidencane ecotope. The rough field map was then used to generate second map using high resolution aeriels (Figure 7). Final percent cover estimates of species, vegetated/non-vegetated, and woody vegetation/non-woody vegetation coverage were made from these finished maps. Percent cover measures did not necessarily add up to 100% as some species ecotopes overlapped. Prior to data analysis, species percent coverage were converted using the Domin-

Krajina scale (Table 4, Appendix A) for visual estimates of cover-abundance (Bullock 2006, Levesque 1996).

Percent patch edge (PPE) was measured as percentage of wetland edge adjacent to edge habitat, where edge habitat was defined as vegetation that was at least one half meter taller than the vegetation of the surrounding upland and was denser, older growth vegetation than was found in the neighboring upland. This edge was likely an artifact of the limited protection provided by the wetland from frequent fire events. This protection could be a result of overflow flooding of the wetland into the adjacent upland or due to the wetland acting as shield between the upland vegetation and unidirectional fire sweeping across the prairie.

Vertical structure within the dry prairie ecosystem was uncommon and was often associated with the presence of freshwater marshes. These marshes likely acted to provide protection from frequent fires and allowed for the establishment of scattered individuals or groups of trees. For some avian species, the presence of this limited vertical structure may act as an incentive for utilization of the associated wetland. Vertical structure measures were based on assessments of dbh, two size classes were defined. The first size class, VSS, included all vertical stems with a dbh less than 5 cm and the second class, VSL, included stems having a dbh greater than 5 cm. The total number and location of stems for each size class were counted and mapped during field visits within three buffers for each site. These buffers extended out from each wetland site perimeter to 10, 20 and 30 m. Final vertical structure maps were generated using field notes and high resolution aerials prior to determining size class stem density per square meter for each buffer. An abbreviated listing of the previously described variables can be found in Table 1.

Statistical analyses

Area measure verification and correlations

Wetland patch area measures taken from NWI maps were compared with area measures from digitized 2005 aerials and GPS based maps using Paired-sample t-tests to determine if there was a significant difference between the area measures of any of these maps. Pearson's correlation test was utilized to determine if there was a correlation between patch area and seasonal guild species richness numbers, isolation, hydroperiod, and vegetation structure variables. If an area-species richness correlation is found, a model will be derived to generate estimates of avian species richness across the landscape. Prior to this test and for all remaining statistical analysis, data transformations were made for normality. Area, edge, and isolation measures were log transformed, PPE, percent cover, and hydroperiod measures were arcsine transformed, and VSS and VSL were transformed using the square root function. Finally, all avian species richness measures were transformed using the $\log(x+1)$ function. Measures of patch water depth and individual species presence/absence data were not transformed prior to data analysis.

Table 1. List of abbreviations and descriptions of multi-scale predictor variables

Variable	Scale	Description
Area	Landscape	size of habitat patch in hectares
ISO _a	Landscape	distance of patch from nearest geographically isolated patch with similar vegetation in meters
ISO _b	Landscape	distance of patch from nearest similar patch that is not geographically isolated in meters
ISO _c	Landscape	average distance of patch from five nearest similar patches that are geographically isolated in meters
Hyd_pan	Hydrological	length of inundation for patch <i>Panicum spp.</i> (Maidencane) ecotope in days
Hyd_pon	Hydrological	length of inundation for patch <i>Pontederia cordata</i> (Pickerelweed) ecotope in days
Hyd_hyp	Hydrological	length of inundation for patch <i>Hypericum flsaviculum</i> (St. John's Wort) ecotope in days
Hyd_spa	Hydrological	length of inundation for patch <i>Spartina bakeri</i> (sand cordgrass) ecotope in days
Water depth	Hydrological	depth of water in the maidencane ecotope
Percent cover	Community	percent of coverage of patch by a single type of vegetation
PPE	Community	percentage of patch edge that has a denser, taller vegetation than adjacent upland matrix
VSS	Community	density of small stems per m ² within 10 m of patch edge
VSL	Community	density of large stems per m ² within 10 m of patch edge

Avian community seasonal species richness and composition measures

Paired t-tests were used to assess if there were differences in seasonal species richness and to investigate possible changes in avian community composition by season. Species richness can change by season due to a number of factors including variability in prey availability, stochastic changes to habitat quality, and/or bolstered numbers from migratory species detected in the survey. Concern for the effect of migratory birds in particular on this study led to the

investigation of seasonal avian community composition as a possible source of differences in seasonal avian species richness. These tests would act as a test for significant seasonal shifts in avian community composition, and if species make up did not significantly change, allow for its elimination as a source of possible seasonal differences in species richness. A sequential Bonferroni test was run to adjust for error associated with multiple comparisons (Rice 1989).

Multicollinearity tests

Multicollinearity occurs when one predictor variable is nearly equivalent to the linear combination of other predictors. These near linear relationships between predictors can lead to erroneous results when calculating multiple linear equations. Collinear variables must be omitted prior to calculations to eliminate deleterious effects that can lead to errors in interpreting resulting equations. Two approaches were utilized to check for multicollinearity in the sampled factors. The first was a principal components analysis (PCA) based approach described by Iles (1993). A PCA was performed on the correlation matrix of the variables and then the square root of the resultant of the largest eigenvalue divided by the smallest eigenvalue was calculated to produce the condition number. If this number exceeded 10, the variables were considered to likely be multicollinear. If the number was less than 10, then the variables were deemed acceptable. In cases where multicollinearity was detected, collinear variables were determined from the resulting output of the PCA conducted on the correlation matrix. The collinear variables were then plotted against each other in a scatter plot to confirm collinearity. One of the collinear variables was then dropped and the PCA was run again with all the remaining variables to determine the resulting condition number. This was done twice, once for each collinear variable. The PCA that produced the lower condition number was accepted and the collinear variable was

dropped from future analysis. The second approach used to test for multicollinearity was the Pearson's test of correlation (Naugle et al. 1999). Sets of similar factors (i.e., percent cover variables, isolation indices) were tested and if any pair of factors were found to have an r value greater than 0.3, then one or more of those variables was dropped prior to further data analysis.

Determination of variables related to species richness

For each season, guild species richness and all remaining landscape, hydrological, and vegetation structure variables that were not found to be collinear were entered into a forward stepwise multiple linear regression using the SPSS 13.1 statistical analysis software. Akaike's Information Criterion (AIC) was used to determine the best model (Burnham & Anderson 1998). Finally, results of these regressions were then used to assess the strength of the area-species richness relationship and support the strength of using patch area and species richness correlation data to produce maps for projecting seasonal guild species richness within isolated wetlands for each conservation area (Figure 8).

Determination of variables related to species presence

For each season, presence/absence data for each species were entered into a forward stepwise logistic regression with all remaining community and landscape scale variables after multicollinearity testing. The JMP 6.0 software package was used to run all logistic regressions. Significant variables remaining after analysis were then used to produce graphs to estimate the probability of occurrence for individual species by season. Finally, if possible, effective seasonal and annual habitat sizes were estimated from these graphs and a patch would be considered part of the effective species habitat when probability of occurrence was greater than 0.5 (Figure 9).

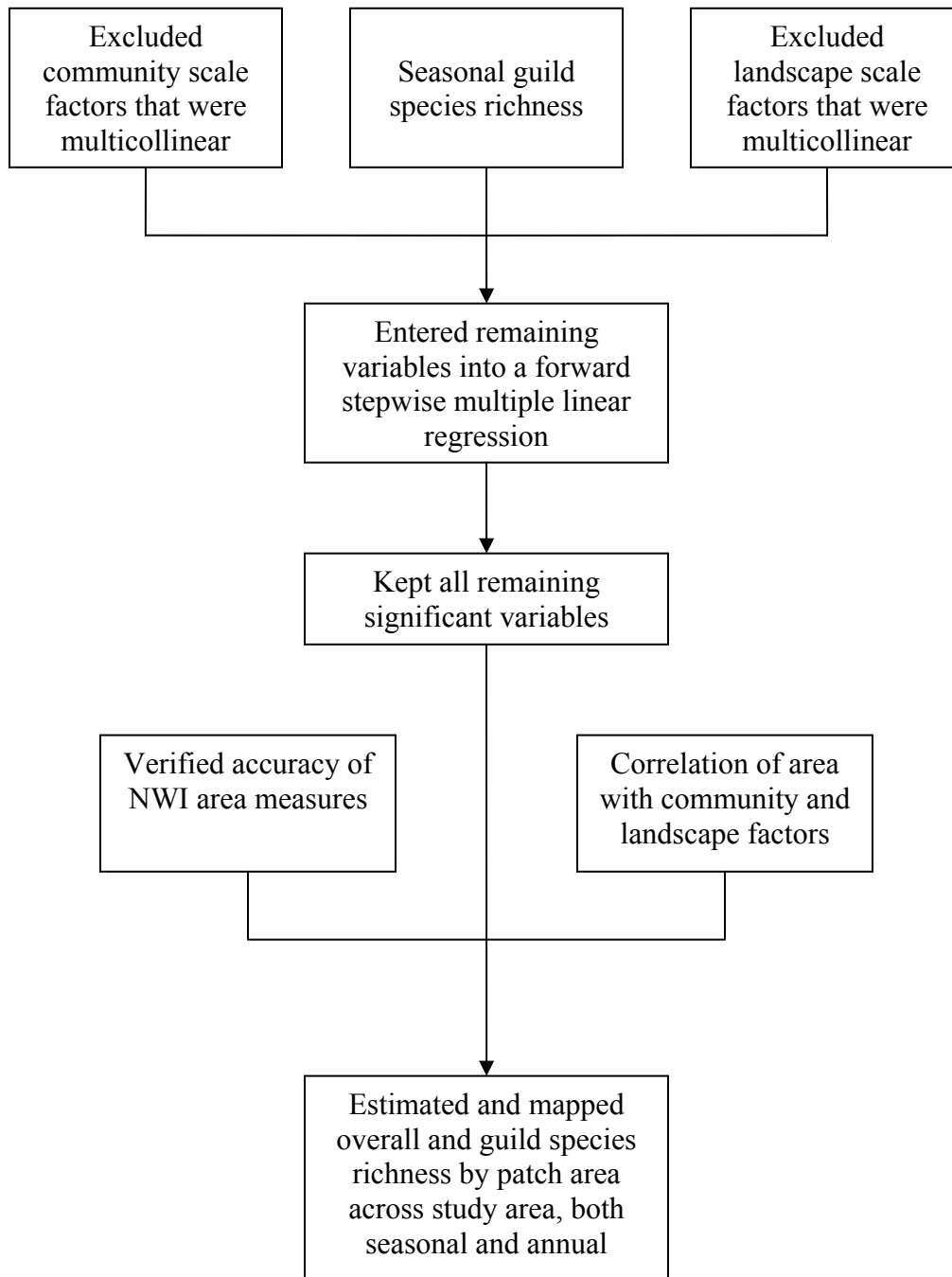


Figure 8. Flow chart showing the variable selection for forward stepwise multiple linear regressions. Following verification of the accuracy of NWI data and correlation of patch area to other community and landscape scale variables, the results of the linear regression were then used to confirm the strength of the area-species richness relationship and its validity for mapping projected patch species richness across the study area

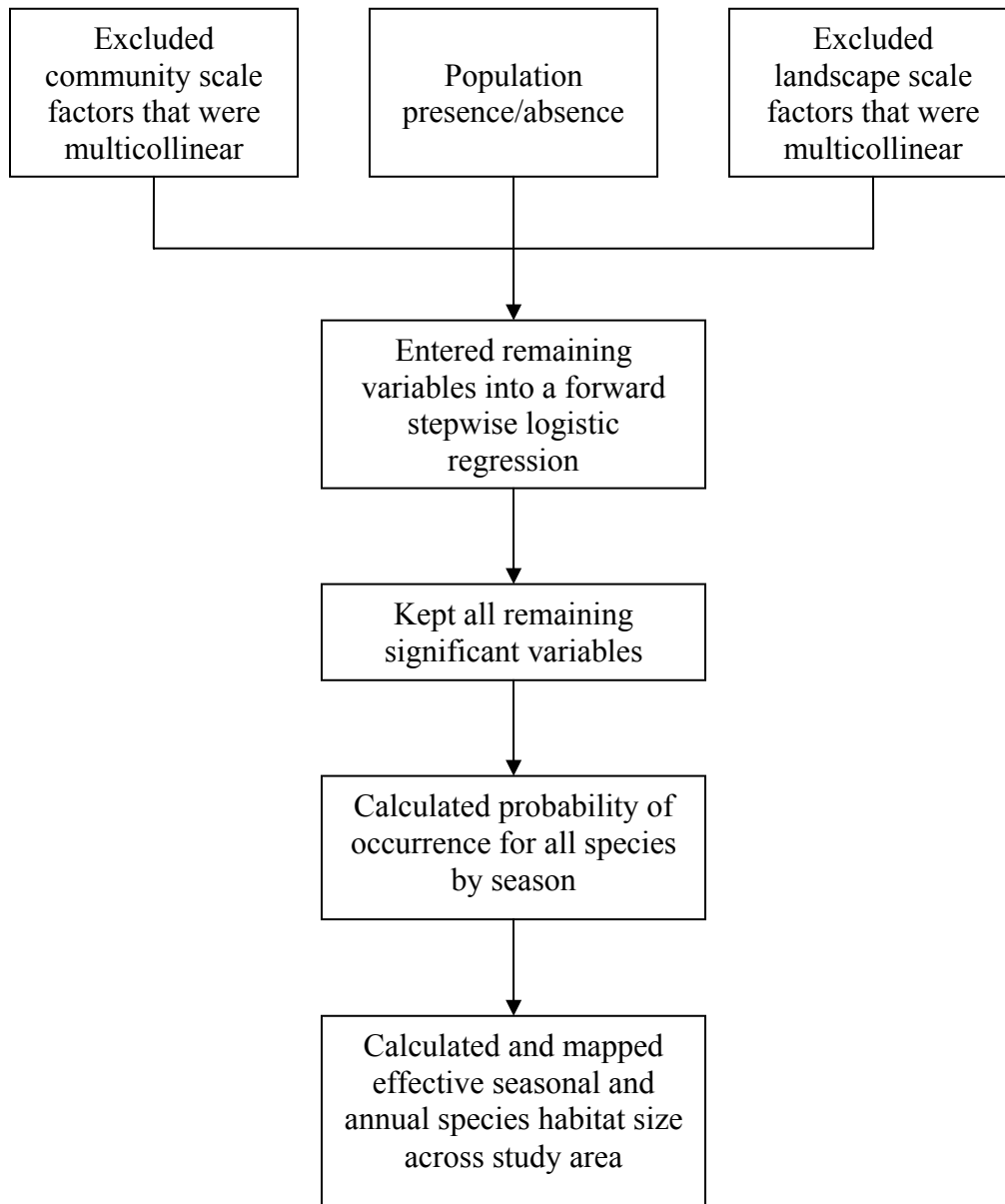


Figure 9. Flow chart showing the variable selection for forward stepwise logistic regression. Probability of occurrence was determined seasonally for each species and then probability of occurrence was used to estimate and map effective seasonal and annual species habitat sizes across the study area.

CHAPTER THREE: RESULTS

Avian richness

Avian species richness varied greatly by foraging guild and by season (Figure 10). Birds classified as perchers showed the greatest species richness for each season and for all foraging guilds, species richness peaked during the winter season. Overall, 45 species of avifauna were detected during sampling. Broken down by foraging guilds, there were five swimming, 11 wading, and 29 perching species. Of the 45 species detected, 25 were only detected from one to three times during the entire nine-month sampling period, classifying them as either difficult to detect or transient species. Despite this variability, there were no significant differences in overall avian species composition between seasons based on paired sample t-tests. However,

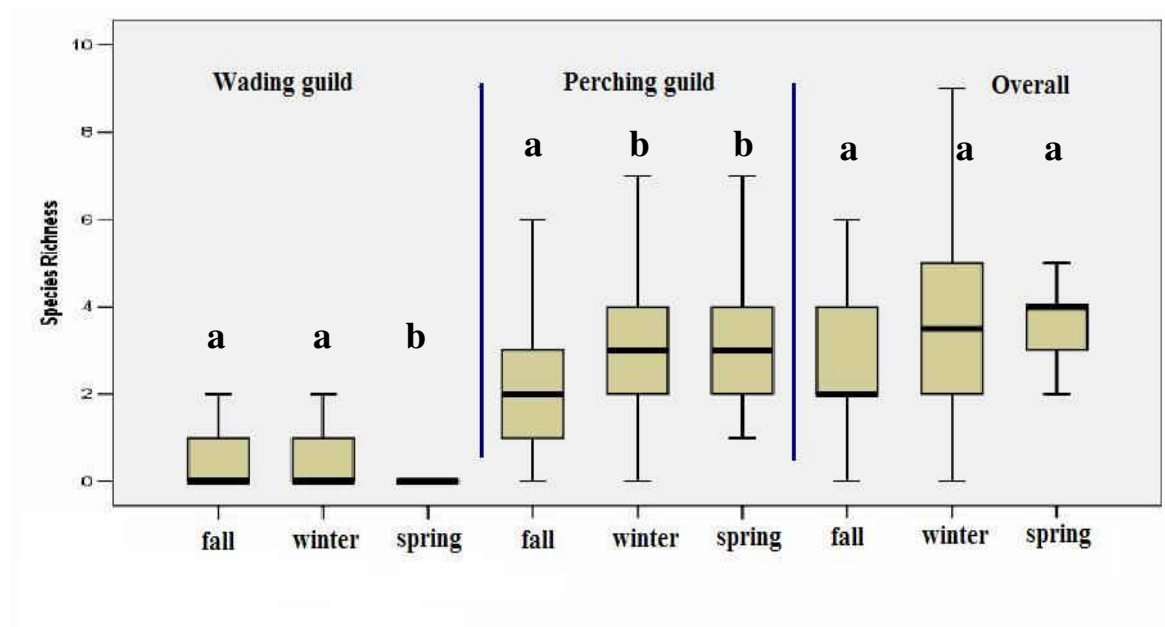


Figure 10. Boxplots comparing overall and guild species richness by season. The boxes represent quartiles, the darker line is the median, and error bars represent group extremes in maxima and minima. Within each group, differences in assigned letter designations for seasons indicate significant differences in observed guild species richness

seasonal species richness did vary significantly in some cases. Wading guild species richness was unchanged between fall and winter seasons, but both of these seasons were significantly different from species richness numbers recorded during spring. For the perching guild, fall richness numbers were significantly different from the other two seasons, while winter and spring numbers did not vary greatly. There were no significant seasonal differences in overall species richness. Swimming guild species richness data was included in analysis of overall species richness data, however, there were insufficient occurrences of swimming species within habitat patches recorded for statistical analysis to be done on swimming foraging guild data alone.

Habitat variables

Geographically isolated freshwater marshes within the conservation areas varied greatly in size, with the majority of sites having an area of less than two hectares (Figure 3). Despite this variation in size, most wetland patches have many shared characteristics. Patch edge to area ratios are similar for most sites as these isolated wetlands are often formed as near circular, gently sloping depressions in the landscape. Sites also shared most of the common ecotopes previously mentioned. Although there are similarities in vegetation and shape between wetland patches, hydrological and community scale variables including hydroperiod, water depth, percent cover, percent patch edge, and edge stem density varied greatly between the sampled wetlands (Figure 11). Vegetation coverages for Maidencane, St. John's Wort, and the beakrush species ranged from a low of 10-20% and a high of 40-75% for these ecotopes and the only discernable pattern in variations of site hydroperiod was that larger wetlands generally had greater periods of inundations for the inner most ecotopes. Finally, there was a correlation between patch area

measures and PPE and both measures of edge stem density, VSS and VSL (Table 2). Edge stem density measures ranged between 0.0004 and 0.0622 stems·m² for VSS and between 0 and 0.0043 stems·m² for VSL. Percent patch edge coverage ranged between 5% and 99% for the sampled sites. All isolation indices were correlated with patch area measures (Table 3). Isolation indices varied greatly among sites, ranging between 118 and 585 m for ISO_a, between 153 and 2931 m for ISO_b and between 250 and 747 m for ISO_c.

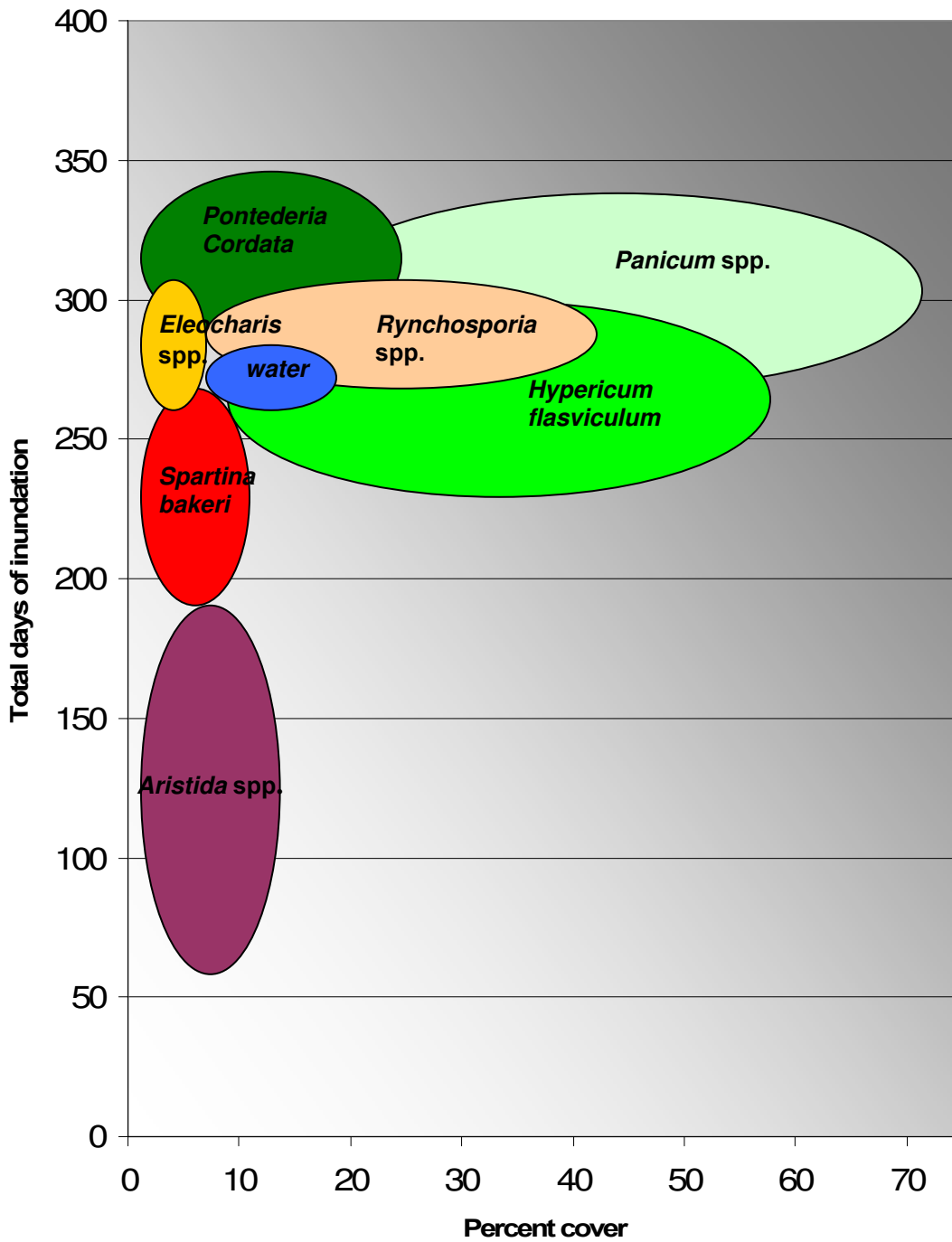


Figure 11. Hydroperiod vs. percent cover in sampled wetlands. The largest ecotopes were usually panic grasses (*Panicum spp.*) and St. John’s Wort (*Hypericum flasviculum*), with the other species or cover types forming much smaller or disjoint patches. Generally, pickerelweed (*Pontederia cordata*) and the panic grasses dominated the center of each wetland, usually having the longest hydroperiod (total number of days of inundation per year)

Table 2. Correlation of area with community scale variables

Variable	Pearson Correlation	Sig. (2-tailed)
VSS	-0.296	0.037
VSL	0.318	0.024
PPE	-0.404	0.004

Table 3. Correlation of area with isolation indices (ISO_c was not included due to multicollinearity with other measures)

Variable	Pearson Correlation	Sig. (2-tailed)
ISO _a	0.398	0.004
ISO _b	0.341	0.015

Multicollinearity

Using the PCA based approach, patch edge, percent cover of horsetail (*Eleocharis* spp.), wiregrass (*Aristida stricta*), and beakrush (*Rynchosporia* spp.), all vertical structure measures in the 20 m and 30 m buffers, and beakrush hydroperiod measures were found to be multicollinear and dropped prior to statistical analysis. Following up with the Pearson's correlation test, isolation index ISO_c and both percent cover and hydroperiod for maidencane were also found to be multicollinear and dropped.

Relationship of environmental variables to species richness

Wading guild species richness was primarily correlated with community level variables (PPE, VSL) during the fall season ($R^2 = 0.448$) (Table 4). Although additional models were produced by the multiple linear regressions, AIC values indicated that only the first two models

explained the data without to great of a loss in information content. Fall wading guild species richness was negatively correlated with both predictors. For the winter and spring seasons, wading guild species richness was positively correlated only with hydroperiod measures. In winter, wading guild species richness was positively correlated with pickerelweed hydroperiod and negatively correlated with sand cordgrass hydroperiod ($R^2 = 0.529$). During the spring season, wading guild species richness was positively correlated with pickerelweed hydroperiod ($R^2 = 0.562$). AIC values indicate all models generated for both these seasons were acceptable. Perching guild species richness was correlated with isolation and edge stem density (ISO_a , VSS) for the fall season ($R^2 = 0.388$) (Table 5). Both of these variables were positively correlated with fall perching guild species richness. Although three models were produced by the multiple linear regressions for this season, AIC values indicated only the first two models were acceptable. During the winter season, only sand cordgrass hydroperiod was found to be a predictor variable ($R^2 = 0.240$), while wetland patch area and water depth, were predictors of perching guild species richness in the spring ($R^2 = 0.415$). No one variable was found to predict perching guild species richness across seasons.

Table 4. Results from forward stepwise multiple linear regressions of seasonal wading guild species richness and independent variables. Correlation (+ or -) of individual predictors to avian species richness is indicated.

Fall season

Model	R²	F	Sig.	Predictors	AIC	ΔAIC
1	0.224	5.211	0.035	PPE(-)	-21.944	0.000
2	0.448	6.885	0.006	PPE, VSL(-)	-18.190	3.755
3	0.598	7.934	0.002	PPE, VSL, ISOa(+)	-14.606	7.338
4	0.718	9.569	0.000	PPE, VSL, ISOa, Hyd_spa(-)	-10.952	10.992
5	0.833	13.998	0.000	PPE, VSL, ISOa, Hyd_spa, ISOb(+)	-6.821	15.124

Winter season

Model	R²	F	Sig.	Predictors	AIC	ΔAIC
1	0.371	10.595	0.004	Hyd_spa(-)	-22.357	0.000
2	0.529	9.539	0.002	Hyd_spa, Hyd_pon(+)	-18.772	3.586

Spring season

Model	R²	F	Sig.	Predictors	AIC	ΔAIC
1	0.562	23.119	0	Hyd_pon(+)	-21.549	0.000

Table 5. Results from forward stepwise multiple linear regressions of seasonal perching guild species richness and independent variables. Correlation (+ or -) of individual predictors to avian species richness is indicated.

Fall season

Model	R²	F	Sig.	Predictors	AIC	ΔAIC
1	0.21	4.791	0.042	ISOa(+)	-23.153	0.000
2	0.388	5.399	0.015	ISOa, VSS(+)	-19.651	3.502
3	0.526	5.914	0.006	ISOa, VSS, PCwater(+)	-16.184	6.968

Winter season

Model	R²	F	Sig.	Predictors	AIC	ΔAIC
1	0.24	5.697	0.028	Hyd_spa(-)	-29.275	0.000

Spring season

Model	R²	F	Sig.	Predictors	AIC	ΔAIC
1	0.224	5.193	0.035	Area(+)	-19.630	0.000
2	0.415	6.027	0.011	Area, Water depth(-)	-16.365	3.265

Estimates of species richness by patch area

Overall, wading, and perching guild seasonal species richness were found to be correlated with wetland patch area (Table 6). While this correlation was not as strong as those found in the regression models, it was significant, matching the expected predator species-area relationship described by figure 1. These correlations also varied by season for guilds and overall predator species richness with the strongest correlations occurring during winter for wading guild and overall species richness and during spring for perching guild species richness. In each case, species richness was found to increase with increases in patch size (Figure 12). Given these correlations, the linear regression equation for each season of overall, wading and perching guild species richness versus patch area comparisons was also used to generate projected seasonal species richness by patch area within each of the conservation areas utilized for this study.

Table 6. Correlations of area with overall and guild seasonal species richness.

guild/season	Pearson Correlation	Sig. (2-tailed)
overall/fall	0.411	0.003
overall/winter	0.508	0.000
overall/spring	0.493	0.000
wading/fall	0.433	0.002
wading/winter	0.496	0.000
wading/spring	0.343	0.015
perching/fall	0.359	0.010
perching/winter	0.418	0.002
perching/spring	0.445	0.001

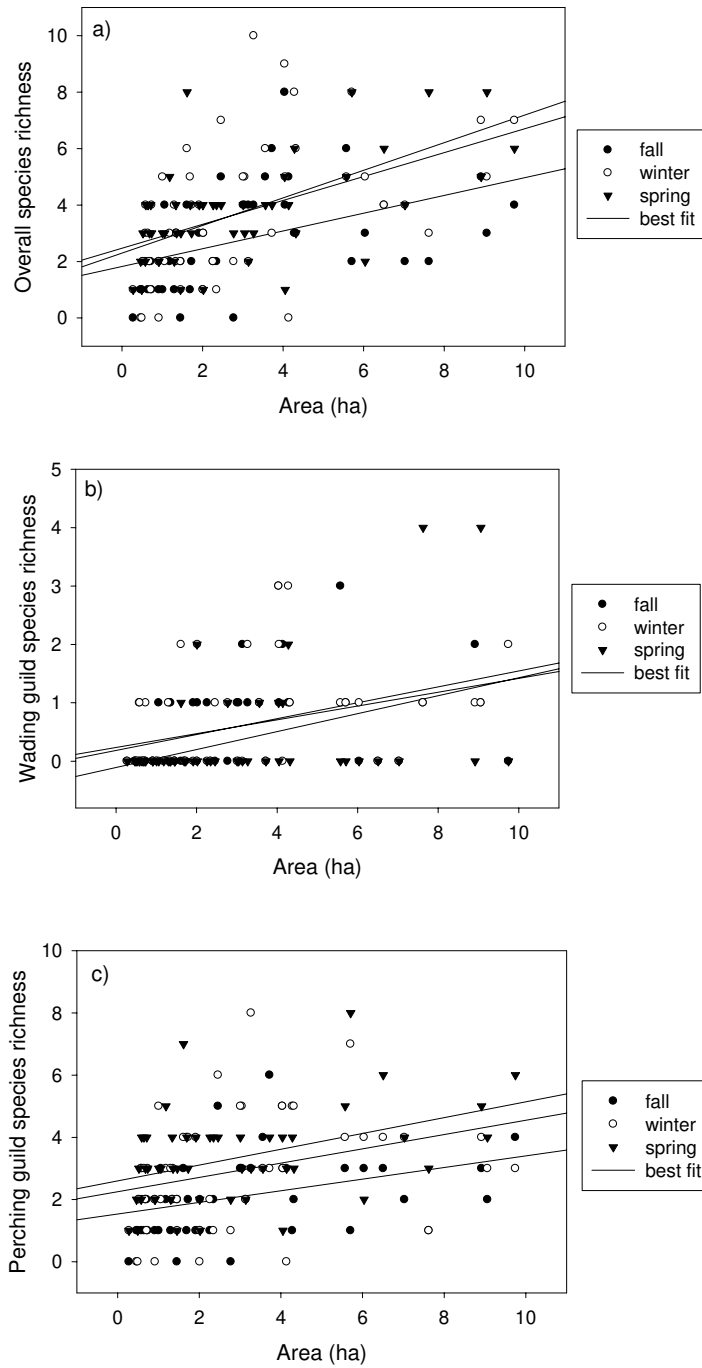


Figure 12. Correlations of overall(a), wading guild(b) and perching guild(c) species richness with patch area.

The expected overall species richness estimated by patch area varied greatly by season. The winter season had the highest expected species richness at 11 and the maximum for expected species richness never fell below six for all seasons (Figure 13). The winter season also had the highest number of patches expected to have zero species present. For the wading guild, results were similar for the fall and winter seasons and projected richness never exceeded one annually, with the majority of sites projected to have no wading species present. However, expected number of patches with wading species present declined greatly for the spring season in comparison with other seasons. Finally, for the perching guild, the greatest expected species richness was during the spring season and the lowest species richness is expected during the fall season. While estimated overall and guild species richness numbers did fluctuate by season, these differences were not found to be significant within any of the guilds.

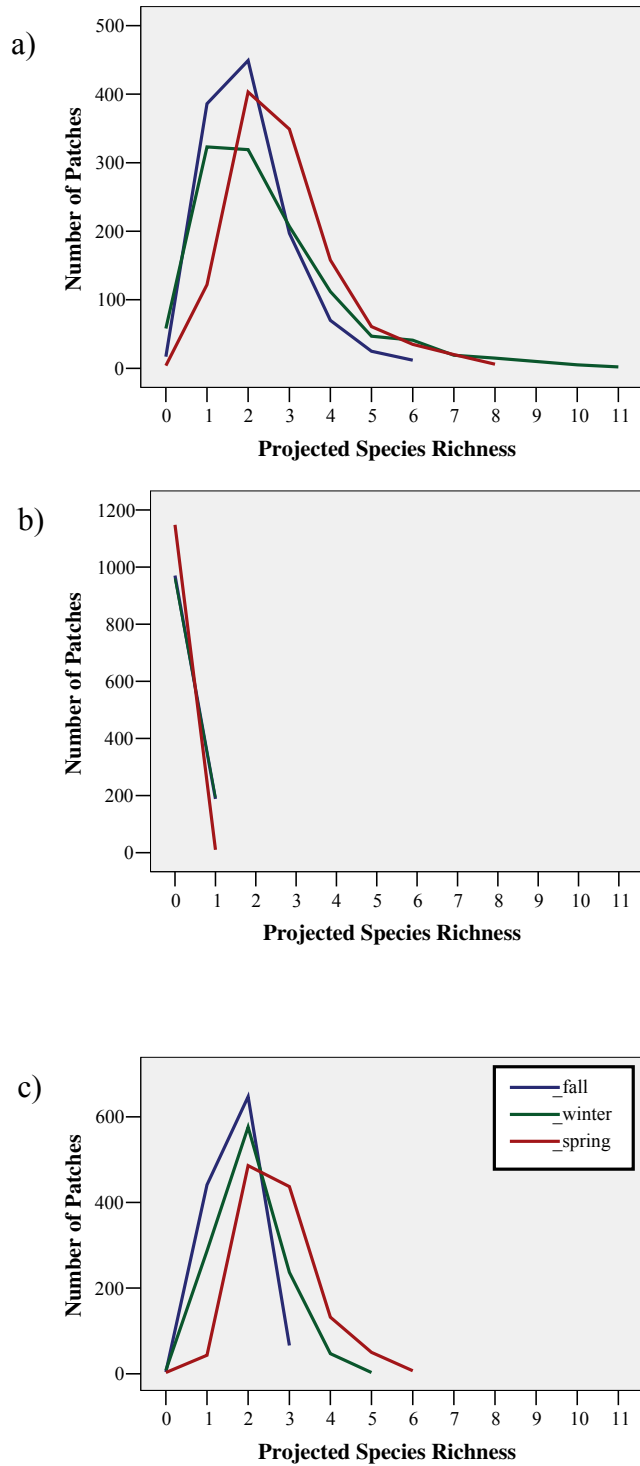


Figure 13. Comparisons of seasonal differences for projected a) overall, b) wading and c) perching guild species richness by number of geographically isolated freshwater marshes

Relationship of environmental variables to species presence

Of the 45 species detected, only 14 were found to occur in sufficient enough numbers to permit forward stepwise logistic regression. Thirteen of these species were detected for all three seasons and the final species, Eastern Towhee, was only present during two seasons. The forward stepwise logistic regression was used to ferret out the strongest correlations between landscape, community and hydrological variables with individual species presence/absence data by season. The results of this analysis were used as a prediction profiler to generate logistic plots of the seasonal probability of occurrence for a species. Ultimately, only five of the remaining 14 species were found to have significant correlations with one or more variables. Presence/absence of Red-winged Blackbirds (*Agelaius phoeniceus*) was found to be correlated with wetland water depth (> 51cm) during the fall season and St. John's Wort hydroperiod (> 248 days) in the spring at a 50% or greater chance of occurrence (Figure 14). Eastern Meadowlarks (*Sturnella magna*) were correlated to ISO_a during fall and spring season, > 254m and > 169m respectively and area (< 0.89ha) in winter (Figure 15). St. John's Wort hydroperiod (< 267 days) and St. John's Wort percent cover (< 26%) were correlated with Eastern Towhee (*Pipilo erythrophthalmus*) presence/absence in spring and sand cordgrass hydroperiod (< 246 days) in winter (Figure 16). Great Egret (*Ardea alba*) presence/absence was correlated with pickerelweed hydroperiod (> 365 days) during winter and spring seasons (Figure 17). Finally, Great Blue Heron (*Ardea herodias*) presence/absence was correlated with ISO_b during winter and fall, > 3150m and > 1778m respectively, and pickerelweed hydroperiod (> 365) during spring (Figure 18).

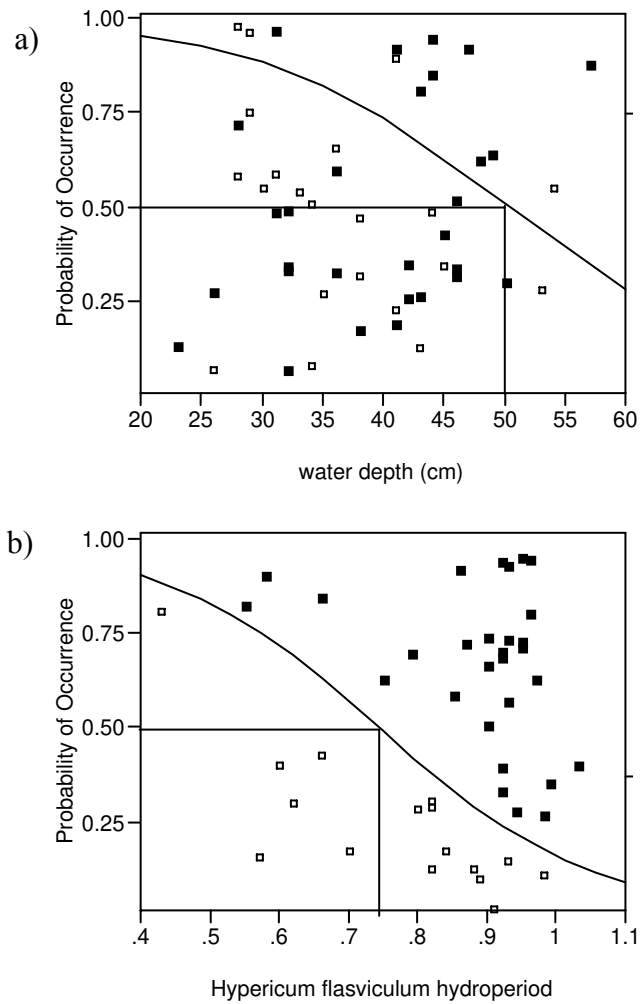


Figure 14. Logistic plots of probability of Red-winged Blackbird occurrence as a function of a) water depth in fall and b) St. John's Wort (*Hypericum flasviculum*) hydroperiod in spring. Hydroperiod measures are arcsine transformed from total number of days of inundation. Solid squares indicate species presence, hollow squares indicate species absence.

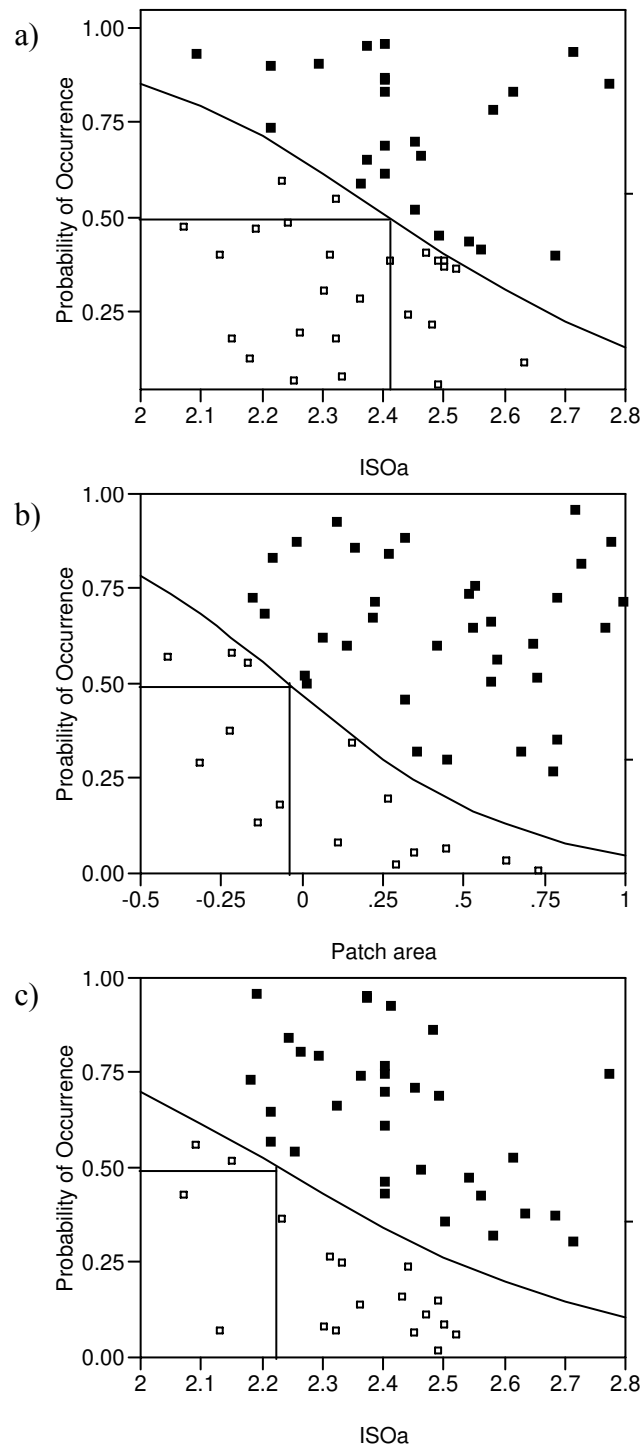


Figure 15. Logistic plots of probability of Eastern Meadowlark occurrence as a function of a) isolation in fall and c) spring and b) area in winter. Isolation and patch area measures are log transformed from distance in meters and hectares, respectively. Solid squares indicate species presence, hollow squares indicate species absence.

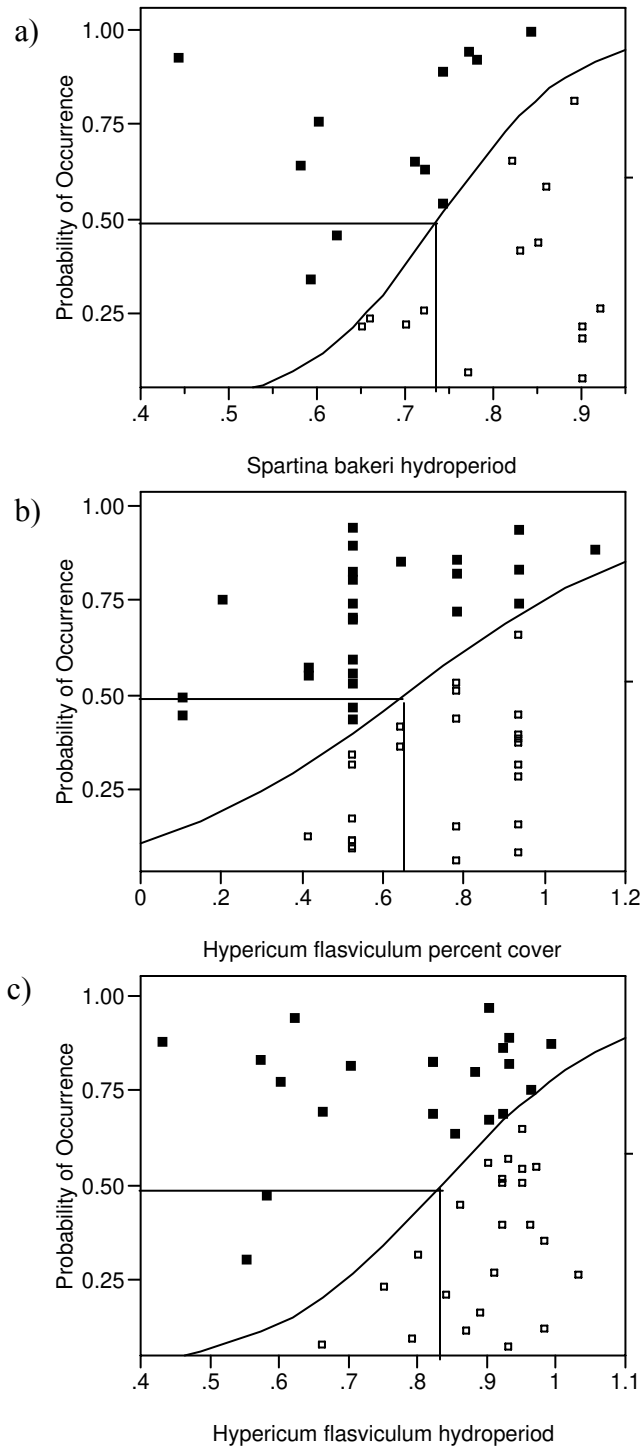


Figure 16. Logistic plots of probability of Eastern Towhee occurrence as a function of a) sand cordgrass (*Spartina bakerii*) hydroperiod in winter and b) St. John's Wort (*Hypericum flasviculum*) percent cover and c) hydroperiod in spring. Hydroperiod and percent cover measures are arcsine transformed from total number of days of inundation and percent cover, respectively. Solid squares indicate species presence, hollow squares indicate species absence.

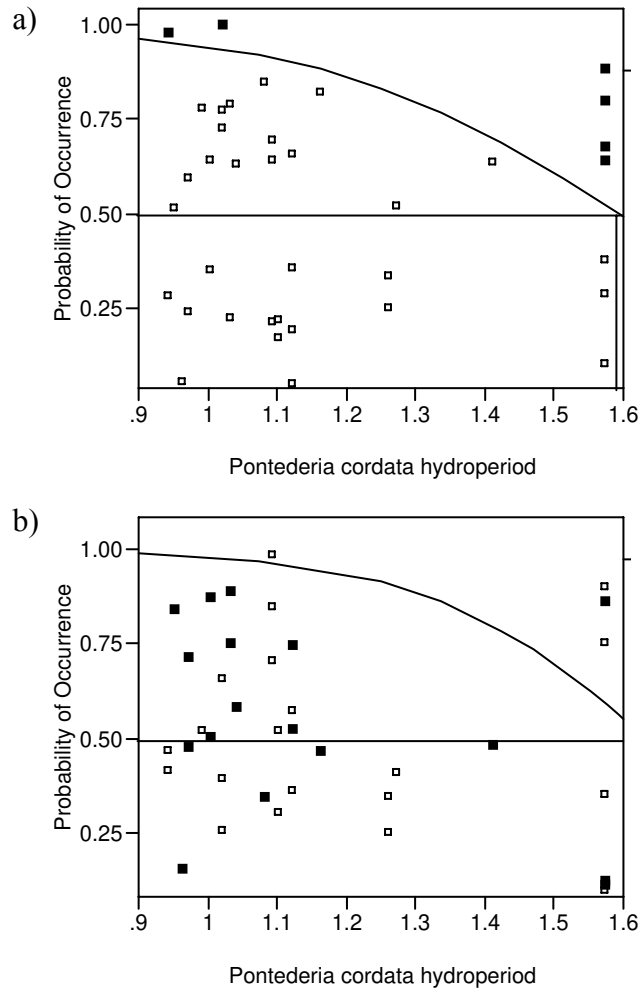


Figure 17. Logistic plots of probability of Great Egret occurrence as a function of hydroperiod in a) winter and b) spring. Hydroperiod measures are arcsine transformed from total number of days of inundation. Solid squares indicate species presence, hollow squares indicate species absence.

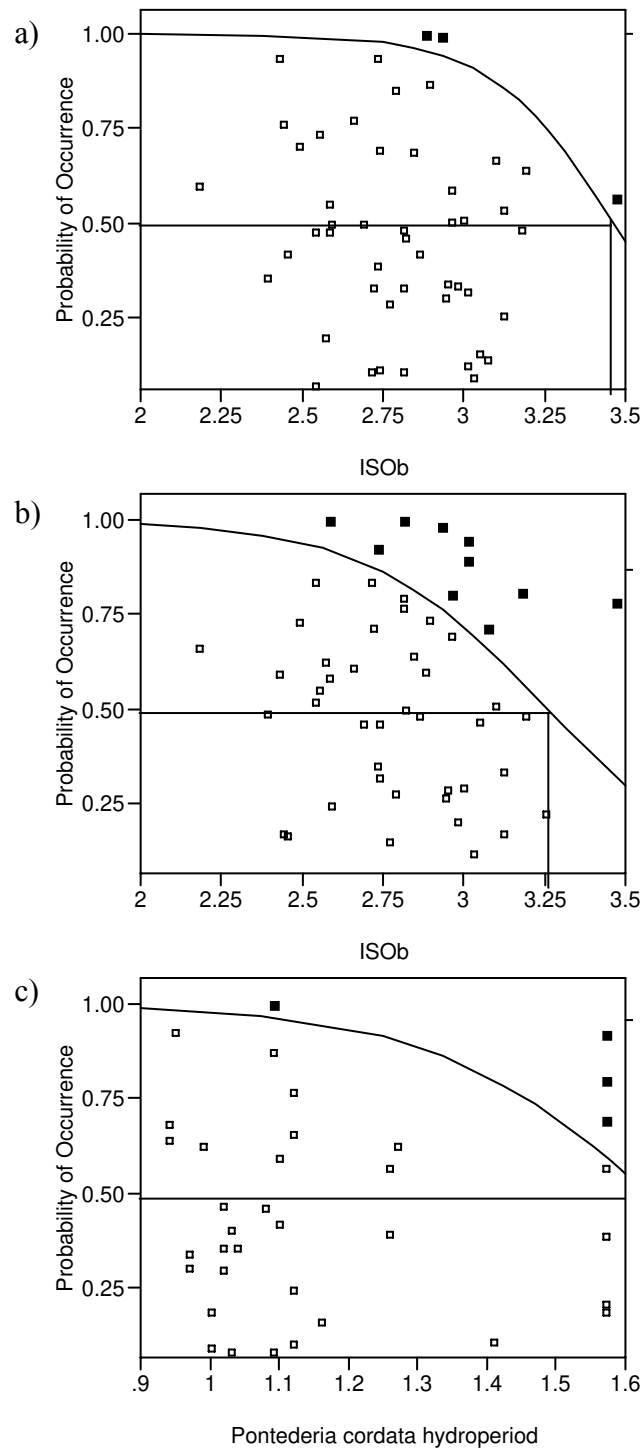


Figure 18. Logistic plots of probability of Great Blue Heron occurrence as a function of isolation in a) fall and b) winter and c) hydroperiod during spring. Isolation measures are log transformed and hydroperiod measures are arcsine transformed from distance in meters and total number of days of inundation, respectively. Solid squares indicate species presence, hollow squares indicate species absence.

CHAPTER FOUR: DISCUSSION

Foraging guilds and habitat use

Within the study area, avian species richness in geographically isolated wetlands was correlated with multiple variables. Hydroperiod, patch area, water depth, vertical structure and isolation all factored into species richness throughout the year. Predictors from multiple scales were shown to interact to determine guild species richness and habitat selection (Naugle et al 1999, Custer et al. 2004, Trocki & Paton 2006). These predictors can and usually do change with each season and each foraging guild. These variations are implicitly linked to foraging preferences associated with the presence/absence of standing water within the patch habitat.

During the fall season, water levels were near maximum for most wetland patches, likely negating the importance of hydroperiod, water depth, and to a lesser extent area (as an indicator of presence of water) as predictors since they are uniform for all patches. For perching guild species, this means that isolation and vegetation structure variables act as predictors observed in other studies (Ozesmi & Mitsch 1997, Mortberg 2001, Pearman 2002). Perching species focus on minimal travel distance between patches (ISO_a) and availability of small stem vertical structure (VSS) within the wetland. As water levels plateau and begin to drop during winter and spring seasons, the predictors for perching guild species richness change. In winter, the focus becomes hydroperiod for the sand cordgrass ecotope. This ecotope is a peripheral zone and during this season is usually drying out or covered by only a shallow inundation, likely a prime zone for foraging for adult and larval stage amphibians, insects, and macroinvertebrates. As the water levels draw farther down through spring, foraging for prey associated within the wetland patches becomes more difficult. Only the larger patches still have water, so area and water depth become the best predictors of perching guild species richness.

A similar story plays out for wading foraging guild species. During the fall season, two vegetation structure variables act as predictors. These factors (PPE, VSL) are likely perceived as landscape variables, acting as vertical signposts embedded within the otherwise flat, ubiquitous landscape of the dry prairie, indicating where wetland habitat can be found from the aerial perspective. In the following seasons, as water levels draw down, these other variables fall away as predictors and wading species rely heavily on hydroperiod. During winter, both sand cordgrass and pickerelweed hydroperiod are predictors, indicating the use of peripheral and the more central areas of wetland patches as foraging sites. Finally, as most the water dries out, the pickerelweed hydroperiod becomes the sole predictor in spring. Hydroperiod has been found to be a strong predictor for many prey species in other studies (Weyrauch & Grubb 2004, Tavernini et al. 2005, Battaglia & Collins 2006). Waders seek out wetlands with a longer hydroperiod and forage in the peripheral shallows and mudflats on the exposed macroinvertebrates, amphibians, and small fish, changing foraging position as the wetland shoreline recedes or expands.

Species level habitat factors

At a finer scale, i.e. the individual species level, predictors of species occurrence varied greatly by species and by season. Water depth acted as a predictor of species occurrence for Red-winged blackbirds during the fall season while St. John's Wort hydroperiod was a predictor during spring. Though both of these predictors represent different variables, they are both types of measures of water presence within the wetland patch. This commonality likely indicates the importance of water for use of isolated marshes for foraging and breeding. In other studies of Red-winged blackbirds, similar results have been found (Ozesmi & Mitsch 1997, Lariviere and Lepage 2000).

Landscape variables proved the best indicator for probability of species occurrence for Eastern Meadowlarks, with ISO_a acting as a predictor in fall and spring and patch area a predictor during winter. This species showed a preference for patches that were located close to other freshwater habitat patches and had a smaller patch area. While this species is generally considered a grassland bird, it was observed often within wetland patches during counts, with less than half of all sightings associated with the wetland edge habitat, which could be construed as upland habitat by some researchers. Little previous work on Eastern Meadowlarks within wetlands can be found in the literature and but patch area relations established by previous studies of grassland habitat match those found in this study, meadowlarks use smaller wetlands over larger patches (Helzer & Jelinski 1999, Horn et al. 2000).

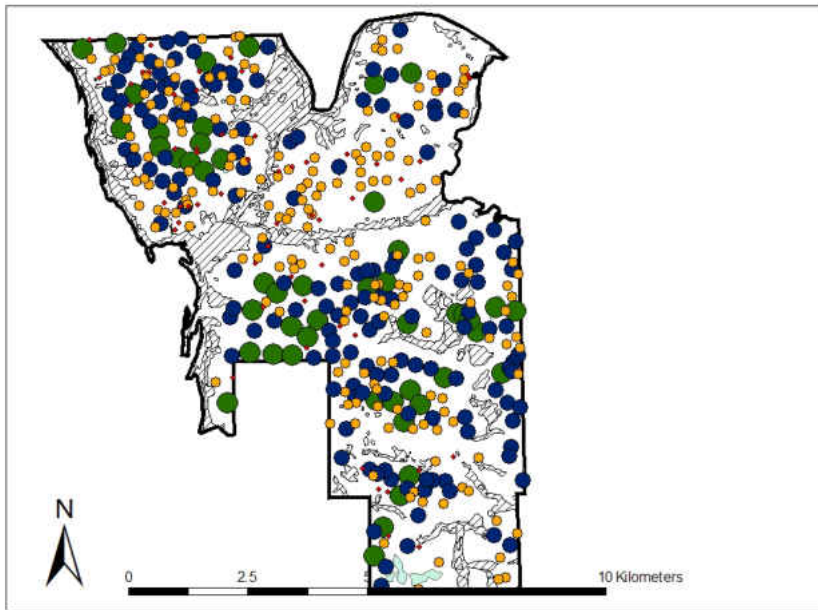
A third perching species, the Eastern Towhee, was found in sufficient enough numbers to generate probability of occurrence models for two seasons. The Towhee was only present during winter and spring, and hydroperiod was found to be a predictor during both seasons. Since the Eastern Towhee is generally considered a forest dwelling and breeding species in previous studies (Twedt et al. 2000, Lohr et al. 2002), little published data are available concerning its habitat use during the non-breeding seasons or its activities in other habitat types. It has also been classified as an opportunistic gap species by some studies (Bell & Whitmore 2000, Greenberg & Lanham 2001, Fink et al. 2006), explaining its use of “edge” habitat, but its use of wetlands and the expansive non-forested dry prairie speaks to the need for more research on this and other avian species during non-breeding periods. Its presence in this study was likely a result of the prey availability within isolated wetlands provided for all avian species present in the dry prairie ecosystem.

Finally, two wading bird species, the Great Egret and Great Blue Heron, were found to have higher probabilities of occurrence based on the longer pickerelweed hydroperiod and the latter also showed a positive correlation to isolated wetlands that were closer to larger, continuous wetlands.

Patch area correlations

At a broader scale, overall avian and guild species richness were correlated with patch area. As expected for generalist predators, it was not a strong correlation, but it was significant, making patch area an easily assessed (through the use of high resolution aerials and published GIS maps) landscape variable that can be used in conjunction with predator species richness to assess metacommunity connectivity as well as to generate projected seasonal patch species richness with applicability toward widespread use as a conservation planning tool. In addition to confirming metacommunity connectivity, average annual patch overall and guild species richness were estimated and maps were generated to show projected clustering patterns that occurred within the landscape (Figures 19-21). Additionally, detailed maps of projected patch species richness for each season and study area by overall or guild species richness can be found in the Appendix C. Ideally, this mapping technique could be used to model the location of diversity hotspots in cases where only a select portion of a natural area can be retained for a proposed reserve.

a)



b)

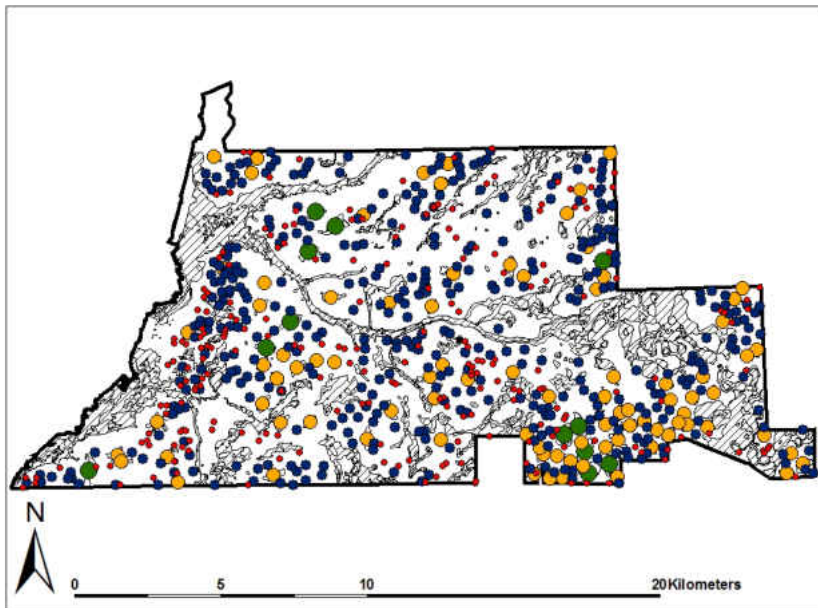
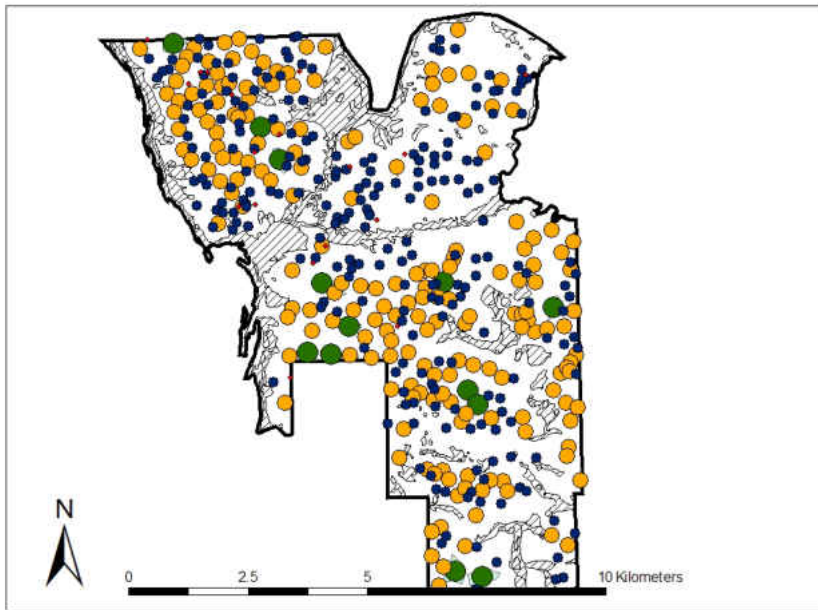


Figure 19. Projected average annual overall species richness clusters for a) TLWMA and b) KPPSP study areas. Red circles indicate an average species richness of ≤ 1 , blue 1-2, orange 2-3, green ≥ 3

a)



b)

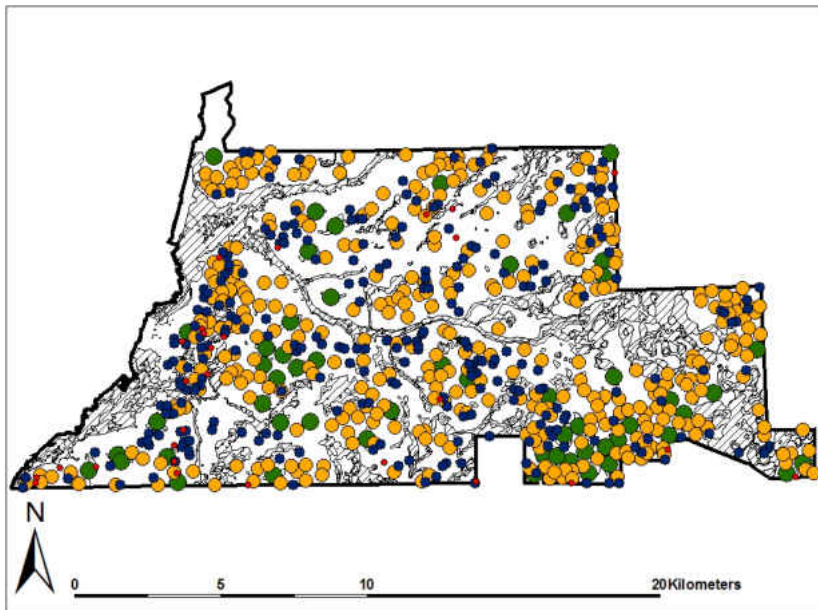
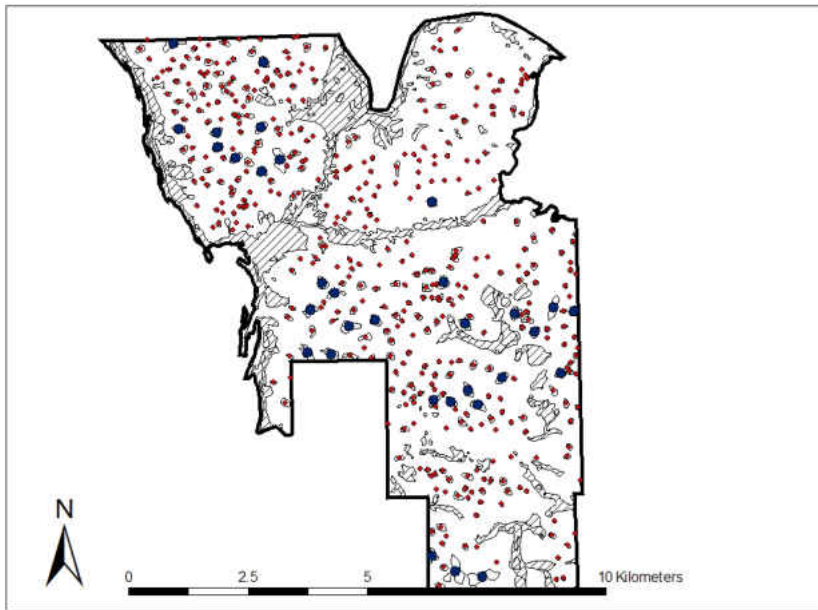


Figure 20. Projected average annual perching guild species richness clusters for a) TLWMA and b) KPPSP study areas. Red circles indicate an average species richness of ≤ 1 , blue 1-2, orange 2-3, green ≥ 3 .

a)



b)

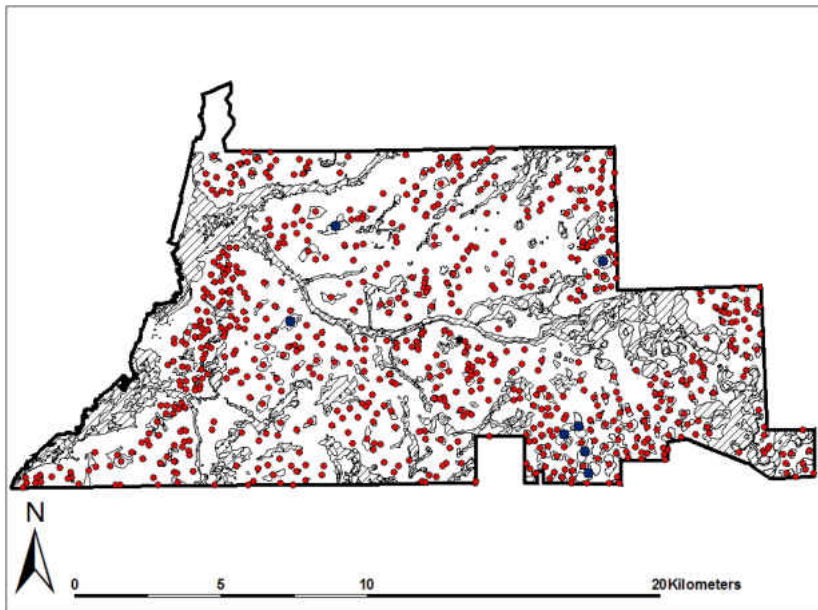


Figure 21. Projected average annual wading guild species richness clusters for a) TLWMA and b) KPPSP study areas. Red circles indicate an average species richness of ≤ 1 , blue 1-2, orange 2-3, green ≥ 3

CHAPTER FIVE: CONCLUSIONS

The metacommunity approach toward analyzing and understanding how populations persist in patchy landscapes is still in development. However, the potential applications of this approach are made clear by the results of this study. By assessing predator species richness in a naturally patchy landscape free of significant anthropogenic effects, i.e. habitat loss, this study shows that the resulting generalist predator species-area relationship matches the expected regression fit line in figure 1, confirming metacommunity connectivity. Its application toward systems subject to habitat loss is the next logical step. Habitat loss is generating greatly fragmented systems and these newly formed patch networks respond by organizing into metacommunities that have issues of connectivity associated with them that change with trophic level. The effects of isolation on a habitat patch vary greatly between birds, mammals, amphibians, and aquatic microorganisms as each perceives the landscape and dispersal barriers differently. The effect of this perception on the resulting metapopulations is not clear, but intuitively if a highly mobile predator species is impacted negatively by habitat loss and fragmentation, it seems likely that less mobile or sessile organisms could be impacted even more negatively. At the same time, the less mobile groups tend to comprise the lower trophic levels, suggesting that any negative fragmentation effects could actually cascade up the food web, influencing the distribution or existence of top predators within the system. This approach to testing for metacommunity connectivity would detect the impacts of habitat loss on a metacommunity.

Additionally, sound conservation strategies require accurate information about the driving forces behind habitat use by local populations and communities. Without the most accurate information, management efforts are haphazard and may even be detrimental. This

becomes clear with the wetland systems studied here. While the species-area relationship works well in describing most upland systems, it should be carefully used in wetland systems. Unlike their upland counterparts, these systems tend to be ephemeral, expressing temporally dramatic local community shifts with changes in hydrology. While area does correlate with predator species richness in this study, it does so while water is present. The presence of water drives the food web in these systems. Without it, aquatic microorganisms disappear, along with much of the macroinvertebrates and amphibians which are prey for the higher trophic predator species, i.e. birds. This relationship became apparent during the final season of this study as dried wetlands were virtually abandoned in favor of the few remaining marshes still inundated. Water is the driving force behind habitat use in geographically isolated freshwater marshes. Conservation policies should carefully consider measures to preserve hydrologic state and variability in maintaining isolated wetland systems in particular.

Finally, future studies should expand beyond the species-area relationship and the limited metacommunity application explored by this study. Metacommunity studies are the next evolution of ecological study toward better understanding the planet and how we can be better stewards for its preservation. The metacommunity concept should be fully explored. Until then broad scale measures like predator species richness and patch area will have to remain the mainstay for the cautious formation of conservation policy into the near future.

APPENDIX A: DATA

Species list, foraging guild classification, breeding status and associated prey

Species	Foraging Guild	Breeding Status	Feeding Classification	Prey
AMBI	W	Y*	Carnivore	fish,insects,frogs,crabs, other invertebrates
AMCR	P	Y	Omnivore	insects, rodents, snakes, lizards, young birds, seeds and fruit
AMRO	P	Y*	Omnivore	insects and fruits
BAEA	P	N	Carnivore	fish, birds, carrion
BEKI	P	N	Carnivore	fish, crayfish, frogs, and insects
BLVU	P	N	Carnivore	carrion
BTGR	P	Y	Omnivore	beetles, crayfish, crabs, insects and grain
BWTE	S	Y*	Omnivore	mostly seeds and vegetation, insects
CAWR	P	Y	Carnivore	insects
CGDO	P	Y	Herbivore	grains and seeds
CONI	P	N	Carnivore	insects, mostly flying insects
COYE	P	Y	Carnivore	aquatic and terrestrial insects
CRCA	P	N	Carnivore	reptiles, birds, mammals,and carrion
EAME	P	Y	Omnivore	mostly insects, some seeds and grains
EAPH	P	N	Omnivore	insects, also eats fruits
EATO	P	Y	Omnivore	insects, earthworms, spiders, seeds and berries
GBHE	W	N	Carnivore	fish, eggs, insects, frogs, snakes, crabs, shrimp, small birds, small mammals
GRCA	P	Y	Omnivore	mostly fruits, some insects and other invertebrate
GREG	W	N	Carnivore	fish, frogs, snakes, small invertebrates, small birds, small mammals
HOME	S	N	Carnivore	small fish, crustaceans, aquatic insects
KILL	W	Y	Carnivore	insects and other invertebrates
KIRA	W	Y	Omnivore	aquatic insects, seed, and other vegetation
LBHE	W	N	Carnivore	fish, frogs, small invertebrates
LEBI	W	Y*	Carnivore	fish, insects, frogs, crabs, other invertebrates
LESC	S	N	Carnivore	snails, crabs, shrimp, insects
LOSH	P	Y*	Carnivore	insects, grasshoppers, caterpillars, rodents, lizards, and small birds
MAWR	P	Y*	Carnivore	insects and spiders
MODO	P	Y	Herbivore	grains and seeds
MODU	S	N	Omnivore	mostly seeds and vegetation, insects, snails
NOBO	P	N	Omnivore	grains, grasses, seeds, and insects
NOCA	P	Y	Omnivore	seeds and fruits, also insects
NOMO	P	Y	Omnivore	mostly fruits, some insects and other invertebrate
OSPR	P	Y	Carnivore	fish
PUGA	W	Y*	Omnivore	frogs, grasshoppers, spiders, other invertebrates, and aquatic vegetation
RBWO	P	N	Omnivore	seeds, nuts, and insects
RSHA	P	N	Carnivore	small mammals, lizards, snakes, frogs, crayfish, and insects
RWBL	P	Y	Omnivore	seeds, grasshoppers, dragonflies, and other insects
SACR	W	Y	Omnivore	plants, seeds, and invertebrates
STHA	P	N	Carnivore	birds, particularly red-winged blackbirds and eastern meadowlarks
SWSP	P	N	Omnivore	seeds and insects
TRSW	P	N	Omnivore	insects and wax myrtle fruits
TUVU	P	N	Carnivore	carrion
WHIB	W	N	Carnivore	aquatic insects, grasshoppers, crabs, crayfish, small snakes, invertebrates
WISN	W	N	Carnivore	worms, insects, and other invertebrates
WOST	W	N	Carnivore	fish, frogs, snakes, aquatic worms, crabs, crayfish, other invertebrates

* denotes species known to breed in habitat and present, but not counted during breeding season

P denotes species is classified as belonging to the perching foraging guild

W denotes species is classified as belonging to the wading foraging guild

S denotes species is classified as belonging to the swimming foraging guild

Sources: Sibley, D.A. 2003; National Geographic 2003; Maher, D.S. & H.W. Kale II 2005

Locations and size of study habitat patches at Three Lakes Wildlife Management Area

SITE	Latitude	Longitude	Area (ha)
11	27° 52' 05" N	81° 10' 10" W	0.74
12	27° 51' 58" N	81° 09' 44" W	0.58
15	27° 50' 34" N	81° 09' 23" W	0.5
16	27° 49' 25" N	81° 08' 17" W	0.71
17	27° 52' 40" N	81° 10' 15" W	0.52
19	27° 54' 23" N	81° 09' 00" W	0.68
21	27° 53' 32" N	81° 11' 00" W	1.06
23	27° 52' 42" N	81° 10' 22" W	1.19
24	27° 51' 17" N	81° 09' 18" W	1.36
26	27° 53' 55" N	81° 08' 55" W	1.34
27	27° 53' 44" N	81° 08' 32" W	1.01
28	27° 51' 50" N	81° 09' 35" W	1.3
30	27° 51' 22" N	81° 09' 35" W	1.69
31	27° 50' 44" N	81° 08' 51" W	2.02
33	27° 51' 23" N	81° 09' 50" W	1.73
34	27° 49' 30" N	81° 00' 31" W	1.92
35	27° 51' 44" N	81° 10' 14" W	2.46
36	27° 49' 33" N	81° 08' 55" W	2.26
37	27° 53' 44" N	81° 09' 12" W	1.62
38	27° 54' 06" N	81° 08' 48" W	2.01
41	27° 50' 34" N	81° 08' 30" W	3.04
45	27° 51' 44" N	81° 10' 42" W	4.28
46	27° 51' 07" N	81° 08' 03" W	3.27
49	27° 52' 36" N	81° 09' 22" W	3.01
54	27° 51' 12" N	81° 07' 52" W	5.58
55	27° 50' 35" N	81° 08' 59" W	4.32
57	27° 51' 44" N	81° 09' 14" W	5.71
59	27° 53' 15" N	81° 11' 30" W	9.06

Locations and size of study habitat patches at Kissimmee Prairie Preserve State Park

SITE	Latitude	Longitude	Area (ha)
10	27° 34' 16" N	81° 02' 16" W	0.46
13	27° 35' 10" N	81° 05' 38" W	0.59
14	27° 34' 53" N	81° 04' 39" W	0.65
18	27° 35' 01" N	81° 07' 53" W	0.28
20	27° 35' 20" N	81° 05' 50" W	0.92
22	27° 35' 28" N	81° 07' 06" W	1.46
25	27° 35' 10" N	81° 06' 52" W	0.92
29	27° 34' 05" N	80° 58' 43" W	1.45
32	27° 33' 40" N	80° 58' 30" W	3.14
39	27° 35' 26" N	81° 07' 50" W	2.35
40	27° 32' 48" N	80° 59' 24" W	4.04
42	27° 35' 10" N	81° 08' 07" W	2.78
43	27° 33' 46" N	81° 01' 13" W	3.73
44	27° 33' 36" N	80° 58' 50" N	4.05
47	27° 36' 10" N	81° 06' 40" W	4.14
48	27° 33' 54" N	80° 58' 29" W	3.56
50	27° 33' 40" N	80° 59' 10" W	7.62
51	27° 35' 45" N	81° 06' 50" W	7.03
52	27° 36' 12" N	81° 02' 46" W	8.92
53	27° 32' 55" N	81° 01' 02" W	6.04
56	27° 34' 26" N	81° 01' 28" W	6.51
58	27° 34' 32" N	81° 02' 54" W	9.75

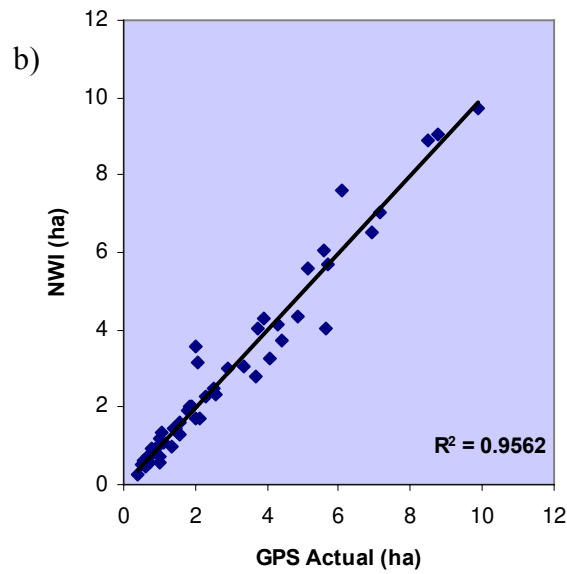
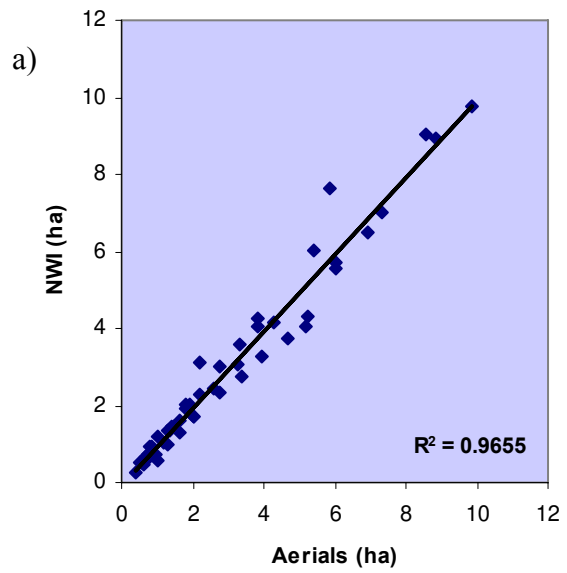
Number of sites species were detected at by season

Species	fall	winter	spring
AMBI	2	1	0
AMCR	5	1	3
AMRO	1	0	0
BAEA	0	1	0
BEKI	1	0	0
BLVU	5	4	3
BTGR	0	0	1
CAWR	0	0	2
CGDO	2	0	3
CONI	0	0	1
COYE	30	40	36
CRCA	1	0	2
EAME	24	35	32
EAPH	4	3	0
EATO	0	20	25
GBHE	3	10	4
GRCA	4	6	1
GREG	10	10	4
HOME	0	4	0
KILL	1	1	0
LBHE	2	0	1
LEBI	2	0	0
LESC	1	2	0
LOSH	1	0	0
MAWR	1	0	0
MODO	2	0	5
MODU	3	5	1
NOBO	0	0	3
NOCA	0	1	2
NOMO	6	6	3
OSPR	0	1	1
RBWO	0	0	2
RSHA	4	7	7
RWBL	13	13	33
SACR	2	3	5
STHA	0	1	0
SWSP	0	1	0
TRSW	0	6	0
TUVU	0	0	2
WHIB	1	0	1
WISN	6	4	1
WOST	0	0	1

Percent cover measures and associated scales. Braun-Blanquet scale included for comparison purposes only

Percent cover measured (%)	Domin-Krajina Scale	Braun-Blanquet Scale
0 to 0.2	1	1
0.2 to 1	2	
1 to 5	3	
5 to 10	4	2
10 to 25	5	
25 to 33	6	3
33 to 50	7	
50 to 75	8	4
75 to 99	9	5
100	10	

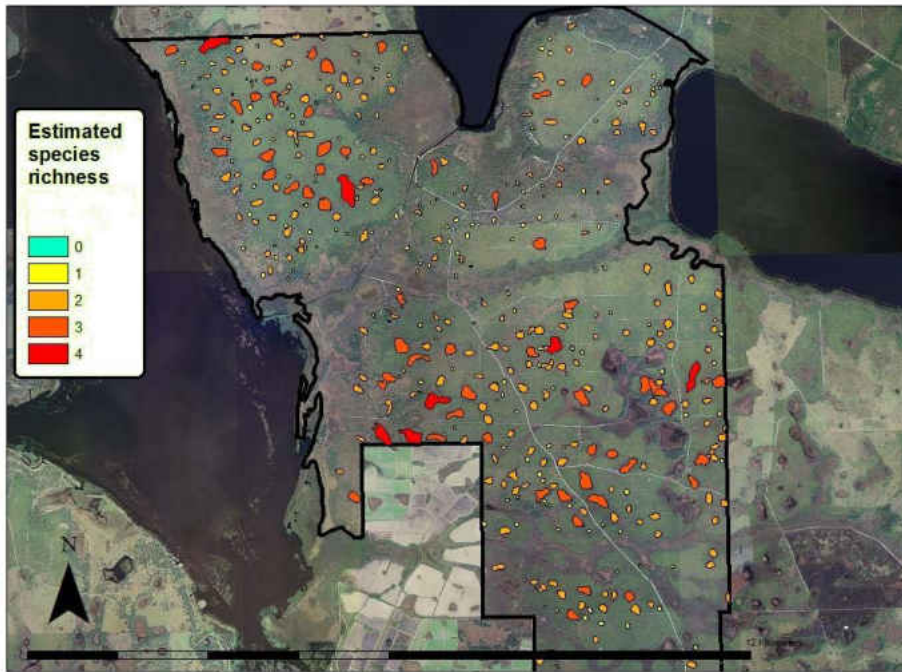
APPENDIX B: NWI MAP VERIFICATION



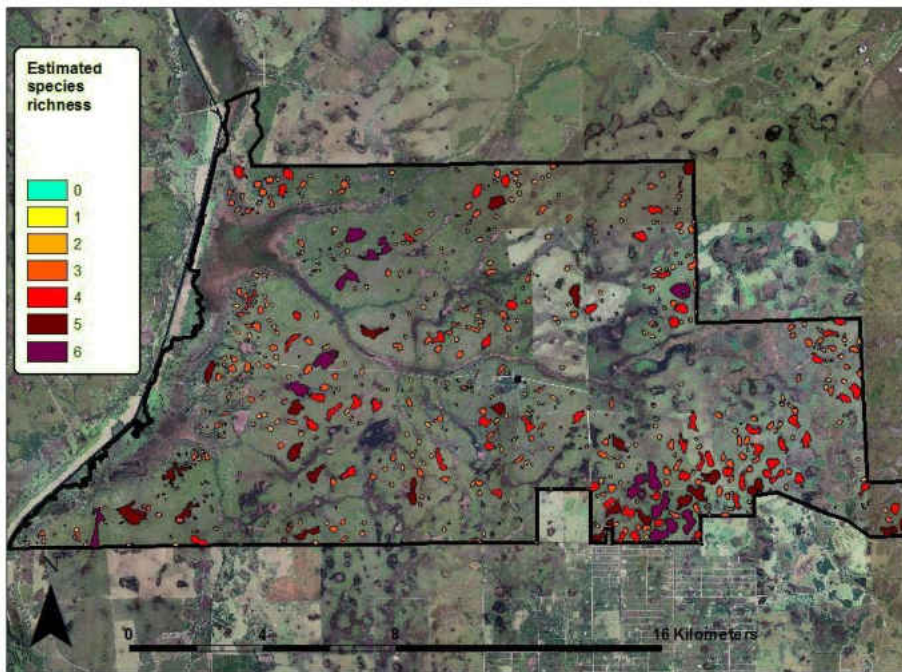
Comparisons of NWI area data with other measures based on a.)2005 high resolution aerials and b.)ground GPS mapping of each site.

APPENDIX C: SPECIES RICHNESS MAPS

a)

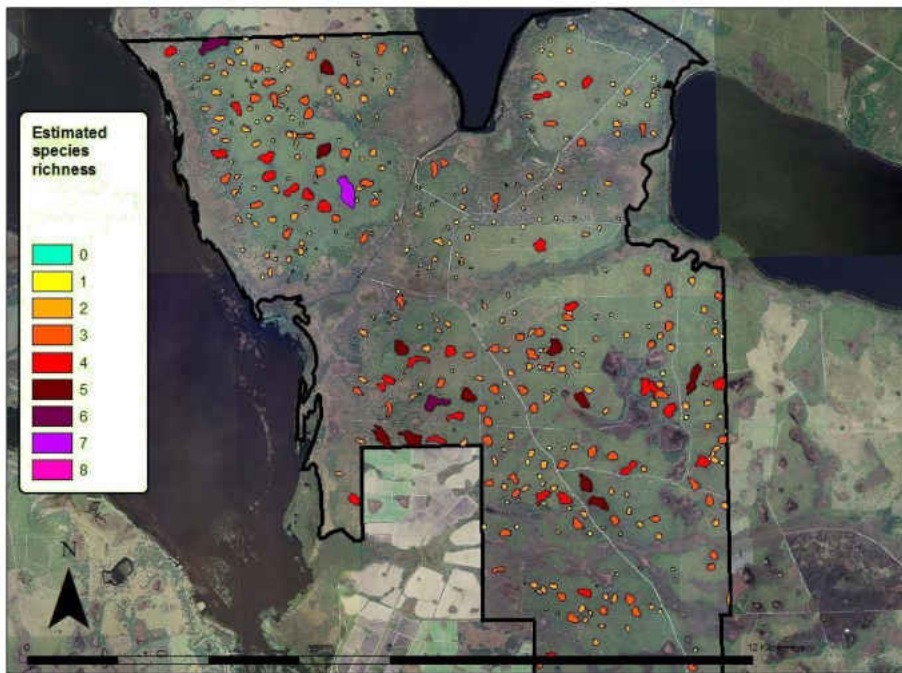


b)

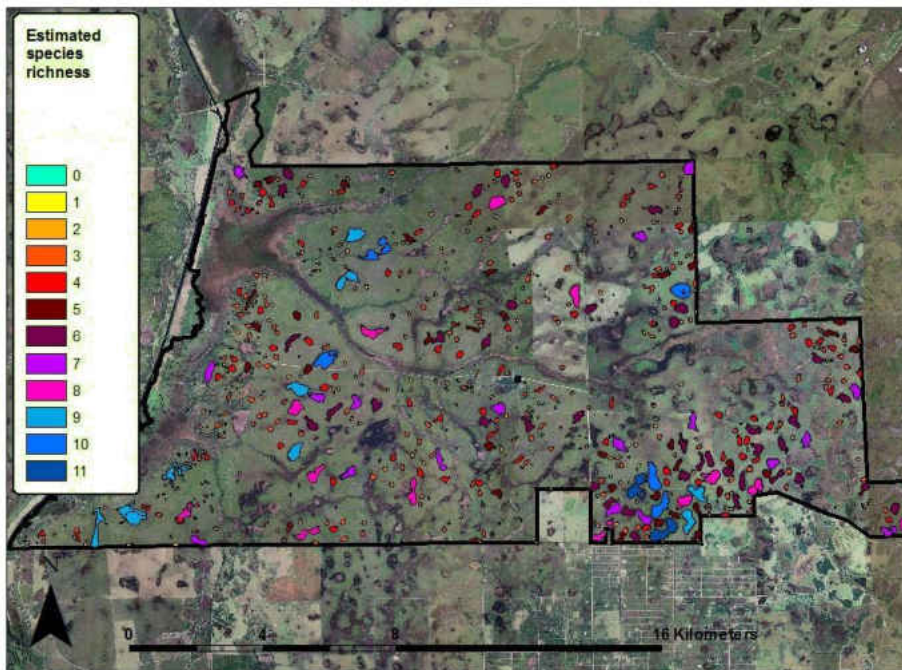


Fall season overall patch species richness estimates for a) TLWMA and b) KPPSP

a)



b)

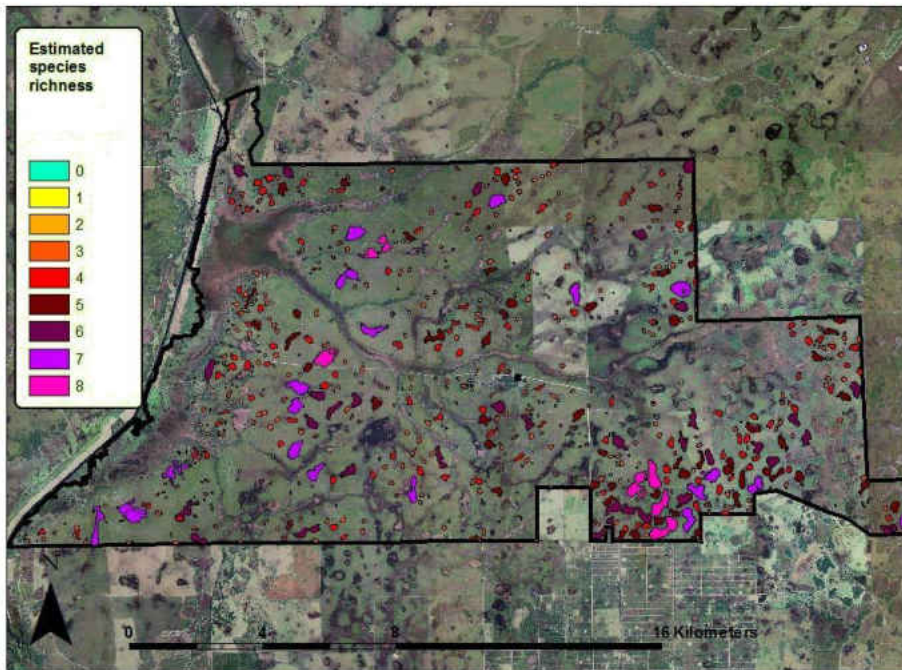


Winter season overall patch species richness estimates for a) TLWMA and b) KPPSP

a)

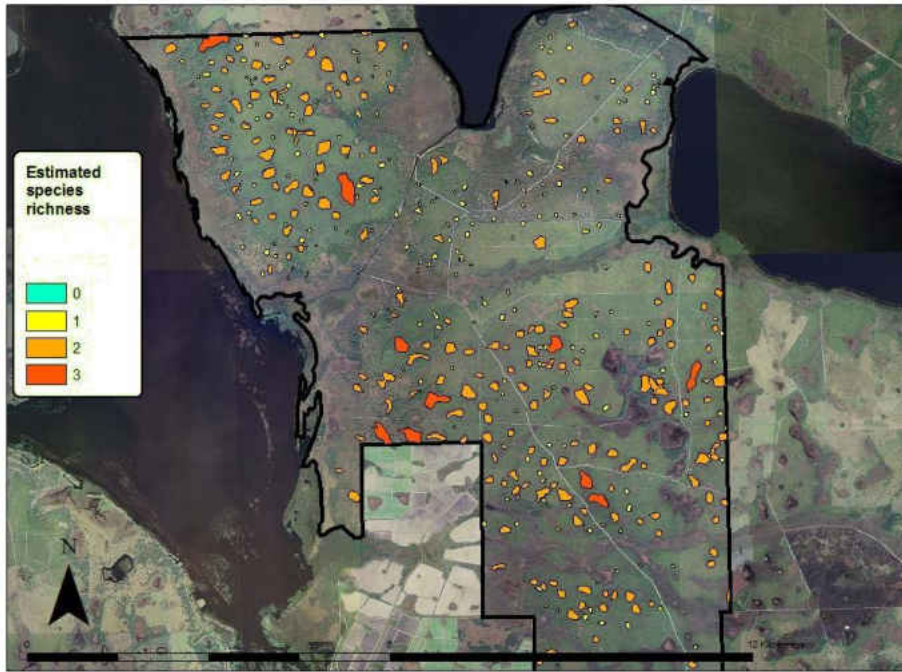


b)

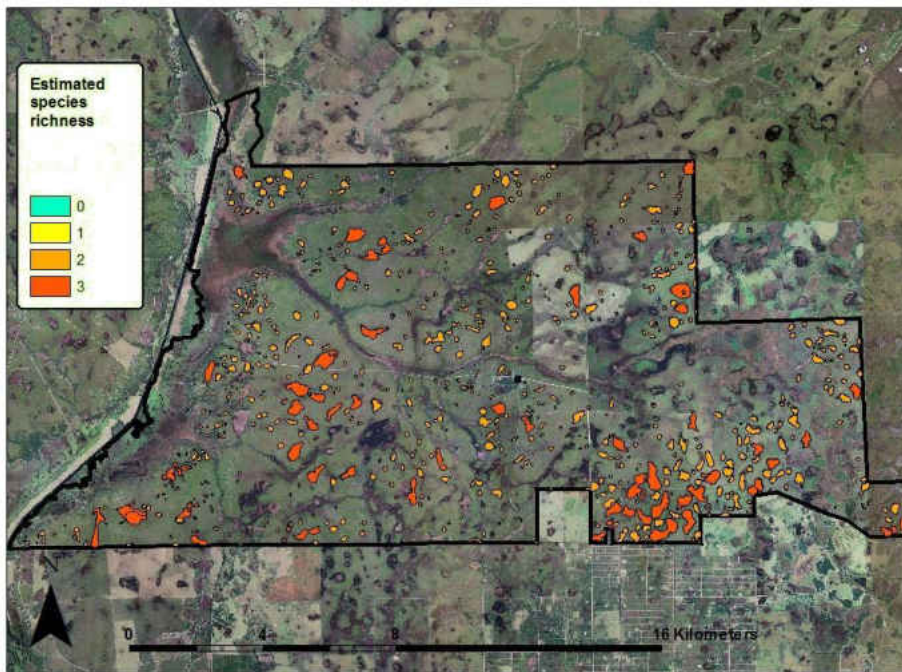


Spring season overall patch species richness estimates for a) TLWMA and b) KPPSP

a)

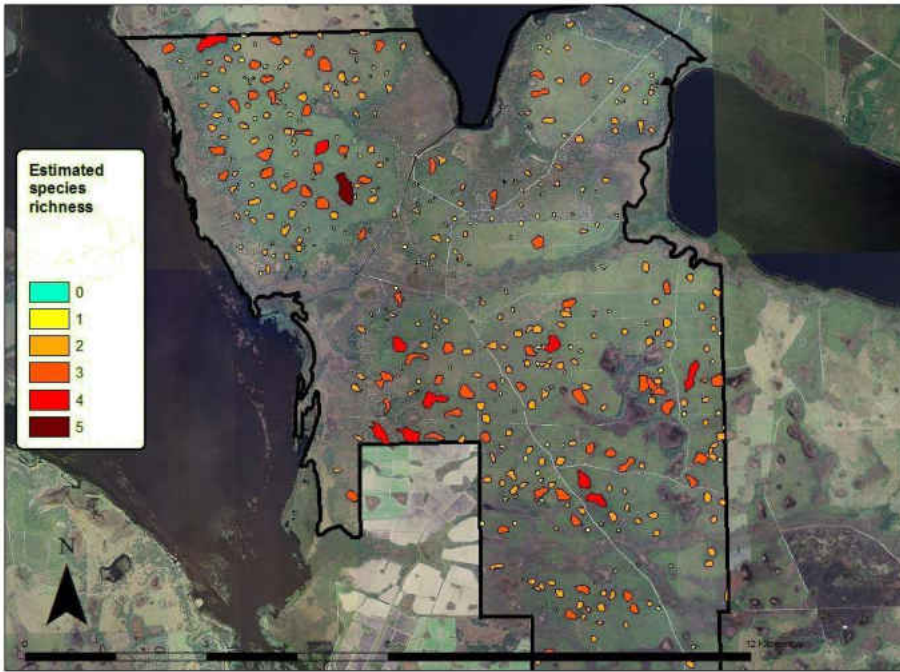


b)

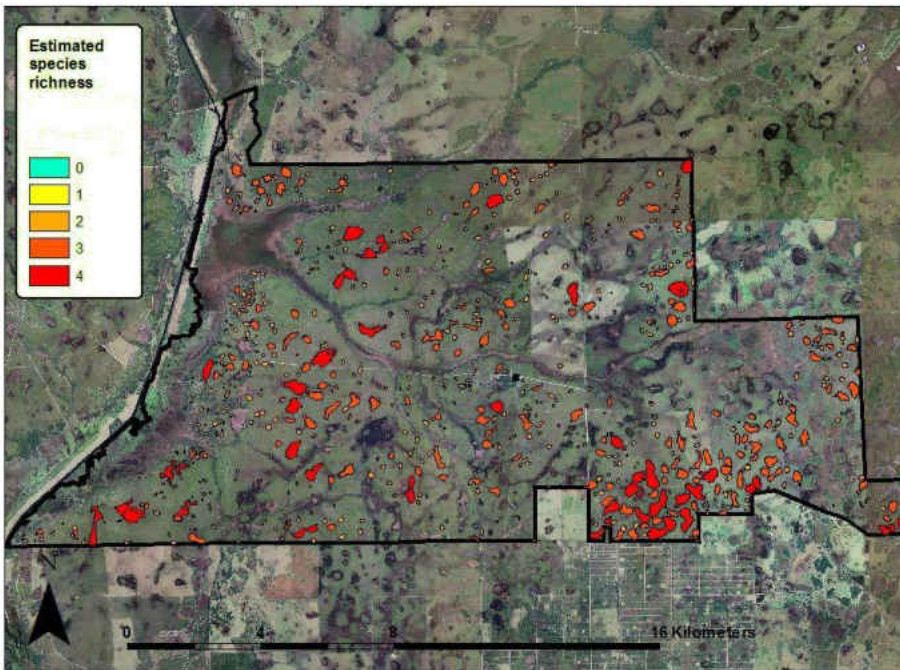


Fall perching guild patch species richness estimates for a) KPPSP and b) TLWMA

a)



b)

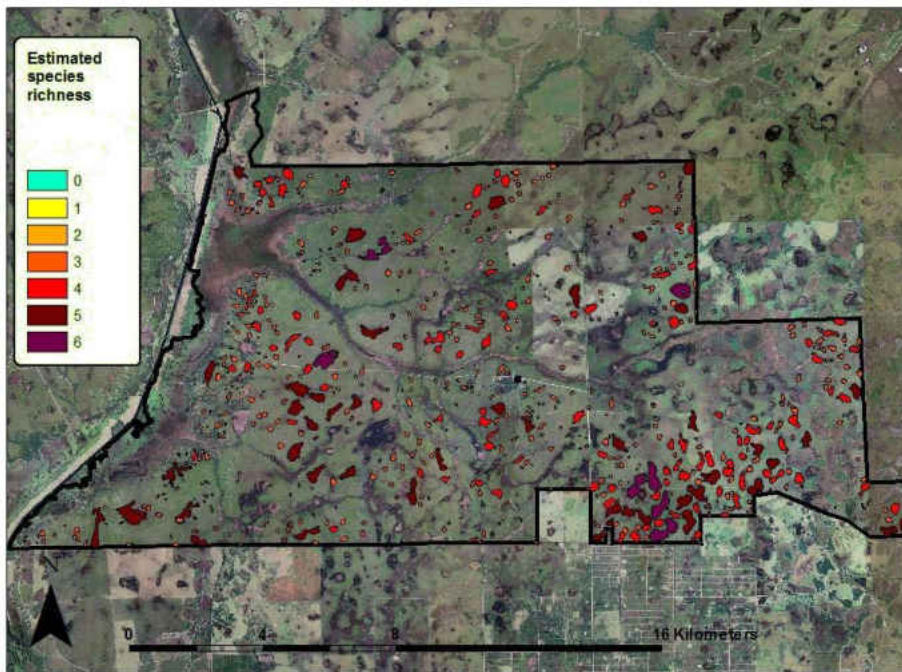


Winter perching guild patch species richness estimates for a) TLWMA and b) KPPSP

a)

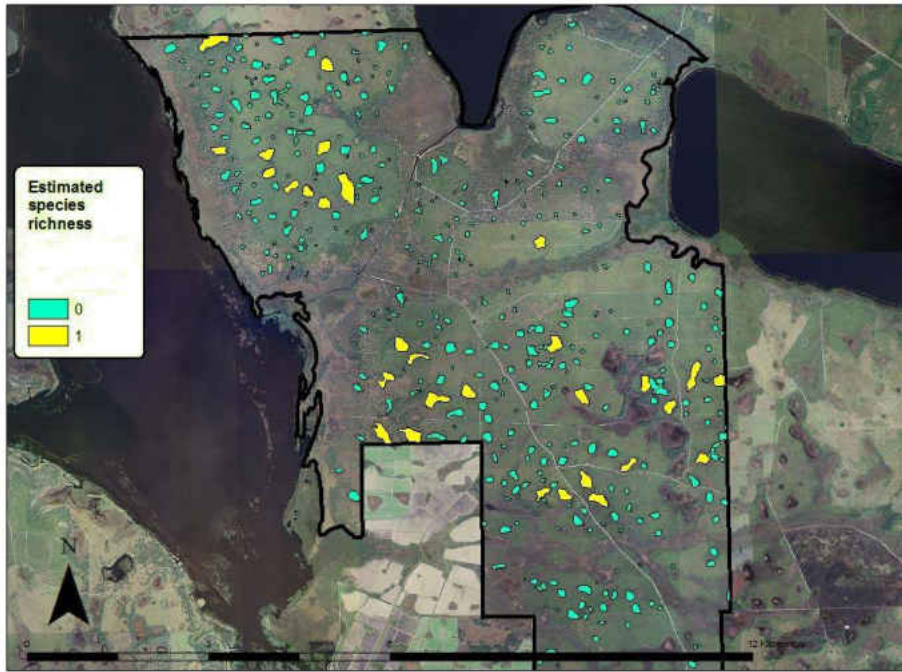


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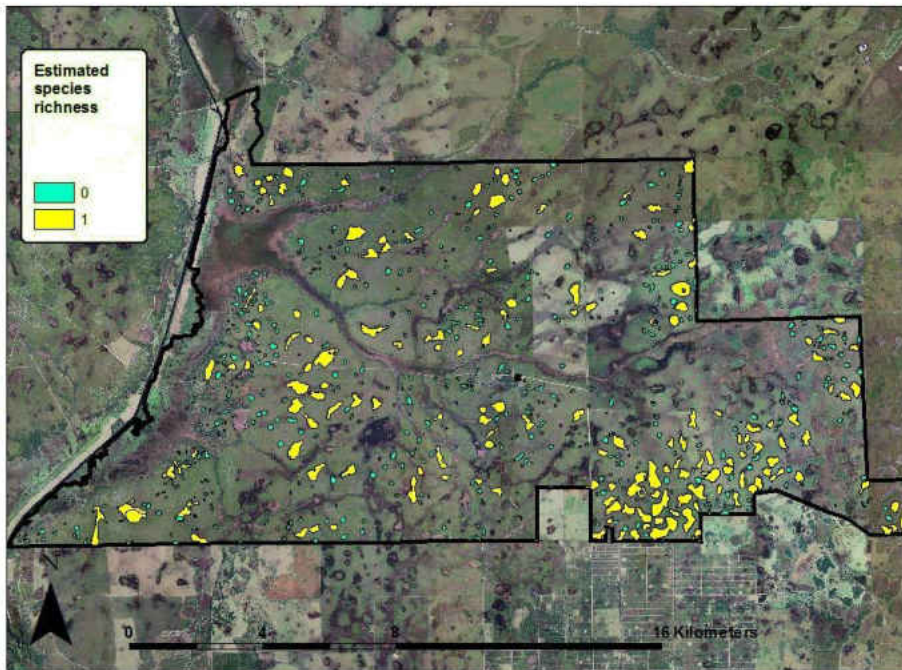


Spring perching guild patch species richness estimates for a) TLWMA and b) KPPSP

a)

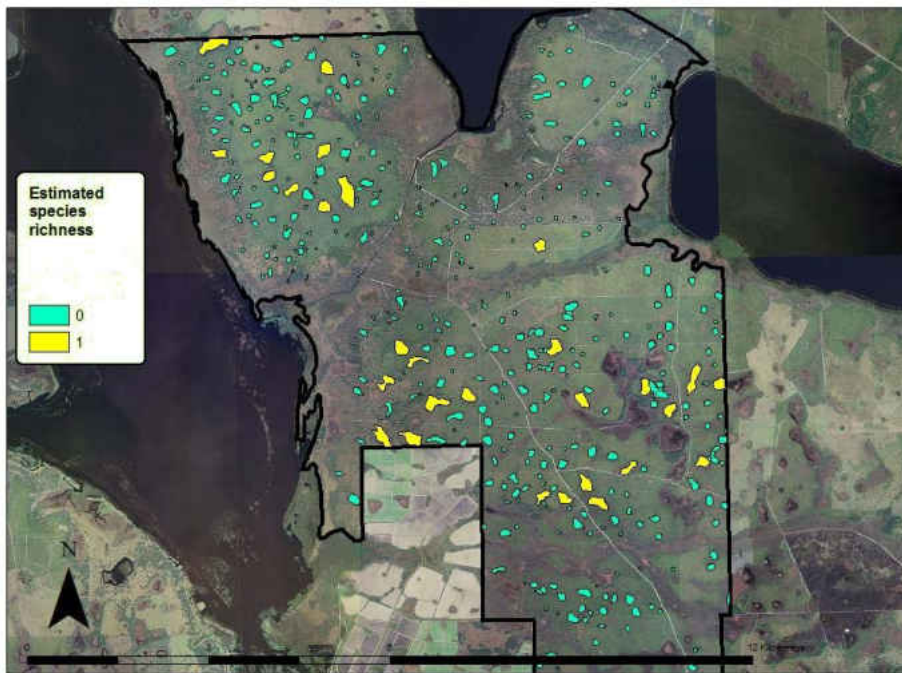


b)

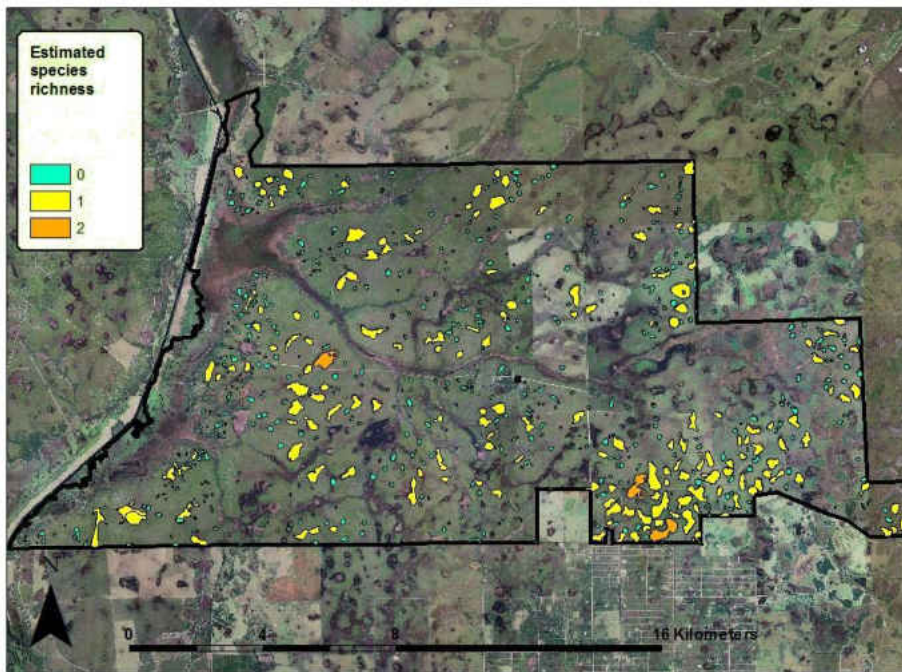


Fall wading guild patch species richness estimates for a) TLWMA and b) KPPSP

a)

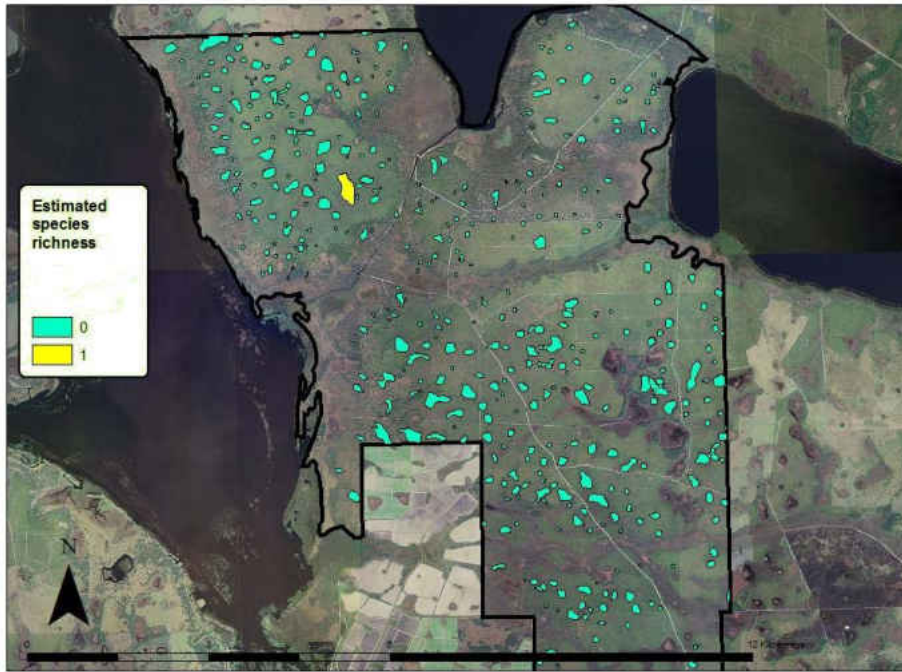


b)

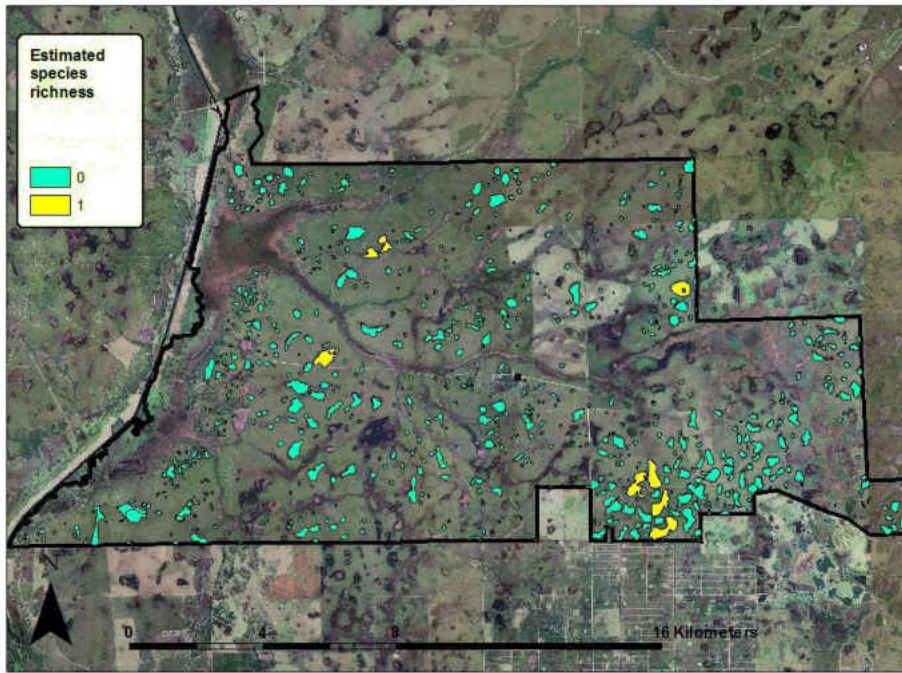


Winter wading guild patch species richness estimates for a) TLWMA and b) KPPSP

a)



b)



Spring wading guild patch species richness estimates for a) TLWMA and b) KPPSP

REFERENCES

- Abrahamson, W. G., and D. C. Hartnett. 1990. Pine flatwoods and dry prairies. Pages 103-149 in R. L. Myers and J. J. Ewel, editors. *Ecosystems of Florida*. University of Central Florida Press, Orlando, FL
- Battaglia, L. L., and B. S. Collins. 2006. Linking hydroperiod and vegetation response in Carolina bay wetlands. *Plant Ecology* 184:173-185.
- Beier, P., and R. F. Noss. 1998. Do habitat corridors provide connectivity? *Conservation Biology* 12:1241-1252.
- Bell, J. L., and R. C. Whitmore. 2000. Bird nesting ecology in a forest defoliated by gypsy moths. *Wilson Bulletin* 112:524-531.
- Bibby, C.J., N.D. Burgess, D.A. Hill, and S. Mustoe. 2000. *Bird census techniques*, 2nd Edition. Academic Press, London. 302 pages
- Bradford, D. F., A. C. Neale, M. S. Nash, D. W. Sada, and J. R. Jaeger. 2003. Habitat patch occupancy by toads (*Bufo punctatus*) in a naturally fragmented desert landscape. *Ecology* 84:1012-1023.
- Breining, D. R., B. Toland, D. M. Oddy, and M. L. Legare. 2006. Landcover characterizations and Florida scrub-jay (*Aphelocoma coerulescens*) population dynamics. *Biological Conservation* 128:169-181.
- Brooks, R. T., and K. L. Doyle. 2001. Shrew species richness and abundance in relation to vernal pond habitat in southern New England. *Northeastern Naturalist* 8:137-148.
- Brown, M., and J. J. Dinsmore. 1986. Implications of marsh size and isolation for marsh bird management. *Journal of Wildlife Management* 50:392-397.
- Bullock, J. 2006. *Plants*. Pages 186-213 in W. J. Sutherland, editor. *Ecological census techniques*, 2nd Edition. Cambridge University Press, Cambridge.
- Burnham, K.P. and D.R. Anderson. 1998. *Model selection and multimodel inference: A practical information-theoretic approach* 2nd edition. Springer Science+Business Media, LLC, New York, NY
- Calabrese, J. M., and W. F. Fagan. 2004. A comparison-shopper's guide to connectivity metrics. *Frontiers in Ecology and the Environment* 2:529-536.
- Costanza, R., R. d'Arge, R. deGroot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton, and M. vandenBelt. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387:253-260.

- Custer, C. M., S. A. Suarez, and D. A. Olsen. 2004. Feeding habitat characteristics of the Great Blue Heron and Great Egret nesting along the Upper Mississippi River, 1995-1998. *Waterbirds* 27:454-468.
- Debinski, D.M. and R.D. Holt. 2000. A survey and overview of habitat fragmentation experiments. *Conservation Biology*. 14:342-355
- Dugger, K. M., F. Wagner, R. G. Anthony, and G. S. Olson. 2005. The relationship between habitat characteristics and demographic performance of Northern Spotted Owls in Southern Oregon. *Condor* 107:863-878.
- ESRI. 1999-2006. ArcGIS 9.1 <http://www.esri.com/>.
- FGDL. 2005. Florida Geographic Database Library, www.fgdl.org.
- Fink, A. D., F. R. Thompson, and A. A. Tudor. 2006. Songbird use of regenerating forest, glade, and edge habitat types. *Journal of Wildlife Management* 70:180-188.
- Finlay, J. C., S. Khandwala, and M. E. Power. 2002. Spatial scales of carbon flow in a river food web. *Ecology* 83:1845-1859.
- Florida Ecological Restoration Inventory. 2007. <http://feri.dep.state.fl.us/adsearch.asp>
- Florida Fish and Wildlife Commission. 2007. www.floridaconservation.org.
- Florida Natural Areas Inventory. 2006. <http://www.fnai.org>.
- Forcada, J., P. N. Trathan, K. Reid, E. J. Murphy, and J. P. Croxall. 2006. Contrasting population changes in sympatric penguin species in association with climate warming. *Global Change Biology* 12:411-423.
- Gibbons, D.W. and R.D. Gregory. 2006. Birds. Pages 186-213 in W. J. Sutherland, editor. *Ecological census techniques*, 2nd Edition. Cambridge University Press, Cambridge.
- Gibbons, J. W., C. T. Winne, D. E. Scott, J. D. Willson, X. Glaudas, K. M. Andrews, B. D. Todd, L. A. Fedewa, L. Wilkinson, R. N. Tsaliagos, S. J. Harper, J. L. Greene, T. D. Tuberville, B. S. Metts, M. E. Dorcast, J. P. Nestor, C. A. Young, T. Akre, R. N. Reed, K. A. Buhlmann, J. Norman, D. A. Croshaw, C. Hagen, and B. B. Rothermel. 2006. Remarkable amphibian biomass and abundance in an isolated wetland: Implications for wetland conservation. *Conservation Biology* 20:1457-1465.
- Greenberg, C. H., and J. D. Lanham. 2001. Breeding bird assemblages of hurricane-created gaps and adjacent closed canopy forest in the southern Appalachians. *Forest Ecology and Management* 154:251-260.

- Griffis-Kyle, K. L., and P. Beier. 2005. Migratory strategy and seasonal patterns of bird diversity in relation to forest habitat. *American Midland Naturalist* 153:436-443.
- Hannon, S. J., and F. K. A. Schmiegelow. 2002. Corridors may not improve the conservation value of small reserves for most boreal birds. *Ecological Applications* 12:1457-1468.
- Hanowski, J. M., and G. J. Niemi. 1995. Experimental design considerations for establishing an off-road, habitat-specific bird monitoring program using point counts. USDA Forest Service
- Hanski, I, and M.E. Gilpin. 1997. *Metapopulation biology: ecology, genetics, and evolution*. Academic Press, San Diego, CA. 512 pages
- Hanski, I. and O.E. Gaggiotti. 2004. *Ecology, genetics, and evolution of metapopulations*. Academic Press, New York, NY. 696 pages
- Hansson, L. A., C. Bronmark, P. A. Nilsson, and K. Abjornsson. 2005. Conflicting demands on wetland ecosystem services: nutrient retention, biodiversity or both? *Freshwater Biology* 50:705-714.
- Harrison, S., and A.D. Taylor. 1997. Empirical evidence for metapopulation dynamics. Pp. 27-42 in I.A. Hanski and M.E. Gilpin (eds.) *Metapopulation biology: ecology, genetics, and evolution*. Academic Press, San Diego, CA
- Helzer, C. J., and D. E. Jelinski. 1999. The relative importance of patch area and perimeter-area ratio to grassland breeding birds. *Ecological Applications* 9:1448-1458.
- Holt, R. D., and M. F. Hoopes. 2005. Food web dynamics in a metacommunity context. Pages 68-94 in M. Holyoak, M. A. Leibold, and R. D. Holt, editors. *Metacommunities: spatial dynamics and ecological communities*. University of Chicago Press, Chicago, IL.
- Holt, R. D., J. H. Lawton, G. A. Polis, and N. Martinez. 1999. Trophic rank and the species-area relationship. *Ecology* 80:1495-1504.
- Holyoak, M., M. A. Leibold, N. M. Mouquet, R. D. Holt, and M. F. Hoopes. 2005. *Metacommunities: a framework for large-scale community ecology*. Pages 1-32 in M. Holyoak, M. A. Leibold, and R. D. Holt, editors. *Metacommunities: spatial dynamics and ecological communities*. University of Chicago Press, Chicago, IL.
- Horn, D. J., R. J. Fletcher, and R. R. Koford. 2000. Detecting area sensitivity: A comment on previous studies. *American Midland Naturalist* 144:28-35.
- Huff, M. H., K. A. Bettinger, H. L. Ferguson, M. J. Brown, and B. Altman. 2000. A habitat-based point-count protocol for terrestrial birds, emphasizing Washington and Oregon. USDA Forest Service

- Iles, T. C. 1993. Multiple Regression. Pages 127-172 in J. C. Fry, editor. *Biological Data Analysis*. IRL Press at Oxford University Press, Oxford, UK.
- Jackson, R. B., S. R. Carpenter, C. N. Dahm, D. M. McKnight, R. J. Naiman, S. L. Postel, and S. W. Running. 2001. Water in a changing world. *Ecological Applications* 11:1027-1045.
- Jones, K. L., G. L. Krapu, D. A. Brandt, and M. V. Ashley. 2005. Population genetic structure in migratory sandhill cranes and the role of Pleistocene glaciations. *Molecular Ecology* 14:2645-2657.
- Koenig, W. D. 2005. Persistence in adversity: Lessons from the ivory-billed woodpecker. *Bioscience* 55:646-647.
- Krauss, J., I. Steffan-Dewenter, and T. Tschardt. 2003. How does landscape context contribute to effects of habitat fragmentation on diversity and population density of butterflies? *Journal of Biogeography* 30:889-900.
- Kritzer, J.P. and P.F. Sale. 2006. *Marine metapopulations*. Academic Press, New York, NY. 544 pages
- Kudray, G. M., and M. R. Gale. 2000. Evaluation of National Wetland Inventory maps in a heavily forested region in the upper Great Lakes. *Wetlands* 20:581-587.
- LABINS. 2006. Land boundary identification system. www.data.labins.org.
- Lariviere, S., and M. Lepage. 2000. Effect of a water-level increase on use by birds of a lakeshore fen in Quebec. *Canadian Field-Naturalist* 114:694-696.
- Leibowitz, S. G. 2003. Isolated wetlands and their functions: An ecological perspective. *Wetlands* 23:517-531.
- Levesque, E. 1996. Minimum area and cover-abundance scales as applied to polar desert vegetation. *Arctic and Alpine Research* 28:156-162.
- Levey, D. J., B. M. Bolker, J. J. Tewksbury, S. Sargent, and N. M. Haddad. 2005. Effects of landscape corridors on seed dispersal by birds. *Science* 309:146-148.
- Levins, R. 1969. Some demographic and genetic consequences of environmental heterogeneity for biological control. *Bulletin of the Entomological Society of America*. 15:237-240
- Lindenmayer, D.B. and J.F. Franklin. 2002. *Conserving forest biodiversity: a comprehensive multi-scaled approach*. Island Press, Washington, D.C. 351 pages

- Lobel, S., T. Snall, and H. Rydin. 2006. Metapopulation processes in epiphytes inferred from patterns of regional distribution and local abundance in fragmented forest landscapes. *Journal of Ecology* 94:856-868.
- Lohr, S. M., S. A. Gauthreaux, and J. C. Kilgo. 2002. Importance of coarse woody debris to avian communities in loblolly pine forests. *Conservation Biology* 16:767-777.
- Lynch, J. F., and D. F. Whigham. 1984. Effects of Forest Fragmentation on Breeding Bird Communities in Maryland, USA. *Biological Conservation* 28:287-324.
- MacArthur, J. W., and E. O. Wilson. 1967. *The theory of island biogeography*. Princeton University Press, Princeton, N.J. 203 pages
- Maher, D.S. and H.W. Kale II. 2005. *Florida's birds: A field guide and reference, 2nd Edition*. Pineapple Press, Sarasota, Florida. 359 pages
- Moilanen, A., and M. Nieminen. 2002. Simple connectivity measures in spatial ecology. *Ecology* 83:1131-1145.
- Mortberg, U. M. 2001. Resident bird species in urban forest remnants; landscape and habitat perspectives. *Landscape Ecology* 16:193-203.
- National Geographic. 2003. *Reference atlas to the birds of North America*. National Geographic Society, Washington, D.C. 480 pages
- Naugle, D. E., K. F. Higgins, S. M. Nusser, and W. C. Johnson. 1999. Scale-dependent habitat use in three species of prairie wetland birds. *Landscape Ecology* 14:267-276.
- Noss, R.F. 1993. Wildlife corridors. Pp. 43-68 in D.S. Smith and P.C. Hellmund (eds.) *Ecology of greenways*. University of Minnesota Press, Minneapolis, MN.
- Noss, R.F. and K.M. Daly. 2006. Incorporating connectivity into broad-scale conservation planning. Pp. 587-619 in K.R. Crooks and M. Sanjayan (eds.) *Connectivity conservation*. Cambridge University Press, Cambridge, UK.
- Opdam, P., D. Vandorp, and C. J. F. Terbraak. 1984. The effect of isolation on the number of woodland birds in small woods in the Netherlands. *Journal of Biogeography* 11:473-478.
- Orzell, S.L. and E.L. Bridges. 2004. Species composition and environmental characteristics of Florida dry prairies from the Kissimmee river region of south-central Florida. Pages 100-135 in R. Noss editor. *Land of Fire and Water: The Florida Dry Prairie Ecosystem*. Proceedings of the Florida Dry Prairie Conference.

- Ozesmi, U. and W.J. Mitsch. 1997. A spatial habitat model for the marsh breeding red-winged blackbird (*Agelaius phoeniceus* L.) in coastal Lake Erie wetlands. *Ecological Modelling* 101(2-3): 139-152
- Paracuellos, M. 2006a. How can habitat selection affect the use of a wetland complex by waterbirds? *Biodiversity and Conservation* 15:4569-4582.
- Paracuellos, M. 2006b. Relationships of songbird occupation with habitat configuration and bird abundance in patchy reed beds. *Ardea* 94:87-98.
- Pearman, P. B. 2002. The scale of community structure: Habitat variation and avian guilds in tropical forest understory. *Ecological Monographs* 72:19-39.
- Peintinger, M., A. Bergamini, and B. Schmid. 2003. Species-area relationships and nestedness of four taxonomic groups in fragmented wetlands. *Basic and Applied Ecology* 4:385-394.
- Polis, G. A., W. B. Anderson, and R. D. Holt. 1997. Toward an integration of landscape ecology and food web ecology: the dynamics of spatially subsidized food webs. *Annual Review of Ecology and Systematics* 28:289-316.
- Polis, G. A., M. E. Power, and G. Huxel. 2004. *Food webs at the landscape level*. University of Chicago Press, Chicago, IL.
- Postel, S., and S. Carpenter. 1997. Freshwater ecosystem services. Pages 195-214 in G. Daily, editor. *Nature's services: societal dependence on natural ecosystems*. Island Press, Washington, D.C.
- Rice, W.R. 1989. Analyzing tables of statistical tests. *Evolution* 43:223-225
- Richter-Boix, A., G. A. Llorente, and A. Montori. 2007. Structure and dynamics of an amphibian metacommunity in two regions. *Journal of Animal Ecology* 76:607-618.
- Scheffers, B. R., J. B. C. Harris, and D. G. Haskell. 2006. Avifauna associated with ephemeral ponds on the Cumberland Plateau, Tennessee. *Journal of Field Ornithology* 77:178-183.
- Schooley, R. L., and L. C. Branch. 2006. Space use by round-tailed muskrats in isolated wetlands. *Journal of Mammalogy* 87:495-500.
- Serrano, L., and K. Fahd. 2005. Zooplankton communities across a hydroperiod gradient of temporary ponds in the Donana National Park (SW Spain). *Wetlands* 25:101-111.
- Sibley, D.A. 2003. *The Sibley field guide to birds of eastern North America*. Alfred A. Knopf, New York. 431 pages

- Skorka, P., R. Martyka, and J. D. Wojcik. 2006. Species richness of breeding birds at a landscape scale: which habitat type is the most important? *Acta Ornithologica* 41:49-54.
- Stolt, M. H., and J. C. Baker. 1995. Evaluation of National Wetland Inventory maps to inventory wetlands in the Southern Blue Ridge of Virginia. *Wetlands* 15:346-353.
- Tavernini, S., G. Mura, and G. Rossetti. 2005. Factors influencing the seasonal phenology and composition of zooplankton communities in mountain temporary pools. *International Review of Hydrobiology* 90:358-375.
- Taylor, W. K. 1998. Florida wildflowers in their natural communities. University of Florida Press, Gainesville, FL
- Tiner, R. W. 2003. Geographically isolated wetlands of the United States. *Wetlands* 23:494-516.
- Tischendorf, L., and L. Fahrig. 2000. How should we measure landscape connectivity? *Landscape Ecology* 15:633-641.
- Triantis, K. A., M. Mylonas, K. Lika, and K. Vardinoyannis. 2003. A model for the species-area-habitat relationship. *Journal of Biogeography* 30:19-27.
- Trocki, C. L., and P. W. C. Paton. 2006. Assessing habitat selection by foraging egrets in salt marshes at multiple spatial scales. *Wetlands* 26:307-312.
- Twedt, D. J., R. R. Wilson, J. L. Henne-Kerr, and R. B. Hamilton. 2001. Nest survival of forest birds in the Mississippi Alluvial Valley. *Journal of Wildlife Management* 65:450-460.
- Volkov, I., J. R. Banavar, S. P. Hubbell, and A. Maritan. 2003. Neutral theory and relative species abundance in ecology. *Nature* 424:1035-1037.
- United States Fish & Wildlife Service. 2006. National Wetlands Inventory. www.nwi.gov
- Watson, J. E. M., R.J. Whittaker, and D. Freudenberger. 2005. Bird community responses to habitat fragmentation: how consistent are they across landscapes? *Journal of Biogeography* 32:1353-1370.
- Watts, A., G. Tanner and R. Dye. 2004. Restoration of dry prairie using fire and roller chopping. Pages 225-230 in R. Noss editor. *Land of Fire and Water: The Florida Dry Prairie Ecosystem*. Proceedings of the Florida Dry Prairie Conference.
- Weyrauch, S. L., and T. C. Grubb. 2004. Patch and landscape characteristics associated with the distribution of woodland amphibians in an agricultural fragmented landscape: an information-theoretic approach. *Biological Conservation* 115:443-450.

Wilson, D. S. 1992. Complex Interactions in Metacommunities, with Implications for Biodiversity and Higher Levels of Selection. *Ecology* 73:1984-2000.

Zamudio, K. R., and A. M. Wiczorek. 2007. Fine-scale spatial genetic structure and dispersal among spotted salamander (*Ambystoma maculatum*) breeding populations. *Molecular Ecology* 16:257-274.