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WETLAND SUITABILITY FOR WATERBIRDS IN ILLINOIS

BY

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THESIS

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ABSTRACT

Waterfowl, marsh birds, shorebirds, and other waterbirds rely on wetlands to meet dynamic physiological needs. However, the midwestern United States has undergone extensive wetland loss, and extant wetlands are often degraded. Currently, there is limited information available to estimate the quantity of suitable habitat for wetland-dependent birds to prioritize wetland management and inform conservation planning processes. Furthermore, the hydrology and other physical characteristics of wetlands that determine their suitability for waterbirds often differ among species, foraging guilds, and temporally. The suitability of stopover wetlands used by spring migrants influences survival and body condition of some species as they prepare for breeding, and most secretive marsh birds species have experienced population declines presumably linked to losses in quantity and suitability of habitat during the breeding season.

Therefore, I identified factors that influenced the availability of suitable waterbird habitat and provided baseline data on the spatial distribution and extent of wetlands capable of supporting migratory and breeding bird populations in Illinois. I assessed habitat conditions in Illinois wetlands during spring, summer, and autumn 2016 – 2017 across Illinois Natural Divisions. Each spring, I surveyed wetland sites for dabbling ducks and other waterbirds (e.g., geese, diving ducks, herons) using aerial surveys, and conducted call-broadcast surveys for secretive marsh birds at sites with emergent vegetation spanning the marsh bird breeding and migration chronology in Illinois. I assessed wetland suitability for dabbling ducks, marsh birds, and shorebirds; waterfowl and other waterbird abundances; and marsh bird occupancy as a function of local wetland characteristics, landscape context and integrity, and wetland management practices and characteristics to assist conservation planners prioritize wetland restoration and enhancement in the Midwest, USA.

Across all survey periods, suitable vegetative and hydrological conditions for wetland-dependent bird guides comprised a small portion of the total area of wetlands and deepwater habitats within the National Wetlands Inventory. Furthermore, important vegetation cover and inundation levels considered suitable for waterbirds varied among NWI classes and Illinois Natural Divisions. Suitable emergent vegetation for migrating and breeding marsh birds and mudflats and shallowly inundated foraging habitats for migrating shorebirds were particularly limited during survey periods, and wetlands with greater complexity and connectivity to other wetland types offered the greatest proportion of suitable habitat resources. Surface water

inundation and vegetation coverage were the most important predictors of waterbird density during spring migration; for every 10% increase in the proportion of inundated vegetation, dabbling duck density increased 34.8%. Similarly, local wetland characteristics such as inundated persistent emergent vegetation and forested cover, and to a lesser extent, persistent emergent vegetation, were important components predicting marsh bird occupancy in Illinois.

My study highlights a poor representation by the National Wetlands Inventory of the quantity of suitable habitat conditions for migratory wetland-dependent birds, specifically wetland inundation and vegetation cover. Future emphasis should be placed on identifying variables appropriate for predicting wetland suitability for waterbirds by combining National Wetlands Inventory data with other available spatial data and addressing demographic responses (e.g., survival, nest success, breeding propensity) of waterbirds to suitable habitat in Illinois since suitable habitat appears to be very limited for some waterbird guilds.

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TABLE OF CONTENTS

CHAPTER 1: GENERAL INTRODUCTION AND LITERATURE REVIEW	1
1.1 REFERENCES.....	6
CHAPTER 2: SUITABILITY OF WETLANDS IN ILLINOIS FOR MIGRATING AND BREEDING WATERBIRDS	11
2.1 ABSTRACT	11
2.2 INTRODUCTION	12
2.3 MATERIALS AND METHODS	18
2.4 RESULTS	27
2.5 DISCUSSION	33
2.6 TABLES AND FIGURES	42
2.7 REFERENCES	103
CHAPTER 3: LOCAL AND LANDSCAPE CHARACTERISTICS THAT INFLUENCE WATERBIRD USE OF WETLANDS DURING SPRING MIGRATION IN ILLINOIS	114
3.1 ABSTRACT	114
3.2 INTRODUCTION	114
3.3 MATERIALS AND METHODS	118
3.4 RESULTS	124
3.5 DISCUSSION	126
3.6 TABLES AND FIGURES	133
3.7 REFERENCES	145
CHAPTER 4: HABITAT CHARACTERISTICS ASSOCIATED WITH MIGRATING AND BREEDING SORA IN ILLINOIS	154
4.1 ABSTRACT	154
4.2 INTRODUCTION	154
4.3 MATERIALS AND METHODS	158
4.4 RESULTS	163
4.5 DISCUSSION	164
4.6 TABLES AND FIGURES	170
4.7 REFERENCES	181
CHAPTER 5: SUMMARY	188

CHAPTER 1: GENERAL INTRODUCTION AND LITERATURE REVIEW

Wetlands are declining globally, and some regions such as the midwestern United States have experienced extensive wetland losses since the 1900s (Tiner 1984, Mitsch and Gosselink 2000). Remaining wetlands are often degraded by factors including agricultural practices that alter hydrology (e.g. tiling or constructing drainage ditches), sedimentation, and invasive species (Zedler and Kercher 2005). Degraded wetlands are often less suitable for wetland-dependent species such as waterbirds (Soulliere et al. 2007, Forcey et al. 2011). Furthermore, losses of wetland area and degraded suitability of remaining wetlands are leading causes for waterbird population declines (Howe et al. 1989, Conway et al. 1994). Wetlands and deepwater areas (Cowardin et al. 1979), despite losses, are essential habitats for waterbirds. However, not all resources may be accessible to waterbirds at a given time. Furthermore, factors including wetland management, connectivity, complexity, hydrology, and climate impact wetlands and waterbirds. Here, I assessed local and landscape characteristics that impact wetlands and waterbird use focused on three waterbird groups of conservation interest and concern in Illinois—dabbling ducks, marsh birds, and shorebirds.

Dabbling ducks (*Anas* spp.), marsh birds (e.g., primarily Rallidae, also Ardeidae and Podicipedidae), and shorebirds (i.e., *Charadriiformes*) have diverse habitat requirements, and wetland managers often manipulate wetland hydrology and vegetation to provide food and other habitat resources. Active management for waterbirds (i.e., primarily for waterfowl) includes frequent drawdowns (e.g., <3-year intervals) to modify hydrology as well as soil or vegetation manipulation (e.g., disking, mowing, herbicide applications) to encourage annual vegetation communities (e.g., hemi-marsh, moist-soil management; Kaminski et al. 2009, Gray et al. 2013). Alternatively, passive management may include infrequent soil and vegetation disturbances and may lead to more woody and perennial species (Kross et al. 2008). Pankau (2008) found actively managed wetlands produced productive habitat even in drought years, whereas unmanaged wetlands remained dry. Kaminski et al. (2009) also found that waterbird use of managed wetlands was significantly greater than unmanaged wetlands because active hydrological management increased available habitat. Further, Smith et al. (2012) noted that wetlands with annual draw-downs or de-watering provide the most foraging habitat for shorebirds by increasing the area and availability of mudflats.

Wetland connectivity to other wetland types and to rivers and streams may impact the amount of suitable habitat for waterbirds and waterbird use. Highly connected wetlands or groups of wetlands in close proximity, known as wetland complexes, support various waterbird species and increase waterbird diversity by supplying varied microhabitats and areas for forage (Weller 1988, Fairbairn and Dinsmore 2001). For instance, Beatty et al. (2014a) found that mallards selected resource units adjacent to emergent wetlands, open water, and woody wetlands during spring migration, and Kahler (2013) found an increase in common gallinule (*Gallinula galeata*) and Virginia rail (*Rallus limicola*) abundance in areas with greater percent emergent marsh within 100-m and 10-km. On the other hand, wetland connectivity to river systems may have variable impacts on waterbird habitat. Stafford et al. (2010) found that submerged and floating-leaved aquatic vegetation, an important food source for many waterbird species (Fredrickson and Reid 1988, Combs and Fredrickson 1996), has decreased substantially in Illinois River Valley since the 1950s. They posited that this decline was likely due to extensive degradation of floodplain wetlands from sedimentation, highly fluctuating hydrology during the growing season associated with land development and channelization, and exotic species (e.g., exotic carp; Havera 1999). However, wetlands associated with large river systems with periodic flooding could provide more shorebird habitat than isolated palustrine and lacustrine wetlands (Smith et al. 2012).

Habitat complexity (Weller and Spatcher 1965) and interspersions of disparate vegetation types (Ringelman and Longcore 1982, Baschuk et al. 2012) are also important components of waterbird habitat. Avian diversity and abundance is often greatest in 50:50 water to vegetation ratios, known as “hemi-marsh” (Weller and Spatcher 1965, Kaminski and Prince 1981, Smith et al. 2004). Both hemi-marsh and patches of other vegetation types are important for isolating waterfowl pairs during spring and for predator avoidance during brood-rearing and molting periods (Murkin et al. 1982). Rehm and Baldassarre (2007) also found an increase in marsh bird abundance in marshes with high levels of vegetation and water interspersions. Additionally, aquatic invertebrate abundance, an integral component suitable waterbird habitat, is often highest in areas of interspersions of open water (in dense beds of submerged aquatic vegetation) and emergent vegetation (Voigts 1976) with a mixture of wetland types (Murkin et al. 1992).

Surrounding anthropogenic land use and disturbance impacts wetlands (Zedler and Kercher 2005, Mack 2006) and waterbirds (Havera et al. 1992, Peterson and Niemi 2007). For

instance, extensive leveeing and drainage has eliminated over 50% of the natural wetlands in the Illinois River Valley, and extant wetlands are further degraded by sedimentation, exotic species, and eutrophication from agricultural run-off (Havera 1999). Furthermore, Beatty et al. (2014) demonstrated that mallards shift wetland use in response to anthropogenic disturbances, site proximity to protected areas and sanctuaries, and total wetland area (i.e., within specified buffer). Wetland landscapes are an essential aspect of staging, stopover, and wintering habitat for migratory waterbirds (Webb et al. 2010, Pearse et al. 2012). Yet, habitat suitability and conditions of wetland landscapes such as the Prairie Pothole Region vary temporally and spatially and are closely related to environmental variables (e.g., weather and seasonal phenology; Herfindel et al. 2012).

Wetland hydrology is an integral part of a wetland system (Mitsch and Gosselink 2007) that dictates the availability and suitability for waterbirds (e.g., foraging habitat, water depth; Isola et al. 2000, Conway et al. 2005) and subsequent vegetation communities and functions within a wetland (Wilcox and Meeker 1999). Water availability and depth influence waterbird distribution, habitat use, and key behaviors such as foraging (Conway et al. 2005, Hagy et al. 2014). Water depth influences the accessibility of available foraging habitat directly because waterbird morphology and size limit where waterbirds can forage (Isola et al. 2000, Davis and Smith 2001, Ma et al. 2010, Collazo et al. 2016). A wetland's water regime describes the timing and permanence of inundation, where areas with more permanent water regimes (i.e., permanent, intermittently exposed and semi-permanent) are generally deeper and inundated for longer periods of time than areas with less permanent water regimes (i.e., seasonal, temporary, saturated; Cowardin et al 1979, Johnson et al. 2010). Marsh birds are often associated with more permanent wetlands with stable water levels as this promotes development of dense emergent vegetation such as cattail (*Typha* spp.; Anteau 2012). Furthermore, wetland hydrology, availability, and suitability are tightly linked to climate (e.g., May pond counts; USFWS 2017). Wetland hydrology and climate, though, vary between seasons and from year to year (Niemuth et al. 2010). Due to this variability, the amount of suitable habitat for waterbirds is stochastic and difficult to ascertain.

Adding to this challenge, wetland-dependent birds are primarily migratory and rely on a network of wetlands across a large geographic area to complete annual life histories, making conservation difficult to coordinate (Myers et al. 1987). To better synchronize conservation for

waterbirds across larger geographic areas, Joint Ventures – regional partnerships between federal and state governmental agencies, Native American tribes, and non-governmental organizations – have formed in biologically important conservation regions across the North America. Joint Ventures, such as the Upper Mississippi Great Lakes Region Joint Venture (e.g., Potter et al. 2007, Soulliere et al. 2007, Soulliere et al. 2017) identify waterbird population and habitat needs and prioritize regions and wetland types for conservation by scaling down national goals to establish regional objectives (Potter et al. 2007, Soulliere et al. 2007, Soulliere et al. 2017). Similarly, the Illinois Wetlands Campaign of the Illinois Comprehensive Wildlife Conservation Plan and Strategy set stepped-down state goals for habitat and focal species conservation from the regional Joint Venture plan (IDNR 2005). A major shortfall, however, is a lack of clear understanding of the amount of habitat currently on the landscape that is available to and considered suitable for waterbirds.

To estimate the suitability of available habitat for waterbirds, Joint Ventures, state natural resource departments, the U.S. Fish and Wildlife Service and other agencies frequently exploit existing wetland information and databases, such as the National Wetlands Inventory (NWI). The NWI follows the Cowardin et al. (1979) wetland and deepwater habitat classification system and includes hydrological systems, dominant vegetation or substrate classes (e.g., freshwater emergent, freshwater forested, etc.), water regimes (e.g., ranging from permanently flooded to saturated), and other special modifiers (i.e., impacts from humans or beavers such as impoundments or partial drainage). The NWI data are based primarily on aerial photographs taken in the mid-1980s (Cowardin and Golet 1995) and are often considered outdated especially for regions that have undergone extensive land use change during the past few decades. Furthermore, the aerial imagery was gathered at various times of the year (i.e., either in spring or summer [leaf-on] periods) further impacting patterns of omitting forested, small or seasonal wetlands in NWI data (Tiner 1997, Dvoretz et al. 2012, Matthews et al. 2016).

While the NWI offers baseline data about historic wetlands, due to its weaknesses (i.e., outdated aerial imagery, omission and commission), agencies are forced rely on broad assumptions when using the NWI data. This precludes accurate estimates of the suitability of available habitat for waterbirds (e.g., wetlands inundated <45 cm, inundated emergent vegetation, exposed mudflats, etc.) during critical periods at the scale required by many agencies (e.g., Illinois Department of Natural Resources and Upper Mississippi River and Great Lakes

Region Joint Venture). More specific estimates of waterbird habitat quantity and suitability are needed to supply information to the Illinois Wetlands Campaign of the Illinois Comprehensive Wildlife Conservation Plan and Strategy and Joint Venture for continued management and conservation for waterbirds. Habitat studies during critical periods will shed light on factors that influence wetland suitability for waterbirds, leading to more holistic management approaches, and provide baseline habitat suitability data for future Illinois wetland assessments. A statewide habitat assessment for waterbirds has not been conducted, and amount of suitable habitat during critical periods is not well understood.

Therefore, in my second chapter, I examined the presence and abundance of vegetation and inundation cover indicative of suitable habitat for three waterbird guilds specified as focal groups for conservation by the Upper Mississippi River and Great Lakes Region Joint Venture and Illinois Department of Natural Resources during critical periods in Illinois during 2015–2017. These critical periods and groups included 1) dabbling ducks during spring migration, 2) migrating and breeding marsh birds during late spring and early summer, and 3) migrating shorebirds during spring and autumn migrations to inform and advance habitat conservation. I examined the effects of wetland management, wetland connectivity/complexity, surrounding land use and disturbance, water regime, and climate on the presence and abundance of suitable habitat for waterbirds. In subsequent chapters, I examined local and landscape factors that influenced waterbird use of wetlands during spring migration (Chapter 3) and marsh bird occupancy during spring migration and the breeding period in Illinois (Chapter 4).

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CHAPTER 2: SUITABILITY OF WETLANDS IN ILLINOIS FOR MIGRATING AND BREEDING WATERBIRDS

2.1 ABSTRACT

Wetland-dependent bird populations may be limited by suitable habitat in regions where loss and degradation of wetland is pervasive, such as the midwestern United States. However, currently available spatial datasets, such as the National Wetlands Inventory (NWI), may overestimate habitat availability if total wetland areas include areas which are unsuitable for use by wetland-dependent species. I assessed proportional coverages of suitable habitat in spring, summer, and autumn, corresponding to critical time periods that could have population-level demographic effects, for three focal guilds of wetland-dependent, migratory birds relative to the NWI during 2015–2017. Suitable habitat conditions, including vegetation and water coverage, comprised a small portion of NWI polygons during most survey periods and years, but suitable area varied between NWI classes, Natural Divisions, and focal periods. Shallowly inundated wetlands (<45 cm) suitable for foraging by dabbling ducks comprised 29% of NWI area in Illinois during spring migration. Highly interspersed vegetation and open water (i.e., hemi-marsh) considered suitable dabbling ducks, other waterfowl and marsh birds, was extremely rare and areal cover was <1% of wetland area surveyed. Suitable habitat for migrating and breeding marsh birds (i.e., flooded, emergent vegetation) and migrating shorebirds (i.e., mudflats) was particularly limited during critical periods. For secretive marsh birds, NWI polygons had 5% cover of inundated dense persistent emergent vegetation (e.g., *Typha* spp.) and 9% inundated non-persistent emergent vegetation (e.g., *Polygonum* spp., *Echinochloa* spp., *Leersia* spp., etc.). Estimated mudflat cover for migrating shorebirds was also low (e.g., ≤6%) but was greater in early summer than autumn (e.g., 6% vs. 4%) during 2016–2017. Wetlands with greater complexity and connectivity to other wetland types had the greatest cover of suitable inundation and vegetation. Conservation planners should consider adjusting their estimates of wetland availability from spatial databases, such as NWI, when used to evaluate wetland suitability or supply on the landscape for wetland-dependent migratory birds during critical periods since significant proportions of forested and emergent wetland area was not flooded, were flooded at unsuitable depths, and/or wetlands lacked suitable vegetation resources. Restoration activities focused on wetland complexes may increase suitability of wetlands for a wide variety of bird species. Future research should identify variables appropriate for predicting wetland suitability

for waterbirds by combining National Wetlands Inventory data with other available spatial data to address demographic responses (e.g., survival, nest success, breeding propensity) of waterbirds to habitat suitability in Illinois. Increasing our understanding of the factors influencing the dynamics of wetland suitability and availability throughout the annual cycle will assist future conservation efforts for these species.

2.2 INTRODUCTION

Wetlands are declining globally, and some regions, such as the midwestern United States, have experienced extensive wetland losses since the 1900s (Tiner 1984, Mitsch and Gosselink 2015). For instance, Illinois has lost >85% of its original 3.2 million ha of wetlands (Dahl 2006), and emergent wetlands that provide critical waterbird habitat are still decreasing (USFWS 2015). Remaining wetlands are often degraded by factors including agricultural practices that alter hydrology (e.g., tiling or ditching), sedimentation, and invasive species (Zedler and Kercher 2005). Degraded wetlands often provide fewer resources for wetland-dependent species, such as waterbirds (Soulliere et al. 2007, Forcey et al. 2011), and wetland loss and degradation are likely the leading causes for population declines of some wetland-dependent bird species (Howe et al. 1989, Conway et al. 1994, Anteau and Afton 2011).

Despite significant losses and degradation in North America, wetlands provide essential habitat resources to support waterbirds throughout their annual cycle. Habitat resources include food, vegetation cover (e.g., thermal, escape, etc.), and hydrological characteristics required by waterbirds (Weller and Spatcher 1965, Eddleman 1988, Helmers 1992). However, not all resources may be accessible to waterbirds at a given time, and thus habitat is also influenced by the ecological constraints that limit an organism's ability to access a resource (e.g., competition, physical inaccessibility, etc.; Johnson 2007). Thus, high-quality habitat for many species of waterbirds includes suitable habitat resources (e.g., vegetation that provides food and cover) but also a presence of surface water at the appropriate depths and hydroperiod to positively affect occupancy, abundance, or population demographics. Unfortunately, only limited information is available to monitor the quantity and suitability of available habitat for wetland-dependent birds during critical periods (i.e., migration, breeding) of their annual cycle. Adding to this challenge, wetland-dependent birds are primarily migratory and rely on a network of wetlands across a

large geographic area to complete annual life histories, making conservation difficult to coordinate (Myers et al. 1987).

However, the timing of availability and physical characteristics of wetlands that determine their suitability to support waterbirds (i.e., relative quality) often differs among species and foraging guilds. Regional conservation planners, including the Illinois Department of Natural Resources and Upper Mississippi River and Great Lakes Region (UMRGLR) Joint Venture prioritize habitat conservation delivery within their geographies and consider dabbling ducks (e.g., Soulliere et al. 2017), secretive marsh birds (Soulliere et al. 2007), and shorebirds (Potter et al. 2007) priority guilds for conservation in the Midwest. However, there is a critical need for a clear understanding of the amount of habitat on the landscape that is available and considered suitable for these waterbird guilds to improve biological models and step-down objectives from continental plans.

Dabbling ducks (*Anas* spp.) migrate through Illinois during autumn and spring (Havera et al. 1999), and stopover and pre-breeding activities potentially affect survival and future breeding success of dabbling ducks (Arzel et al. 2006, Devries et al. 2008). Thus, waterfowl conservation plans (e.g., UMRGLR-Joint Venture, Soulliere et al. 2017) consider spring migration to be a critical period for waterfowl and indicate that the amount of suitable habitat may affect future breeding success and influence populations (Arzel et al. 2006, Straub et al. 2012). Specifically, the quantity and suitability of foraging habitat is presumed to be limiting (Straub et al. 2012). Dabbling ducks typically forage for seeds, tubers, and invertebrates (Swanson et al. 1985) within 30–45 cm of the water's surface, and thus require shallowly flooded areas to access benthic forage resources (McGilvrey 1966, Sousa and Farmer 1983, Isola et al. 2000, Taft et al. 2002). Further, mallards (*Anas platyrhynchos*) and other dabbling ducks increase selection of woody and forested wetlands during spring migration (Beatty et al 2014b, Williams 2018). Dabbling ducks also select wetland areas of interspersed vegetation with equal proportions of vegetation to water known as hemi-marsh (Weller and Spatcher 1965, Kaminski and Prince 1981).

Secretive marsh birds (e.g., primarily Rallidae, also Ardeidae and Podicipedidae) also migrate through and breed in Illinois (Kleen et al. 2004) and rely on emergent wetlands. Most marsh birds require emergent vegetation, both non-persistent emergent (NPE) and dense

persistent emergent vegetation (PEM), for breeding and cover (Darrah and Kremetz 2010) and often avoid forested wetlands (Bolenbaugh et al. 2011). Although vegetation cover preference varies among species (Weller and Spatcher 1965, Lor and Malecki 2006), American coots (*Fulica americana*), common gallinules (*Gallinula galeata*) (Weller and Spatcher 1965) and grebes (Family *Podicipedidae*) nest over water and prefer interspersions of dense emergent vegetation (Austin and Buhl 2011) and use deeper and more open wetlands than those of other marsh birds (Brackney and Bookhout 1982). In contrast, soras (*Porzana carolina*), least bitterns (*Ixobrychus exilis*), and Virginia rails (*Rallus limicola*) nest above water in dense emergent vegetation (Weller and Spatcher 1965) in shallow to intermediate water depths (Greise et al. 1980, Johnson and Dinsmore 1986). However, water depth may be relatively unimportant in contrast to the presence of emergent vegetation. Marsh bird population declines are presumably linked to losses in quantity of suitable wetland habitat (Conway et al. 1994, Soulliere et al. 2007, Bolenbaugh et al. 2011, Conway 2011), and decreases in emergent vegetation and hemi-marsh conditions were likely the driver in multiple marsh bird species declines from 1980 – 2005 in Illinois (Ward et al. 2010). Therefore, the waterbird strategy committee of the UMRGLR-Joint Venture (e.g., Soulliere et al. 2007) posits that breeding habitat is likely limiting in this region.

Shorebirds (Order: Charadriiformes) are a morphologically diverse group of wetland dependent birds that have experienced declines >70% in some areas since the 1970s (Howe et al. 1989). Population declines are tied to wetland loss and decreased wetland suitability (Howe et al. 1989, Brown et al. 2001, Potter et al. 2007). Suitable foraging habitat for shorebirds differs interspecifically, but generally, shorebirds forage in wetlands with sparse to no vegetation (Baker 1979, Twedt 2013), and within wetlands with exposed mudflats and very shallow water (i.e., <10 cm; Helmers 1992, Taft et al. 2002, Skagen et al. 2005, Canepuccia et al. 2007, Potter et al. 2007). Water depth is a vital wetland characteristic for shorebirds, as they forage in a narrower range of water levels than other wetland birds (Taft et al. 2002, Collazo et al. 2016). However, shorebird habitat is variable and often scarce in wetlands, and much of the shallow water and mudflat zones used by shorebirds has been lost or altered in the Midwestern United States (Skagen et al. 2008). Migratory shorebirds are seasonally dependent on a sequence of sites to complete their extensive annual migrations (i.e., between Southern and Northern Hemispheres; Myers et al. 1987). Thus, resting and refueling during migration is paramount (Myers 1983, Helmers 1992). Moreover, the shorebird strategy committee of the UMRGLR-Joint Venture

(e.g., Potter et al. 2007) and others (Norris 2015) consider spring migration more limiting than autumn, despite seasonal differences that suggest less habitat is available during autumn migration (Skagen et al. 1999, Horath et al. 2002, Gray et al. 2013, Twedt 2013).

Conservation planners typically use existing spatial databases, such as the National Wetlands Inventory (NWI), to estimate potential habitat availability for waterbirds on the landscape, despite the lack of reliable estimates of habitat suitability. The NWI follows the Cowardin et al. (1979) classification system and is based primarily on aerial photographs taken in the mid-1980s (Cowardin and Golet 1995). Although the NWI provides a nationally-available dataset of wetlands' type and spatial distribution, much of the data are outdated, many small or seasonal wetlands are omitted, and classifications may not accurately reflect true habitat availability or suitability for waterbirds (Tiner 1997, Dvoretz et al. 2012, Matthews et al. 2016). For instance, the NWI lacks specificity regarding wetland plant species composition and wetland inundation depths, which greatly influence habitat value to waterbirds. Antiquated aerial imagery and patterns of omission and commission force agencies to rely on broad assumptions when using the NWI data, precluding more accurate estimates of the suitability of available habitat for waterbirds.

Wetland management through hydrological (e.g., provide preferred foraging depths) and vegetation manipulation (e.g., mowing to create a hemi-marsh, moist-soil management; Gray et al. 2013) is a way for managers to provide desired habitat conditions for waterbirds during critical periods (Kaminski et al. 2009, Smith et al. 2012). More intensively managed or actively managed wetlands may produce suitable habitat for multiple waterbird guilds through hydrological manipulation, even in drought years (Pankau 2008, Smith et al. 2012). Similarly, Kaminski et al. (2009) found that waterbird use of managed wetlands was significantly greater than unmanaged wetlands because active hydrological management increased available habitat. Therefore, intensity of wetland management may increase the amount of suitable habitat available to waterbirds, especially waterfowl.

Highly connected wetlands and groups of wetlands of varying types in close proximity are known as wetland complexes. Wetland complexes support various waterbird species and increase waterbird diversity by supplying varied microhabitats and resources to meet dynamic

physiological needs such areas for forage (Dwyer et al. 1979, Brown and Dinsmore 1986, Weller 1988, Fairbairn and Dinsmore 2001, Pearse et al. 2012, Beatty et al. 2014b). Proximity to varying wetland types increases the chances that suitable habitat will be present (Dwyer et al. 1979). On the other hand, wetland connectivity to river systems may have variable impacts on waterbird habitat. Stafford et al. (2010) found that submerged and floating-leaved aquatic vegetation, an important food source for many waterbird species, has decreased substantially in the Illinois River Valley due to connectivity to the Illinois River that has undergone extensive degradation from sedimentation, highly fluctuating hydrology, and exotic species (e.g., exotic carp; Havera 1999). However, wetlands associated with large river systems with periodic flooding could provide more shorebird habitat than isolated palustrine and lacustrine wetlands (Smith et al. 2012).

Habitat complexity (Weller and Spatcher 1965) and interspersions of disparate vegetation types (Ringelman and Longcore 1982, Baschuk et al. 2012) are also important components of waterbird habitat. Avian diversity and abundance are often greatest in 50:50 water to vegetation ratios, known as “hemi-marsh” (Weller and Spatcher 1965, Kaminski and Prince 1981, Smith et al. 2004). Both hemi-marsh and patches of other vegetation types are important for isolating waterfowl pairs during spring and for predator avoidance during brood-rearing and molting periods (Murkin et al. 1982). Additionally, aquatic invertebrate abundance, an integral component of waterbird habitat suitability, is often greatest in areas of interspersions of open water (in dense beds of submerged aquatic vegetation) and emergent vegetation (Voigts 1976) with a mixture of habitat types (Murkin et al. 1992).

Surrounding anthropogenic land use and disturbance impacts wetlands (Zedler and Kercher 2005, Mack 2006) and waterbirds (Havera et al. 1992, Peterson and Niemi 2007). For instance, extensive leveeing and drainage has eliminated over 50% of the natural wetlands in the Illinois River Valley, and extant wetlands are further degraded by sedimentation, exotic species, and eutrophication from agricultural run-off (Havera 1999). While studying the impact development on wetlands and wetland-dependent birds, Ward et al. (2010) found that increased development within 2 km of wetlands was associated with drastic changes to wetland vegetation structure: wetlands tended to transform into unvegetated, open ponds or dense monocultures of vegetation, both less suitable for many wetland birds. Furthermore, Beatty et al. (2014a,b)

demonstrated that mallards shift wetland use in response to anthropogenic disturbances, site proximity to protected areas and sanctuaries, and proximity to emergent or woody wetlands (i.e., within specified buffer).

Wetland hydrology is an integral part of a wetland system (Mitsch and Gosselink 2015) that dictates the availability and suitability for waterbirds (e.g., foraging habitat, water depth; Isola et al. 2000, Conway et al. 2005) and subsequent vegetation communities and functions within a wetland (Wilcox and Meeker 1999). For instance, the decline in hemi-marsh conditions in Ward et al.'s (2010) study, were likely due to changes in natural wetland hydrology from surrounding development. Furthermore, water depth influences the accessibility of available foraging habitat directly because waterbird morphology and size limit where waterbirds can forage (Isola et al. 2000, Davis and Smith 2001, Ma et al. 2010, Collazo et al. 2016). A wetland's water regime describes the timing and permanence of inundation, such that areas with more permanent water regimes (i.e., permanent, intermittently exposed and semi-permanent) are generally deeper and inundated for longer periods of time than areas with less permanent water regimes (i.e., seasonal, temporary, saturated; Cowardin et al 1979, Johnson et al. 2010). Furthermore, wetland hydrology, availability, and suitability are tightly linked to climate (e.g., May pond counts; USFWS 2017). Wetland hydrology and climate, though, vary between seasons and from year to year (Niemuth et al. 2010). Due to this variability, the amount of suitable habitat for waterbirds is stochastic and difficult to ascertain.

I identified factors that influenced the availability of suitable waterbird habitat and determined the extent and spatial distribution of wetlands capable of supporting focal guilds of migratory birds during critical periods in Illinois to guide future conservation planning efforts. I examined the presence and abundance of vegetation and water inundation characteristics in wetlands across Illinois for three waterbird guilds during periods that could have population-level demographic effects. I assumed these critical periods included 1) spring migration for dabbling ducks, 2) spring migration and summer breeding for marsh birds, and 3) spring and autumn migration for shorebirds. I examined the effects of wetland management, wetland connectivity and complexity, surrounding land use and disturbance, water regime, and climate on the presence and abundance of habitat characteristics deemed important for guilds to meet life history requirements. A statewide habitat assessment of this magnitude has not been previously

conducted for waterbirds, and temporally-specific estimates of waterbird habitat will inform the Illinois Wetlands Campaign of the Illinois Comprehensive Wildlife Conservation Plan and Strategy and UMRGLR Joint Venture for continued management and conservation for waterbirds.

2.3 MATERIALS AND METHODS

Study area

I studied wetlands identified by the National Wetlands Inventory (NWI) across Illinois on both public and private land. The study spanned 14 Natural Divisions throughout Illinois, which are distinct ecoregions differentiated by topography, soils, glacial history, flora, and fauna (Figure 2.1; Schwegman 1973). I excluded the Lake Michigan Natural Division based on assumptions that changes in inundation and habitat suitability over time were minimal and habitat availability was likely captured by NWI. Despite human alteration and loss of >85% of its wetlands (Dahl 1990, 2006), Illinois continues to support large populations of migratory waterbirds as well as some breeding populations. Illinois lies within the heart of the Mississippi Flyway, with breeding grounds primarily to the north and wintering grounds to the south for most species of wetland-dependent waterbirds (Havera 1999).

In Illinois, average monthly precipitation between February and April 2016 – 2017, the spring survey period, was 6.8 cm and 9.4 cm respectively, and the Palmer Hydrological Drought Index (PHDI) values ranged from 1.37 – 3.13 indicating wetter conditions than average (NCDC 2010). During the summer survey period, between April and June 2016 – 2017 in Illinois, average monthly precipitation was 9.1 cm and 8.4 cm respectively, and the PHDI values ranged from 0.66 – 2.17 indicating wetter conditions than average (NCDC 2010). Average monthly precipitation in the autumn survey period, between August and September 2016 – 2017, in Illinois was 7.9 cm and 7.2 cm respectively, and the PHDI values ranged from 1.2 – 1.68 indicating wetter conditions than average (NCDC 2010).

Site selection

I surveyed wetland habitat conditions for three guilds of wetland-dependent birds—dabbling ducks, secretive marsh birds, and shorebirds—during 2015–2017. I conducted surveys

during critical periods when I hypothesized these guilds are most limited by habitat in this region. Survey periods included 1) spring for northerly migrating dabbling ducks (mid-February–mid-April), 2) summer for migrating and breeding secretive marsh birds and other waterbirds, (mid-April–mid June) and 3) autumn for southerly migrating shorebirds (late July–early September). Annually, I selected 100 – 120 25-ha plots overlapping NWI polygons and visited each plot once in each of the three seasons (i.e., 300 – 360 site visits annually; Figure 2.2). I consolidated NWI wetland polygons into 6 classes according to major wetland cover types (i.e., emergent, forested, lake, pond, riverine and other; Cowardin et al. 1979; Table 2.1), and I omitted any polygons < 0.5 ha. I generated plots randomly using a spatially-balanced stratification (Theobald et al. 2007) based on consolidated NWI wetland classes and Natural Divisions. Specifically, I created a wetland density map for the state using 10-m cell sizes and a search window of 0.25-km × 0.25-km. I excluded NWI wetland polygons <0.35 km from a Natural Division border to ensure plots were entirely placed within a distinct Natural Division. I then converted the density map to an inclusion probability map, where the highest density was one and the lowest density was zero for the entire state of Illinois. Using the Reversed Randomized Quadrant-Recursive Raster tool (RRQRR) in ArcGIS 10.4 (ESRI 2011), I created an overrun of two times the target number of plots overlapping consolidated NWI polygons in a spatially balanced design using Natural Divisions as strata (Theobald et al. 2007). Excess plots served as secondary plots if primary sites were deemed unsamplable (see below). I selected survey plots with equal inclusion probabilities for the six NWI wetland classes. Then, I used Neyman allocation method to assign survey effort proportionate to NWI wetland area within Natural Divisions ensuring ≥ 3 plots per Natural Division (Neyman 1934). This sampling design with unequal effort among strata allowed for unbiased estimation (Johnson et al. 2009) and was flexible to uncertainty in removing and adding sites (e.g., denied access by landowner; Theobald et al. 2007). Furthermore, this process has been shown to generate results representative of the larger population of unsampled wetlands (Miller 2016, Tozer et al. 2018). Following the spring season, I randomly replaced half of the plots containing predominantly forested polygons (i.e., 50 –100% cover) with plots with emergent polygons to better reflect habitat used by focal species for summer (i.e., marsh birds) and autumn (i.e., shorebirds), which often avoid forested wetlands (Johnson et al. 2009, Bolenbaugh et al. 2011).

I requested permission to access the plots from private landowners using informational letters which included a postcard and return postage for their response. If a plot was inaccessible because of denied landowner permission or safety concerns, I attempted to shift the plot <500 m from the original plot's bounds. However, if accessibility could not be resolved through shifting, I replaced the plot with another randomly generated plot (~37% of inaccessible plots). I attempted to randomize plot visitation order within each season; however, I visited plots near one another on the same day or trip to increase efficiency.

For each consolidated NWI wetland polygon present within a plot, I mapped vegetation and surface water coverage using a field PC (Archer Models 1 and 2, Juniper Systems Inc., Logan, UT, USA) with a GPS on foot or by boat. Inundated areas, as well as areas classified as mud or patchy inundation were separated, and I distinctly mapped vegetation communities (e.g., emergent/herbaceous, woody, etc., Table 2.3). Vegetation and inundation patches were mapped if they were at least 6 m wide or $\geq 36 \text{ m}^2$, which is the resolution of Synthetic Aperture Radar (SAR) satellite imagery used by a companion study. To further measure important waterbird vegetation and other wetland cover, I visually estimated total flooded area and vegetation cover within each NWI class present in the plot at each seasonal visit. I waded or boated inundated areas to assess depth profile in at least 10 locations to evaluate the proportion of the wetland polygon that was shallowly inundated (i.e., <45 cm for dabbling ducks and <10 cm for shorebirds; Table 2.2). Next, I estimated areal cover of the inundated area of emergent (dense persistent and non-persistent), submersed and floating-leaved aquatic vegetation (SAV-FLAV), scrub-shrub (woody vegetation <6 m), and forested (woody vegetation >6 m) vegetation. I estimated the inundated area covered by non-rooted FLAV such as *Lemna* spp. as well as the area that was open water and un-vegetated. Then, I approximated the area of the polygon that was not inundated and proportion of exposed mudflats (<30% vegetation), and in 2017 only, I estimated total shorebird foraging habitat (i.e., mudflats and inundation <10 cm; Table 2.2).

For each NWI wetland polygon, I noted evidence of active or passive wetland management including hydrological modification, levees, mowing or disking, moist-soil plant management for waterfowl, and water-control structures (e.g., culverts, stop-log structures; Table 2.4). Given the presence of these characteristics, I assigned a relative management intensity on an ordinal scale (e.g., none, low, medium, and high; Table 2.4). This scale ranged from

unmanaged bottomland forests to highly managed impoundments with water control structures, water delivery and removal systems, and food plots specifically for waterfowl.

While at a plot, I studied the plot map, and determined based on wetlands present during observations, whether each NWI wetland polygon was within a wetland complex (Table 2.4). Here, I defined a wetland complex as a consolidated NWI polygon that was proximal to other wetland types. Additionally, I scored each NWI wetland polygon's connectivity to rivers or streams on an ordinal scale ranging from 0 – 7 (no connectivity to permanent connection; Table 2.4). These scores were based upon observations of hydrological characteristics (i.e., secondary indicators).

I assessed wetland complexity both qualitatively and quantitatively. First, I visually estimated horizontal interspersion (i.e., complexity) of NWI classes present at a site noting the number of distinct patches and the degree to which they were interspersed (Mack 2001, US EPA 2011 [Patch Mosaic Complexity]; Figure 2.3). Specifically, I evaluated each NWI wetland polygon from a “plan view” (i.e., as if looking down from above). Using tracks and visual observation, I scored each NWI class's horizontal interspersion on an ordinal scale ranging from 0 to 4, where 0 indicates a monotypic or not interspersed area, and 4 indicates a highly interspersed area with many vegetation types (Figure 2.3, Table 2.4). Additionally, I calculated edge density (ED), a quantitative measure of the interspersion of vegetation and water (i.e., open and aquatic bed) (m/ha) (Rehm and Baldassarre 2007). I first created thematic cover maps by digitizing inundation and vegetation cover in ArcGIS using the recorded tracks and fieldnotes. I differentiated inundation and cover polygons at a site (e.g., Figure 2.4) and did this separately for each seasonal visit. Next, I clipped cover polygons by NWI wetland polygons to exclude non-NWI area, dissolved by cover (e.g., open, short herbaceous, etc.) and inundation type (e.g., inundated, dry, etc.) by NWI class, and converted cover maps to raster format (1-m resolution). I used the Spatial Statistics by Patch Analyst extension (i.e., the Fragstats interface) in ArcGIS 10.4 (Rempel et al. 2012) to calculate ED of the cover maps by taking the total length of edge in meters in a given NWI class and dividing it by the total area of the NWI class to get a density of meters of edge per hectare. This standardized metric was then comparable to other sites and other NWI classes at a given site due to its quantitative nature (i.e., edge increases in direct proportion to mixing; McGarigal et al 2002).

I used a modified version of the Ohio Rapid Assessment Method (ORAM) during 2016 – 2017 (Version 5.0; Mack 2001). With an emphasis on anthropogenic disturbance, the ORAM scores wetland integrity using six metrics including surrounding land use, hydrology, habitat alteration and development, interspersed, and vegetation communities (Table 2.5). While ORAM has both a narrative and quantitative section, I only used the quantitative portion, as the narrative is specific to Ohio (Mack 2001). I scored wetland polygons using a subset of the ORAM metrics and submetrics: 2b, 3a-e, 4a-c, 6a-d (Mack 2001; Table 2.5) on a gradient similar to ORAM’s method of scoring wetlands along a quality gradient for regulatory purposes. I collected data and calculated a modified ORAM score at the NWI wetland class level.

I incorporated potential impacts of surrounding landscape by calculating the Landscape Development Intensity Index (LDI; Brown and Vivas 2005) in ArcGIS. The LDI represents a measure of human disturbance on a gradient relative to intensity of land use (Brown and Vivas 2005, Mack 2006). Low LDI values (≤ 2) represent land uses that are more natural (e.g., wetlands, grasslands, forests), whereas high LDI values (>10) represent highly altered/disturbed surrounding land uses (e.g., row-crop agriculture, urbanization). I computed the LDI using the 2011 National Land Cover Dataset (NLCD; Fry et al. 2011) at two spatial scales – a local 100-m and landscape-level 5-km buffer (e.g., mallard foraging flight distance; Webb et al. 2010; Table 2.4). I calculated the proportion of disparate land uses surrounding sites and assigned LDI coefficients based on intensity of the land uses following recommendations by Mack (2006). I used the LDI values for the two spatial scales, 100-m and 5-km, along with the proportion of the landscape composed of wetlands (i.e., both emergent and woody) as classified by NLCD in my analysis (e.g., Tozer et al. 2010, Beatty et al. 2014b; Table 2.4).

To determine the influence of water regime and climate on the presence and abundance of cover and inundation considered suitable for waterbirds, I first extracted the classified water regime from the NWI polygons present at a given plot. The NWI describes water regime using available aerial imagery and soil data (Tiner 1997). This parameter is difficult to describe accurately (Cowardin and Golet 1995) especially with a single site visit during a given season or year. Thus, I included NWI water regime modifiers understanding they are imperfect and broad categorizations (Table 2.4). Observed non-tidal water regime modifiers in decreasing permanence of surface water included permanently flooded, intermittently exposed, semi-

permanently flooded, seasonally flooded, temporarily flooded, and saturated (Cowardin et al. 1979). I extracted the weekly Palmer's Hydrological Drought Indices (PHDI) for the week and region of the site visits from the National Climate Data Center (NCDC 2010), to assess the impact of climate on wetland suitability and accessibility for waterbirds. The PHDI was developed to index long-term moisture supply. The PHDI also includes precipitation, evapotranspiration, and soil moisture (NCDC 2010) information and reflects the hydrological impacts of drought (Liu 2011). A PHDI value greater than zero indicates conditions wetter than average, whereas a value less than zero indicates conditions drier than average (Table 2.4).

Statistical Analyses

To evaluate the extent of suitable wetlands and deepwater areas for waterbirds, I selected aforementioned cover and inundation metrics (Table 2.2) as response variables and formulated biologically plausible model combinations using potential predictor variables (Table 2.4). For each NWI wetland polygon within each seasonal visit, I calculated the proportion that met aforementioned "suitability" standards for the waterbird guild of interest. For spring migrating dabbling ducks, this included the proportion of each NWI wetland polygon that was: 1) shallowly inundated (i.e. <45cm) and, 2) flooded forest regardless of depth. For late spring migrating and breeding marsh birds and other waterbirds, I calculated the proportion of each NWI wetland polygon containing: 1) flooded, dense persistent emergent vegetation (PEM; e.g., *Typha* spp.) and, 2) flooded non-persistent emergent vegetation. Finally, for autumn migrating shorebirds, I calculated the proportion of each NWI wetland polygon that was: 1) mudflat and, 2) mudflat and very shallowly inundated (i.e., < 10cm). I was unable to include the cover of hemi-marsh for spring migrating dabbling ducks and breeding marsh birds as it was infrequently encountered hindering its inclusion in suitability models.

I conducted all statistical analyses in Program R Version 3.4.2 (R Core Team 2017). Prior to building model sets, I checked predictor variables for collinearity and excluded highly correlated variables ($r > |0.5|$) to avoid multicollinearity and associated bias in the parameter estimates and inflation of standard errors (Zuur et al. 2010). I plotted raw predictor and response variables and determined that modeling interactions was unnecessary (Zuur et al. 2010). I examined data for outliers in both the predictor and response variables using box plots and

Cleveland dotplots (Zuur et al. 2010). I omitted identified outliers likely due to observer errors or unknown bias. To check for homogeneity of variance, I plotted the residuals versus fitted values (Gotelli and Ellison 2013). Heterogeneity of variances was observed due primarily to copious zeros in the response variables, thus causing a large right skew and unbalanced (i.e., cone-shaped) residual plots (Zuur et al. 2009). Therefore, I used a conditional model approach (Fletcher et al. 2005). For each analysis, I created two datasets: one with a binomial response variable indicating presence or absence of the suitability metric within each NWI wetland polygon at a site, and a second, truncated dataset, including the proportion of each NWI polygon containing the suitability metric where present (i.e., >0%). I then analyzed data separately using candidate model sets with a logistic and linear regressions, respectively. More than 90% of NWI polygons contained shallow inundation (< 45 cm) for spring migrating dabbling ducks, thus I used ordinary regression instead of the conditional approach. I included only predictor variables and variable combinations with strong biological reasoning from the literature and a priori evidence to avoid over parameterization (Burnham and Anderson 2002). I grouped variables representing primary biological hypotheses into four model sets: wetland management, wetland connectivity/complexity, surrounding land use and disturbance, and water regime and climate (Table 2.4). I included all combinations of additive effects within each model set. To account for inherent differences among Natural Divisions and NWI classes and to scale-up model estimates to the NWI and Natural Division level, I included these variables in all model sets. Thus, my null or base model included NWI class and Natural Division.

I modeled the probability that an NWI polygon contained suitable habitat (Table 2.2) with logistic regression using the ‘glm’ function (package stats in program R; R Core Team 2017). To determine relative model fit, I examined AIC_c scores as well as null and residual deviance, where null deviance indicated the response predicted by a null model (i.e., only an intercept), and residual deviance indicated the response predicted by a model with independent predictor variables. I calculated McFadden’s pseudo R^2 for top models to ascertain a model’s predictive ability (i.e., the closer to 1 the better a model predicts 0’s and 1’s correctly; McFadden 1974). I checked for overdispersion (i.e., more variation than expected given the data) by dividing the null deviance by degrees of freedom ($\chi^2/df=(\hat{c})$). For the logistic model sets, (\hat{c}) was <1.4 in all cases and often < 1, indicating less variation than expected, so I did not adjust models for overdispersion.

To model the proportion of an NWI wetland polygon containing suitable waterbird habitat, I chose beta regression because of its suitability to account for the continuous but restricted (i.e., 0, 1) nature of the proportional response variable (i.e., proportion of NWI class of with a suitable cover). Estimation via beta regression is performed by maximum likelihood, and an additional precision parameter, Φ , is included. Beta regression is useful for modeling proportions (Ferrari and Cribari-Neto 2004; Smithson and Verkiulen 2006), as it can deal with heteroskedasticity or skewness common to proportional data (Ferrari and Cribari-Neto 2004). Beta regression employs two link functions, one for the parameter μ and one for the precision parameter Φ , which link the linear predictors with the observations (Smithson and Verkiulen 2006). Thus, beta regression is, essentially, a generalization of linear regression when the dependent variable is a proportion. If an NWI wetland polygon contained 100% of the suitability metric, I transformed the response variable to truncate the data for analysis (i.e., $Y \leq 1$) (Smithson and Verkiulen 2006). I conducted beta regression (package `betareg` in R; Cribari-Neto and Zeileis 2019) using the same candidate model sets used in the logistic regression step. To assess fit of the beta regression models, I plotted standardized residuals versus fitted values and observed them for detectable patterns (Ferrari Cribari-Neto 2004). Additionally, I obtained a measure of explained variation by computing the pseudo R^2 , (R_p^2), defined as the square of the sample correlation coefficient between \hat{n} and $g(y)$ (Ferrari and Cribari-Neto 2004).

I ranked and compared models using Akaike's Information Criterion corrected for small sample size (AIC_c ; Burnham and Anderson 2002). I calculated model weight (w_i) and considered models within $2 \Delta AIC_c$ of the top model to be competitive (Arnold 2010; Burnham et al. 2011). Then, I compared top models across model sets (i.e., wetland management, wetland connectivity/complexity, surrounding land use/disturbance, and water regime /climate) to determine relative variable importance by examining model weights (w_i). I compared AIC_c score ranks to determine important variables in determining the probability of a given habitat metric (i.e., the logistic portion) and the cover of the metric (i.e., beta regression portion). After examining top models and competing models (i.e., $<2 \Delta AIC_c$), I determined there were uninformative parameters included in subsequent models and that most competing models only differed by one parameter (Arnold 2010). I followed recommendations by Arnold (2010) by

opting to discard these models and continue with the top model in each set, as model averaging did not alter estimates substantially.

Using the top model estimates across response variables and model sets, I combined the logistic and ordinary (i.e., beta) regression model via equations (1 – 3) and conducted parametric bootstrapping to develop 85% confidence intervals (i.e., conditional model; Fletcher et al. 2005) (Arnold 2010, R Core Team 2017). I produced 85% confidence intervals for the conditional estimates using equation (1) with 10,000 bootstrapped samples. I randomly generated values of $\hat{E}(Y)$ by resampling β and θ , using equations (2) and (4):

$$\hat{E}(Y) = \hat{\pi}\hat{\mu} \quad (1)$$

where

$$\hat{\pi} = \exp(x'\hat{\beta}) / \{1 + \exp(x'\hat{\beta})\} \quad (2)$$

and

$$\hat{\mu} = \exp(w'\hat{\theta}) / \{1 + \exp(w'\hat{\theta})\} \quad (3)$$

where $\hat{\beta}$ is a vector of estimates of the coefficients from the logistic regression model, and x' is the corresponding vector of predictor variables. Similarly, $\hat{\theta}$, is a vector of estimates, and w' is the vector of predictor variables for the beta regression model (Fletcher et al. 2005).

To estimate the total acreage of NWI wetlands across Illinois with suitable waterbird habitat, I extracted beta estimates from my base logistic- and beta-regression models (i.e., Natural Division and NWI class), created a matrix of all wetland polygons from the NWI layer (excluding NWI polygons <0.5 ha), and calculated area estimates by multiplying the total area (ha) of each NWI wetland polygon by conditional estimates. To account for variation within conditional estimates, I used non-parametric bootstrapping (2,000 iterations) to randomly generate conditional area estimates for each NWI polygon and summed total area across all polygons within each NWI class, Natural Division, and for the state of Illinois. I produced 85% confidence intervals of the estimate by taking the 7.5 and 92.5 percentiles from the bootstrap iterations (Arnold 2010).

2.4 RESULTS

I assessed 2015 and 2016 – 2017 data separately due to differences in metrics assessed.

Migrating Dabbling Ducks

During the spring season (21 February – 14 April, 2016; 12 February – 12 April, 2017), I visited 233 plots and surveyed 4,093.0 ha of NWI wetlands for the presence and coverage of shallow water and inundated forest. Of the NWI wetland polygon classes, forested was the most surveyed, accounting for 55% of the area surveyed, followed by lake (20%), emergent (15%), pond (5%), riverine (5%), and other (<1%). Via visual estimation, inundation was 51.8% (46.9 – 56.7%; 85% CI) in emergent and 32.6% (29.6 – 35.6%) in forested classes (Table 2.6). Inundation was greater (>80%) in pond, lake, and riverine (Table 2.6).

The emergent NWI class had the greatest raw proportion (37.4% [33.5 – 41.5%]) of shallow inundation followed by pond (30.5% [26.1 – 34.9%]; Table 2.6). The top model predicting the coverage of shallow water in the cover and complexity model set outperformed top models of the other sets (i.e., $\Delta AIC_c = 19.0$ and $w_i = 0.99$; Table 2.7). The best supported model included horizontal interspersed and wetland complex in addition to the base model with Natural Division and NWI class (Pseudo $R^2 = 22\%$; Table 2.8). As horizontal interspersed increased, so too did the estimated cover of shallow inundation, and a wetland located within a wetland complex had a greater proportion of shallow inundation (Figure 2.6, Table 2.9). For instance, an emergent NWI polygon with an increased horizontal interspersed of 4 had an estimated 43.5% (35.9 – 51.4%) cover of shallow inundation, whereas an emergent polygon with horizontal interspersed of 0 had an estimated 23.3% (16.5 – 31.8%) cover of shallow inundation (Table 2.9). Similarly, an emergent wetland located within wetland complex had a 44.1% (23.7 – 67.8%) greater proportion of shallow inundation than an emergent wetland not located within a wetland complex (Figure 2.6, Table 2.9).

The mean predicted coverage of shallow inundation was 181,882 ha (179,824 – 184,068 ha) accounting for 29% of NWI wetland polygon area in Illinois included in this study (i.e., omitting polygons <0.5 ha; Table 2.12). Forest NWI polygons had the greatest areal cover of shallow water with 81,508 ha (81,046 – 82,768 ha), whereas emergent polygons had the greatest

proportion of shallow inundation (37%; Table 2.10, 2.11). Among Natural Divisions, the Northeastern Morainal had the greatest coverage of shallow water at 44%, but the Southern Till Plain Natural Division had the greatest areal coverage with 42,225 ha (41,583 – 42,883 ha; Table 2.12, 2.13)

The forested NWI class had the greatest raw proportion of inundated forest with 24.1% (23.7 – 24.5%), followed by emergent at 11.4% (10.9 – 11.9%) and riverine at 10.1% (8.9 – 11.3%; Table 2.6). The top model for predicting shallow inundation in the cover and complexity model set outperformed top models of the other model sets for both the logistic and beta regression (i.e., $\Delta AIC_c = 11.5$ and $w_i = 0.99$ [logistic] and $\Delta AIC_c = 5.7$ and $w_i = 0.95$ [beta regression]; Table 2.15). The best-supported model for the logistic portion included horizontal interspersions and wetland complex (McFadden's $R^2 = 22\%$), whereas the best supported model for the beta regression portion included wetland complex and edge density (Pseudo $R^2 = 21\%$; Table 2.16). An NWI polygon located within a wetland complex was 1.9 (1.0 – 3.3) times more likely to contain inundated forest (Figure 2.8, Table 2.17). Inundated forest cover was negatively related to edge density, and when inundated forest was present, it declined 9.5% (4.2 – 14.6%) for every 100 m/ha increase in edge density (Figure 2.9). As horizontal interspersions increased to 4, the odds that inundated forest was present in an NWI polygon was 2.4 (0.6 – 6.0) times greater than for a monotypic wetland with horizontal interspersions 0 (Figure 2.7, Table 2.17). Accordingly, forested NWI polygons were 2.4 (0.5 – 2.9) times more likely to contain inundated forest, and, when present, had 79.3% (46.2 – 120%) greater inundated forested cover than emergent polygons (Table 2.17).

Estimated coverage of inundated forest across Illinois was 125,832 ha (119,257 – 132,608) during spring accounting for approximately 20% of NWI wetland polygon area in Illinois included in this study (Table 2.12). The forest NWI class had the greatest areal and percent cover of inundated forest with 81,912 ha (81,047 – 82,769 ha) and 25% of forested polygon cover (Table 2.10, 2.11). Among Natural Divisions, the Southern Till Plain had the greatest areal cover with 30,472 ha (29,424 – 31,308 ha), and the Coastal Plain had the greatest percent cover of inundated forest with 36% cover (Table 2.13, 2.14).

Migrating and Breeding Marsh Birds

During the summer survey season (12 April – 4 June, 2016; 17 April – 9 June, 2017), I visited 248 plots and surveyed 4,558.7 ha of NWI wetland polygons. Of the consolidated NWI classes, forested accounted for 45% of the area surveyed, followed by emergent (25%), lake (19%), pond (5%), riverine (5%), and other (<1%). Average inundation in emergent and forested polygons was 58.5% (54.3 – 62.7%) and 46.1% (42.5 – 49.7%) respectively, whereas inundation in the more permanent classes of pond, lake, and riverine was >85% (Table 2.6).

Average cover of inundated dense PEM (e.g., *Typha* spp.) was greatest in emergent polygons at 8.5% (6.4 – 10.6%) followed by pond (5.5% [2.9 -8.1%]; Table 2.6). The wetland connectivity/complexity model set ranked the highest for the logistic portion (i.e., $\Delta AIC_c = 40.5$ and $w_i = 0.99$, McFadden's $R^2 = 27\%$; Table 2.18). However, in the beta regression, the null model including Natural Division and NWI class ranked highest in all model sets indicating insufficient variables to predict the cover of inundated PEM given presence (Pseudo $R^2 = 34\%$; Table 2.19). The best-supported model for the logistic portion included horizontal interspersions, connectivity to rivers, and wetland complex (Table 2.19). Wetland classes that were disconnected from rivers had the greatest predicted proportion of inundated dense PEM, with an overall decreasing trend as connectivity increased (Figure 2.10). For instance, the odds of a wetland containing inundated dense PEM was 44.1 (17.2 – 111.3) times less likely with a river connectivity of 7 than for a wetland with no river connection (i.e., 0; Table 2.20). Similarly, NWI polygons within a wetland complex were 97.4% more likely to have inundated PEM than those not associated with wetland complexes (Figure 2.11, Table 2.20), and as horizontal interspersions increased to 4, an NWI polygon was 14.4 (4.5 – 42.5) times more likely to have inundated dense PEM compared to a wetland with horizontal interspersions of 0 (Figure 2.11, Table 2.20).

Estimated coverage of inundated PEM across Illinois was 29,702 hectares (26,512 – 34,487 ha) accounting for 5% of Illinois NWI polygon area included in this study (Table 2.12). Not surprisingly, emergent classes had the greatest areal cover and percent cover of PEM with 11,179 ha (10,951 – 11,395 ha) and 16% areal cover (Table 2.10, 2.11). The Northeastern

Morainal Natural Division had the greatest areal and percent cover of PEM with 12,644 ha (11,939 – 13,334 ha) and 23% cover (Table 2.13, 2.14).

Average cover of inundated non-persistent emergent vegetation (e.g., moist-soil vegetation; NPE) was greatest (17.4 % [15.1 – 19.7%]) in emergent classes (Table 2.6). Again, the top model in the cover and complexity model set ($w_i = 1.0$ [logistic], $w_i = 0.99$ [beta regression]) outperformed top models of the other model sets for both the logistic and beta regression (i.e., $\Delta AIC_c \geq 52.2$ [logistic] and $\Delta AIC_c \geq 25.5$ [beta regression]; Table 2.21). The best-supported model for the logistic portion included horizontal interspersion, edge density, and wetland complex (McFadden's $R^2=22\%$), whereas the top model for the beta regression was the same but omitted the variable edge density (Pseudo $R^2=27\%$; Table 2.22). As horizontal interspersion increased, so too did the probability of and predicted proportion of inundated NPE (Figure 2.13, Table 2.23). For instance, when an NWI class had a horizontal interspersion value of 4, it was 16.6 (7.6 – 34.8) times more likely to have inundated NPE than an NWI class with horizontal interspersion of 0 (Figure 2.13, Table 2.23). Furthermore, the proportion of inundated NPE, when present, increased 15.6% (2.4 – 30.5%) for every 100 m/ha increase in edge density (Figure 2.15), and NWI polygons located within a wetland complex had greater predicted cover of inundated NPE (Figure 2.14, Table 2.23). For example, the odds that an NWI polygon contained inundated NPE was 50% (3.5 – 117.5%) greater if the polygon was within a wetland complex, and when present, the proportion of inundated NPE was 22.2% (32.4 – 52.1%) greater (Table 2.23).

The estimated mean coverage of inundated NPE across Illinois was 56,187 hectares (51,376 – 61,646 ha). The forest NWI class had the greatest areal cover with 26,421 ha (25,871 – 26,953 ha), whereas, the emergent NWI class had the greatest percent cover with 15% (Table 2.11, 2.12). Of Natural Divisions, the Southern Till Plain had the greatest areal cover of NPE with 10,886 ha (10,201 – 12,206 ha), but the Lower Mississippi River Bottomlands had the greatest percent cover with 13% (Table 2.13, 2.14).

Migrating Shorebirds

During the autumn survey season (25 July – 14 September, 2016; 20 July – 13 September, 2017), I visited 245 plots and surveyed 4,557.7 ha of NWI wetlands. Of the

consolidated NWI classes, forested was the most surveyed class, accounting for 45% of the area surveyed, followed by emergent (25%), lake (20%), pond (5%), riverine (5%), and other (<1%). Average inundation was less than spring and summer survey periods, and average inundation in emergent and forested classes was 33.9% (29.7 – 38.1%) and 21.8% (18.9 – 24.7%) respectively (Table 2.6). Further, inundation in the more permanent classes pond, lake, and riverine averaged >65% (Table 2.6).

I included the variable season in shorebird models to provide information on whether northward (i.e., summer survey period) migrating or southward (i.e., autumn survey period) migrating shorebirds are more limited by habitat in Illinois. Besides the lake class (average cover of 10% [9.9 – 10.1%]), the average cover of exposed mudflats for migrating shorebirds was low for all NWI classes (i.e., <3%; Table 2.6) in both summer and autumn. The top model in the cover and complexity model set again outperformed top models from other sets for the logistic regression (i.e., $\Delta AIC_c = 85.6$ and $w_i = 1$; Table 2.24); however, there was less certainty in the beta regression, where the top model from the land use/disturbance model set had similar support and was considered competing (i.e., $\Delta AIC_c = 0.19$ and $w_i = 0.35$; Table 2.24). The best supported model for the logistic regression included wetland complex in addition to the base variables Natural Division, NWI class, and season (McFadden's $R^2 = 18\%$; Table 2.25). Moreover, the top model for the beta regression included horizontal interspersions and river connectivity in addition to the base variables (Pseudo $R^2 = 21\%$; Table 2.25). The NWI polygons that were intermediately connected (i.e., 4) and permanently connected (i.e., 7) to rivers and streams had the greatest mudflat cover (Figure 2.16, Table 2.26). For instance, if an NWI class was intermediately connected or permanently connected to rivers and streams, the odds of it containing mudflat was 3.0 (1.4 – 5.5) and 2.8 (1.4 – 5.2) respectively than a wetland with no river connectivity (i.e., 0; Table 2.26). However, the difference in area is very small (<0.05 difference in proportion of mudflats; Figure 2.16). When mudflats were present, an NWI polygon within a wetland complex had 24.9% (7.0 – 45.9%) greater predicted mudflat cover (Table 2.26). The probability of mudflats increased with horizontal interspersions, and an NWI polygon with high interspersions of 4 was 7.6 (4.4 – 12.7) times more likely to have mudflats compared to an NWI polygon with interspersions of 0 (Table 2.26). While the odds of encountering mudflats was 13.5% (-6.0 – 29.4%) lower in the summer survey period than the

autumn period, when mudflats were present, there was 51.5% (31.7 – 74.4%) greater coverage in summer than autumn (Table 2.26).

Estimated mudflat coverage during summer was 36,812 ha (30,403 to 43,513 ha), accounting for 6% of NWI polygon area in Illinois, whereas the mean estimated mudflat cover during autumn was 25,324 ha (20,734 to 30,177 ha), accounting for 4% of NWI polygon area in Illinois (Table 2.10, 2.12). During summer, the lake NWI class had the greatest areal and percent cover of mudflats with 14,965 ha (9,747 – 20,446 ha) and 10% cover (Table 2.10, 2.11), and the Major Water Bodies Natural Division had the greatest areal and percent cover with 11,855 ha (10,525 – 16,394 ha) and 14% cover (Table 2.13, 2.14). During autumn, the lake NWI class again had the greatest areal cover with 10,482 ha (9,357 – 10,623 ha), but the riverine had the greatest percent cover with 9% (Table 2.10, 2.11). The Major Water Bodies Natural Division again had the greatest areal and percent cover of mudflats with 8,265 ha (5,754 – 10,448 ha) and 10% (Table 2.13, 2.14).

Average cover of shorebird foraging habitat (i.e., mudflats and shallow inundation <10 cm, only measured in 2017) composed a small portion of NWI classes (<3.2%) during summer. In autumn, the average cover of shorebird foraging habitat was greater, and in the lake class, an average cover of 7% (6.7 – 7.3%) was observed. The top model for predicting shorebird habitat in the cover and complexity model set outperformed top models from other sets for the logistic regression ($\Delta AIC_c = 12.54$ and $w_i = 0.996$; Table 2.24); however, the top model from the land use/disturbance model set outperformed the top models from the other model sets ($\Delta AIC_c = 3.66$ and $w_i = 0.82$; Table 2.24). The best supported model for the logistic portion included edge density, horizontal interspersion, and wetland complex in addition to the base variables Natural Division, NWI class, and season (McFadden's $R^2 = 22\%$), whereas the best supported model for beta regression included the proportion of surrounding 5-km buffer composed of wetlands in addition to the base variables (Pseudo $R^2 = 27\%$; Table 2.25). There was an increase in the odds of encountering shorebird habitat as horizontal interspersion increased; compared to horizontal interspersion of 0, an NWI class with interspersion of 4 was 7.1 (2.7 – 16.7) times more likely to have shorebird foraging habitat (Figure 2.19, Table 2.29). Unlike other response variables, an NWI polygon within a wetland complex was 42.0% (6.3 – 64.1%) less likely to have shorebird foraging habitat present; however, when shorebird foraging habitat was present, wetlands

contained 49.8% (8.5 – 106.9%) greater coverage in wetland complexes (Table 2.29). With every 100 m/ha increase in edge density, an NWI polygon was 13.5% (0.7 – 28.0%) more likely to have shorebird habitat (Figure 2.21, Table 2.29), and when shorebird foraging habitat was present, there was a 59.3% (34.5 – 88.7%) increase in the cover with every 10% increase in the proportion of wetlands in the surrounding 5-km buffer (Figure 2.22). In 2017, shorebird foraging habitat was 80.5% (72.0 – 86.4%) less likely during summer and had 26.3% (2.2 – 44.4%) lower cover of shorebird foraging habitat than autumn (Table 2.29).

Estimated shorebird habitat coverage was 6,262 ha (4,032 to 9,382 ha) during summer and 21,744 ha (17,513 to 25,888 ha) during autumn across Illinois in 2017 (Table 2.10). During summer, the Major Water Bodies Natural Division had the greatest areal and percent cover of shorebird habitat in 2017 with 2,496 ha (385 – 2,239 ha) and 3.0% cover (Table 2.13, 2.14), whereas the lake NWI class had the greatest areal and percent cover with 3,221 ha (1,261 – 6,161 ha) and 2% cover (Table 2.10, 2.11). Subsequently, during autumn, the lake NWI class had the greatest areal and percent cover of 10,375 ha (9,490 – 10,546 ha) and 7% cover (Table 2.10, 2.11). Again, during autumn, the Major Water Bodies Natural Division had the greatest areal and percent cover with 8,041 ha (6,007 – 9,347 ha) and 10% cover (Table 2.13, 2.14).

2.5 DISCUSSION

Suitable habitat resources for dabbling ducks, marsh birds, and shorebirds comprised a small proportion of NWI wetlands during all survey periods. Furthermore, important cover varied among NWI classes and Illinois Natural Divisions. Shallow inundation, considered to provide suitable foraging habitat for dabbling ducks, had the greatest cover of any suitability metric measured, comprising an estimated 29% of wetland area in Illinois during spring migration; however, this does not include adjustments for extremely dense vegetation that likely was inaccessible. Furthermore, highly interspersed emergent vegetation and open water (i.e., hemi-marsh) considered suitable for dabbling ducks, other waterfowl and marsh birds, was extremely rare and areal cover was <1% of wetland area surveyed. Suitable habitat for migrating and breeding marsh birds and migrating shorebirds was particularly limited during critical periods. For secretive marsh birds, cover of important emergent vegetation was low with an estimated 5% cover of inundated dense persistent emergent vegetation (PEM; e.g., *Typha* spp.)

and only 9% inundated non-persistent emergent vegetation (NPE; e.g., moist-soil) across NWI polygon area in Illinois. Even in emergent NWI polygons, both inundated persistent and non-persistent emergent vegetation comprised just 20% of NWI polygon area. Estimated mudflat cover for migrating shorebirds was also low (e.g., $\leq 6\%$) but was greater in summer than autumn (e.g., 6% vs. 4%) during 2016 – 2017, suggesting that mudflats were more limited in cover during autumn migration in Illinois. However, differences in shorebird habitat between seasons and between years of the study suggest that mudflat and shallow inundation (i.e., <10 cm) cover is highly variable. Wetlands with greater complexity and connectivity to other wetland types had the greatest cover of suitable inundation and vegetation. Therefore, restoration efforts focused on wetland complexes may increase suitability of wetlands for a wide variety of bird species. Additionally, conservation planners should dramatically reduce estimates of wetland availability from spatial databases, such as NWI, when used to evaluate wetland suitability or supply on the landscape for wetland-dependent migratory birds during critical periods.

The best supported models consistently included variables associated with wetland connectivity and complexity; however, coefficients of determination were $<34\%$ and a substantial portion of this variation was often explained by the base variables NWI class and Natural Division. While results such as these are not uncommon for ecological field studies, there are likely other abiotic and biotic factors that I did not consider that influence important inundation and vegetation cover for waterbirds. For instance, soil conditions, nutrient inputs, flood recurrence interval, water quality, geomorphology (e.g., Mitsch and Gosselink 2015) and vegetation life history features (van der Valk and Davis 1978, van der Valk 1981) likely impact wetland vegetation and hydrological cover considered important for waterbirds.

Wetland connectivity to rivers and streams had variable impacts on habitat suitability. River connectivity appeared to be a negative predictor of dense persistent emergent vegetation (e.g., *Typha* spp.) but an overall positive predictor of mudflat cover in Illinois wetlands. These effects were stronger and were more apparent in the NWI polygons and plots that contained major river and stream systems and in Natural Divisions such as the Major Water Bodies, Upper Mississippi River and Illinois River Bottomlands, and Wabash River Border. Wetlands with intermediate and permanent connectivity to rivers had the greatest estimated mudflat coverage. Seasonal fluctuation observed in some river systems (e.g., Illinois, Mississippi, and Ohio Rivers)

in Illinois may have influenced the presence and cover of mudflat habitat. For instance, hydrological fluctuations in permanently connected wetlands could provide exposed mudflats along the edges of rivers, whereas intermediate connections could provide extensive mudflat habitat after large flood pulses (Sparks et al. 1997). My results are similar to those of Albanese and Davis (2013), who reported that the greatest area of shorebird habitat was associated with permanent riverine and temporary wetlands. Furthermore, Smith et al. (2011) posited that wetlands associated with large river systems may have more shorebird habitat than those that are disconnected, and Twedt (2013) determined that proposed levees and infrastructure in southeast Missouri that would sever river connectivity would severely deplete shorebird foraging habitat in both spring and autumn. However, fluctuating hydrology and sedimentation associated with large river systems likely created conditions unsuitable for dense PEM vegetation establishment and persistence (Harris and Marshall 1963). Sedimentation increases turbidity and thereby reduces the penetration of light needed by submerged aquatic vegetation and precludes wetland plants (e.g., emergent vegetation, PEM) from rooting in the soft bottoms created from deposition (Havera 1999). This possibly elucidates why NWI polygons that were disconnected from rivers had the greatest predicted cover of dense PEM with an overall decreasing trend as river connectivity increased.

The presence and estimated cover of suitable habitat characteristics was greater when NWI classes were within wetland complexes (i.e., other wetland types; Figures 2.6, 2.8, 2.12) or when NWI polygons had greater horizontal interspersion. NWI polygons within a wetland complex had higher wetland density in the surrounding 100-m and 5-km buffers, and diversity of contiguous or nearby wetlands may have offered waterbirds varying food, cover, and predator avoidance at local and landscape scales (Fleming 2010). Also, in areas of high wetland density NWI polygons potentially were more insulated than isolated wetlands from impacts from surrounding land use such as urbanization (e.g., increased impervious surfaces, altered hydrology) or agriculture (e.g., increased sedimentation and runoff) that influence wetland cover and hydrology (Skagen et al. 2008, Ward et al. 2010). Here, many wetlands determined to be within a wetland complex were located in large river floodplains where they likely received inputs of floodwater and or periods of increased inundation. Water level fluctuation and periods of high water impact vegetation cover in wetlands (Low and Bellrose 1944, Weller and Spatcher 1965, van der Valk and Davis 1978). For instance, periods of high water or water level

fluctuation may drown some wetland vegetation or limit the establishment of woody vegetation, whereas subsequent low water levels may expose mudflats and allow germination of numerous wetland vegetation species. Thus, inputs of floodwater and fluctuating inundation could provide more diverse wetland vegetation structure and regions of suitable habitat for waterbirds in wetlands located within wetland complexes over more isolated wetlands.

Both landscape and local habitat-level complexity are key components of the wetland-complex concept (Pearse et al. 2012). Here, local, NWI polygon-level horizontal interspersion and edge density were predictors of suitable inundation and vegetation cover. As horizontal interspersion increased, so too did the probability of presence and cover of suitable metrics for waterbirds. Generally, NWI polygons that were patchier and had more diverse vegetation structure were more likely to have greater estimates of suitable cover and inundation. Waterbirds are often positively associated with habitat heterogeneity (Gordon et al. 1998) and increased horizontal and vertical structure (Riffell et al. 2001). My results may further support that wetlands proximal to other wetlands and wetland types offer a greater diversity of resources through spatially variable inundation and vegetation cover.

On the other hand, edge density, a quantitative measure of edge interface between open water and vegetation per hectare, had very slight relationships with suitable metric presence and cover. When used by others (e.g., Rehm and Baldassarre 2007) edge density represented the interface between open water and just emergent vegetation per unit area; however, hemi-marsh and even inundated emergent vegetation did not compose large areas in the study, precluding this approach specifically. Here, I also calculated edge density from GPS tracks and thematic maps created at each visit to a plot where observers mapped distinct inundation and vegetation boundaries by foot or by boat potentially adding detail that could not be differentiated while using aerial imagery (e.g., differentiating inundated dense vegetation) or other methods (Kaminski and Prince 1981, Murkin et al. 1997), potentially limiting comparisons with other studies.

While others have found that wetland management influences wetland habitat characteristics and suitability for waterbirds (Fleming 2010), this model set performed poorly. Active management of wetlands for waterfowl often increases the suitability and use of habitat

by waterfowl and other waterbirds (Pankau 2008, Kaminski et al. 2009) by providing preferred foraging depths or vegetation structure (e.g., hemi-marsh conditions). However, instances where managers perform spring drawdowns to elicit moist-soil vegetation growth may detract from inundation and cover important for marsh birds or shorebirds (e.g., Fredrickson and Taylor 1982, Gray et al. 2013, Bradshaw 2018). Furthermore, many management areas in Illinois are highly reliant on river levels to perform recommended management practices (O'Neal and Heske 2007), and based on my observations, infrastructure was often damaged from large flood events or in disrepair due to other reasons (e.g., budget, etc.) and therefore unusable, and most NWI polygons were unmanaged. In this case and at this scale, management structures and intensity did not predict suitable inundation and vegetation for waterbirds, and NWI class, Natural Division, and connectivity to other wetlands or rivers were more relevant.

While surrounding land use or proportion of the surrounding landscape composed of wetlands impacts waterbirds (Havera et al. 1992) and wetlands (e.g., hydrology; Zedler and Kercher 2005, Ward et al. 2010), the land use and disturbance model set was not competitive. I included variables from local and landscape scales including the Landscape Development Index (LDI) representing the intensity of anthropogenic land within a 5-km buffer, the proportion of wetland at 5-km and 100-m, and modified Ohio Rapid Assessment Method (ORAM) score representing an anthropogenic gradient within NWI polygons. While others (Stapanian et al. 2004, Peterson and Niemi 2007) have used ORAM to assess anthropogenic disturbance and vegetation structure within wetlands, the modified ORAM score was not competitive against the connectivity/complexity models and had small effect sizes. However, in general, as ORAM score increased (i.e., anthropogenic disturbance decreased), so too did the probability of and predicted estimates of the suitable cover and inundation metrics. The LDI has been correlated to other disturbance gradients (Mack 2006), but, in this case, the LDI at the landscape-scale was unrelated to habitat suitability metrics. However, Ward et al. (2010) found that development within 2 km of wetlands seemingly impacted wetland structure by increasing the prevalence of open water ponds and dense vegetation, thereby eliminating emergent hemi-marsh areas. Generally, as the proportion of wetlands increased at the 5-km landscape-scale, there were increases in suitable metrics (e.g., shorebird habitat and mudflats; Figure 2.22). However, wetland density at this scale was low (5 – 15%), and the variable representing the proportion of 5-km buffer composed of wetlands was in the best-supported model for only shorebird foraging

habitat. This metric was only measured in 2017, so estimates should be considered more tentative. Perhaps wetland loss is too high in Illinois to predict wetland conditions at these scales.

While the water regime and climate model set containing NWI water regimes and the Palmer Hydrological Drought Index (PHDI) did not compete with the wetland connectivity and complexity models, there were still some overall trends. Regarding the models predicting shallow water (<45cm) accessible to foraging dabbling ducks, the top model contained Natural Division, NWI and water regime and performed better than the null model. The predicted estimates for shallow water did follow the NWI water regime general trends in that temporary, seasonal, and semi-permanent water regimes had greater proportions of shallow inundation than more permanent water regimes (e.g., intermittently exposed and permanent). While not statistically significant in this study, understanding broad-scale dynamics and trends of wetland water regimes identified by the NWI would allow users to better relate digital NWI data to wetlands and habitat characteristics over large areas (Niemuth et al. 2010). Others have used climate data to relate habitat availability and suitability (e.g., Albanese and Davis 2013), and here, as the PHDI increased, indicating wetter conditions than average, I observed an increase in suitable habitat metrics (e.g., increased inundated NPE and mudflats). However, both the 2016 and 2017 survey periods had wetter conditions than average (NCDC 2010). Thus, these results likely do not represent drought conditions. Less precipitation may limit wetland suitability for some waterbirds; however, in more permanent wetlands, lower water conditions resulting from drought could increase the cover of mudflats or shallow inundation suitable for shorebirds and dabbling ducks. Therefore, estimates should be reexamined during a future drought period, and further refinement and inclusion of climate variables could increase the model performance and ability to predict the probability of and cover of inundation and cover critical for waterbirds.

I primarily used qualitative approaches to assess wetland suitability. I visually assessed wetland cover in the field and employed a rapid assessment method to survey impacts to wetland habitat (Mack 2001). Wetlands are frequently surveyed using rapid assessment methods, and typically, these approaches involve the measurement of field metrics (Sutula et al. 2006), qualitative measures of a biological or physical attribute that reflects habitat suitability for wildlife or wetland condition relative to pristine reference wetlands (Fennessy et al. 2004).

While rapid assessment methods have been used successfully to discern impacts on wetlands and wetland suitability (Fennessy et al. 2004, Stapanian et al. 2004, Peterson and Niemi 2007), the modified ORAM score, representing disturbance, was not a major predictor of inundation and vegetation cover considered suitable for waterbirds. Traditionally, the ORAM is intended for assessment using an entire wetland (Mack 2001), whereas we assessed spatially balanced random plots and conducted our assessment at the NWI polygon level. In addition, many plots contained deepwater habitats that are not considered wetlands by most rapid assessment methods (e.g., too deep, open or unconsolidated; Cowardin et al. 1979). Perhaps these adjustments precluded the adoption of a modified ORAM score to predict wetland suitability for waterbirds.

The NWI is currently the most comprehensive wetland data available; however, due to errors of omission and commission (Dvoretz et al. 2012, Matthews et al. 2016), it likely does not include all the wetland area within Illinois and includes areas of non-wetland upland. Furthermore, I omitted very small NWI wetlands <0.5 ha, so the estimates of suitable wetland cover here do not include small wetlands mapped by NWI or wetland area omitted by NWI. Small wetlands are important for wetland-dependent wildlife (e.g., Gibbs 1993) despite evidence that some species are area sensitive and prefer larger wetlands (Brown and Dinsmore 1989). However, the NWI often omits small, temporary wetlands (Tiner 1997, Matthews et al. 2016), suggesting that small wetlands may not be adequately mapped, and it is unclear whether small wetlands comprise a substantial proportion of wetland area or provide suitable habitat in Illinois. In addition to small wetlands, other non-wetland areas such as uplands as well as agricultural lands may also provide suitable habitat at different times due to large rain events that produce sheet water and temporary shallow inundation (Lagrange and Dinsmore 1989), and these areas were not considered in this analysis. While small wetlands and non-NWI wetlands were not included in this assessment, a concurrent study examining the spatial extent of wetland inundation and vegetation cover may provide further insight into small wetlands mapped and omitted by NWI (J. R. O'Connell, Southern Illinois University).

Habitat suitability can be difficult to study and often involves either the direct measurement of habitat attributes (e.g., vegetation cover or structure, prey availability) or the measurement of variables associated with individuals or populations (e.g., demographics, distribution, or body condition; Johnson 2007). While ambiguity in the ecological literature

regarding definitions and spatiotemporal scales of habitat and habitat suitability or quality deepen this issue (Block and Brennan 1993, Hall et al. 1997, Johnson 2007), some features posited to govern habitat suitability can be quantified in well-studied systems. While habitat suitability entails more than vegetation cover and structure, with well-studied species, we have enough understanding of diet/foraging resources, and habitat preferences to attempt to answer this question (Johnson 2007). Further research is needed to study habitat-specific measures of density, reproduction, and survival to offer alternative measures of habitat suitability to coincide with this large-scale assessment of wetland suitability for waterbirds during critical periods in Illinois.

Waterbirds likely experience cross-seasonal effects, and since migratory species cross a plethora of jurisdictional boundaries, management is often difficult to coordinate (Myers et al. 1987). Illinois had undergone extensive wetland loss, and due to the importance of habitat throughout the annual cycle, it is imperative to conserve and manage remaining wetland habitat to meet the needs of wetland-dependent wildlife. According to the Illinois Wildlife Action Plan's Wetlands Campaign (INDR 2005), remaining natural wetlands (excluding floodplain forest) now occupy about 1% of Illinois, whereas only ~2,700 hectares (0.05%) are reported as high quality (IDNR 2005). Here, I provided further insight into the suitability of wetlands for waterbirds by identifying temporal relationships of inundation and suitability metrics (Table 2.2) between seasons and recognized that wetlands with greater complexity and high connectivity to other wetland types offer the greatest amount of suitable inundation and cover. Results may be used by the Upper Mississippi River and Great Lakes Region Joint Ventures and Illinois DNR, and this sets the stage for future waterbird habitat research at varying spatial scales in Illinois (i.e., Natural Divisions). Furthermore, results of this study will aid in the development of a model to predict habitat suitability and availability in collaboration with Southern Illinois University (J. R. O'Connell). Future research objectives and emphasis should be placed on identifying variables appropriate for predicting wetland suitability for waterbirds by combining NWI data with other available spatial data and addressing demographic responses (e.g., survival, nest success, breeding propensity) of waterbirds to habitat suitability in Illinois. Additionally, the NWI water regimes could be incorporated into the future habitat quantity or suitability assessments to provide more finely tuned estimates. Increasing our understanding of the factors

influencing the dynamics of wetland suitability and availability throughout the annual cycle will assist future conservation efforts for these species.

2.6 TABLES AND FIGURES

Table 2.1. Wetland classifications types used in analyses 2015–2017. For more information, see the National Wetland Inventory Mapper (<https://www.fws.gov/wetlands/data/Mapper-Wetlands-Legend.html>).

Wetland Type	Wetland Type Abbreviation ^a	NWI Map Code	Cowardin System and Class	General Description
Freshwater Forested and Shrub-scrub	Forested	PFO, PSS	Palustrine forested and/or palustrine shrub	Forested swamp or wetland shrub bog or other wetland with 30% woody vegetation cover >1 meter in height
Freshwater Emergent	Emergent	PEM	Palustrine emergent	Herbaceous marsh, fen, swale and wet meadow, non-woody
Freshwater Pond	Pond	PUB, PAB	Palustrine unconsolidated bottom, palustrine aquatic bed	Pond, small wetland with open water or aquatic bed vegetation only
Riverine	Riverine	R	Riverine wetland and deepwater	River or stream channel
Lake	Lake	L	Lacustrine wetland and deepwater	Lake or reservoir basin
Other Freshwater Wetland	Other	Misc. types	Palustrine wetland	Farmed wetland, ditches, saline seep, and other miscellaneous wetlands

^aWetland Type Abbreviation is used hereafter in tables and figures.

Table 2.2. Cover and inundation metrics with associated response variable, desired range, and source material that are considered suitable habitat for dabbling ducks, marsh birds, and shorebirds collected during site visits in spring, summer, and autumn during 2016 – 2017 throughout Illinois by INHS and SIU personnel.

Focal Species	Indicator	Metric	Response Variable	Source
Dabbling Ducks	Hydrology	Depth	≤45cm	McGilvrey 1966, Sousa and Farmer 1983, Isola et al. 2000, Taft et al. 2002
	Vegetation	Cover	Flooded forested (i.e., hard mast-producing species; <i>Quercus</i> spp.)	Bellrose 1976, Baldassarre and Bolen 2006, Gray et al. 2013
Marsh Birds	Vegetation	Cover	Flooded dense persistent emergent vegetation (PEM; e.g., <i>Typha</i> spp.) Flooded non-persistent emergent vegetation (NPE; e.g., moist-soil)	Weller and Spatcher 1965, Meanley 1969, Brackney and Bookhout 1982, Eddleman 1988, Gibbs et al. 1992, Lor and Malecki 2006, Darrah and Krementz 2010
Shorebirds	Hydrology	Depth	<10cm	Helmers 1992, Taft et al. 2002, Skagen et al. 2005, Twedt 2013
		Cover	Exposed mudflats	Helmers 1992, Davis 1996, Isola et al. 2000, Twedt 2013, Norris 2015

Table 2.3. Inundation and vegetation categories mapped and separated during site visits 2016 – 2017 and used in thematic map digitizing.

Inundation	Inundated
	Dry
	Mud
Vegetation	Surface (flat mowed grass or floating-leaved aquatic vegetation)
	Herbaceous short (<1m)
	Herbaceous tall (>1m)
	Herbaceous dense cattail (e.g., <i>Typha</i> spp.)
	Herbaceous dense <i>Phragmites</i>
	Woody scrub-shrub (<6m)
	Woody forest (>6m)
Other cover	Open (unvegetated)
	Other (detailed notes needed)

Table. 2.4. Model group, variables, variable code/abbreviation, variable type (continuous or categorical), and description for habitat suitability models.

Model Group	Variable(s)	Acronym	Type	Variable description
Management	Mowing/ disking	MOW	Categorical	Presence of mowing or disking
	Impoundments	IMP	Categorical	Presence or influence from levees, dikes, or impoundments
	Water control structure	WCS	Categorical	Presence or influence from water control structures (i.e., stop logs)
	Management intensity	M_INT	Categorical	None, low, medium, high
Disturbance/ land use	LDI 5km	LDI_5km	Continuous	Landscape development intensity score for surrounding 5-k buffer.
	LDI 100m	LDI_100m	Continuous	Landscape development intensity score for surrounding 100-m buffer.
	Prop wetlands (5k)	Pwet_5km	Continuous	Proportion of wetlands (woody and emergent) in surrounding 5-k buffer.
	Prop wetlands (100m)	Pwet_100m	Continuous	Proportion of wetlands (woody and emergent) in surrounding 100-m buffer.
	ORAM score	ORAM	Continuous	Score representing anthropogenic disturbance and alterations to natural wetland hydrologic regime or habitat, ranging from 0 (highly disturbed/impacted) to 100 (pristine unaltered wetland conditions).
Water regime/ climate	PHDI	PHDI	Continuous	Palmer Hydrological Drought Index to represent precipitation, evapotranspiration, and soil moisture.
	NWI water regime	WATER_REG	Categorical	From wettest to driest, permanent, intermittently exposed, semi-permanent, seasonal, temporary, and saturated.
Wetland cover/ connectivity	River connectivity	RC	Categorical	A factor variable scoring a wetland's connectivity to rivers ranging from 0 (no connection apparent) to 7 (permanent connection).
	Horizontal interspersion	HI	Categorical	An ordinal variable rating a wetland's interspersion and vegetative patch complexity ranging from 0 (monoculture/no interspersion) to 7 (highly interspersed with various vegetation covers/inundation).
	Edge density	ED	Continuous	Open water vegetation edge divided area to represent interspersion.
	Wetland complex	WC	Categorical	Proximity to other wetland types or connected to other wetlands.

Table 2.5. The Ohio Rapid Assessment Method quantitative portion metrics and submetrics (Mack 2001; adapted from Peterson and Niemi 2007) used to calculate anthropogenic disturbance throughout Illinois in NWI-mapped polygons during 2016 – 2017.

Metric 1.	Wetland area (ha)*
Metric 2.	Upland Buffers and surrounding land use
	Submetric 2a. Average buffer width*
	Submetric 2b. Intensity of surrounding land use
Metric 3.	Hydrology
	Submetric 3a. Sources of water
	Submetric 3b. Connectivity
	Submetric 3c. Maximum water depth
	Submetric 3d. Duration of inundation/saturation
	Submetric 3e. Modifications to natural hydrologic regime
Metric 4.	Habitat alteration and development
	Submetric 4a. Substrate Disturbance*
	Submetric 4b. Habitat Development
	Submetric 4c. Habitat alteration
Metric 5.	Special wetlands*
Metric 6.	Plant communities, interspersions, and microtopography
	Submetric 6a. Wetland vegetation communities**
	Submetric 6b. Horizontal interspersions
	Submetric 6c. Coverage of invasive plants
	Submetric 6d. Microtopography*

*indicates that this metric or submetric was not utilized in data analysis.

**indicates that this metric was adapted to fit project's goals

Table 2.6. Proportion (visual estimates) of inundation, shallow water, and important vegetation cover types for spring, summer, and autumn 2016 – 2017 in Illinois (Mean [85% CI]). Aquatic bed includes submerged vegetation (e.g., coontail; *Ceratophyllum demersum*) and floating-leaved aquatic vegetation (e.g., pondweeds; *Potamogeton* spp.). Mudflats are areas of exposed saturated mud with sparse vegetation cover (i.e., <30%), and shorebird habitat includes mudflats and very shallow inundation (i.e., <10cm). Data were collected throughout Illinois in NWI-mapped wetlands by INHS and SIU personnel.

Season	NWI Class	Inundated	Shallow (<45cm)	Non-pers. emergent	Pers. emergent	Aquatic bed	Scrub-shrub	Forest	Mudflats	Shorebird (mudflats/<10cm)
Spring	EM	51.8	37.4	10.9	8.1	3.0	11.5	11.4	2.8	1.8
		(46.9, 56.7)	(33.4, 41.4)	(8.2, 13.6)	(7.7, 8.6)	(2.8, 3.2)	(11.0, 12.0)	(10.9, 11.9)	(2.6, 3.1)	(1.6, 2.0)
	FO	32.6	21.1	0.44	1.2	2.0	6.0	24.1	1.1	1.1
		(29.6, 35.6)	(19.0, 23.2)	(0.4, 0.5)	(1.1, 1.3)	(1.9, 2.1)	(5.9, 6.2)	(23.7, 24.5)	(1.05, 1.15)	(1.0, 1.2)
	LAKE	90.6	21.1	1.5	1.7	3.5	3.5	5.3	10.8	5.7
		(88.3, 92.9)	(17.8, 24.4)	(1.4, 1.6)	(1.6, 1.9)	(3.0, 4.0)	(3.1, 3.9)	(5.0, 5.6)	(10.0, 11.6)	(5.2, 6.2)
	POND	83.9	30.5	3.5	4.7	5.9	8.8	8.1	2.3	3.2
		(79.4, 88.4)	(26.1, 34.9)	(3.1, 3.9)	(4.0, 5.4)	(5.3, 6.5)	(8.1, 9.5)	(7.7, 8.5)	(2.0, 2.6)	(2.8, 3.6)
	RIVERINE	87.0	16.9	0.92	0.24	0.10	2.0	10.1	6.3	4.9
		(82.9, 91.1)	(12.5, 21.3)	(0.8, 1.1)	(0.1, 0.3)	(0.07, 0.13)	(1.8, 2.2)	(8.9, 11.3)	(5.4, 7.2)	(3.5, 6.3)
Summer	EM	58.5	34.3	17.4	8.5	6.9	10.3	16.3	3.4	1.5
		(54.3, 62.7)	(31.2, 37.4)	(15.1, 19.7)	(6.4, 10.6)	(5.2, 7.7)	(9.9, 10.8)	(16.1, 16.6)	(2.2, 4.7)	(0.8, 2.2)
	FO	46.1	22.9	7.9	1.5	2.7	8.7	32.7	2.0	0.11
		(42.5, 49.7)	(20.4, 25.4)	(6.5, 9.3)	(0.8, 2.2)	(2.0, 2.7)	(8.4, 9.0)	(32.6, 32.8)	(1.4, 2.6)	(0.1, 0.2)
	LAKE	91.3	15.8	5.9	2.7	7.0	6.5	8.8	3.4	0.89
		(88.9, 93.7)	(12.7, 18.9)	(4.2, 7.6)	(1.3, 4.1)	(4.6, 9.4)	(6.2, 6.8)	(8.7, 8.9)	(3.3, 3.5)	(0.2, 1.6)
	POND	85.2	28.1	10.3	5.5	15.2	7.6	9.9	2.5	0.97
		(81.1, 89.3)	(23.7, 32.5)	(7.5, 13.1)	(2.9, 8.1)	(10.6, 19.4)	(7.5, 7.7)	(9.3, 10.5)	(1.3, 3.7)	(0.0, 2.0)
	RIVERINE	87.0	12.7	3.2	1.3	0.5	2.8	12.5	7.6	0.06
		(83.1, 90.9)	(8.8, 16.6)	(1.8, 4.6)	(0.0, 2.6)	(0.2, 0.8)	(1.7, 3.9)	(12.0, 13.0)	(3.9, 11.3)	(0.0, 0.1)
Autumn	EM	33.9	22.5	11.0	5.2	3.6	4.5	6.7	0.77	3.2
		(29.7, 38.1)	(19.4, 25.6)	(8.7, 13.4)	(3.5, 6.9)	(2.3, 4.9)	(3.4, 5.6)	(6.0, 7.4)	(0.4, 1.2)	(2.1, 4.4)
	FO	21.8	13.0	3.8	0.59	2.1	3.1	10.0	0.34	1.4
		(18.9, 24.7)	(11.3, 14.7)	(2.9, 4.7)	(0.3, 0.8)	(1.3, 2.9)	(2.4, 3.8)	(9.1, 10.9)	(0.3, 0.4)	(1.0, 1.8)
	LAKE	79.5	24.1	5.5	2.1	7.7	2.6	4.0	10.0	7.0
		(75.6, 83.4)	(20.6, 27.6)	(3.8, 7.2)	(1.0, 3.2)	(5.0, 10.4)	(2.4, 4.1)	(3.0, 5.0)	(9.9, 10.1)	(6.7, 7.3)
	POND	67.5	25.6	5.1	4.6	14.3	5.5	6.3	2.9	2.8
		(61.4, 73.6)	(21.5, 29.7)	(3.2, 7.0)	(2.5, 6.8)	(10.1, 17.9)	(5.1, 5.9)	(5.9, 6.8)	(1.6, 4.2)	(1.5, 4.1)
	RIVERINE	79.9	20.3	2.7	1.1	4.8	2.5	6.3	1.6	1.5
		(74.4, 85.4)	(15.2, 25.4)	(1.5, 3.9)	(0.1, 2.1)	(0.8, 8.8)	(1.0, 4.0)	(4.1, 8.5)	(1.0, 2.3)	(0.7, 2.4)

Table 2.7. Rankings of competing top models (ΔAIC_c) from the four model sets according to Akaike’s Information Criterion for small sample sizes (AIC_c) with the number of parameters (K), model weights (w_i) for predicting shallow inundation (i.e. <45cm) for foraging dabbling ducks during spring migration using data collected during 2016 and 2017 spring migrations (15 Feb – mid-April) through Illinois, USA.

Model sets^a	<i>K</i>	<i>AIC_c</i>	ΔAIC_c	<i>w_i</i>
Connectivity/complexity	25	-511.22	0.00	1.00
Water regime/climate	24	-492.21	19.14	0.00
Land use/disturbance	22	-487.17	25.67	0.00
Management	21	-468.98	42.73	0.00

^aCovariates listed in Table 2.4.

Table 2.8. Rankings of competing models ($\Delta AIC_c < 10$) according to Akaike’s Information Criterion for small sample sizes (AIC_c) with the number of parameters (K), model weights (w_i) for predicting shallow inundation (i.e., <45cm) for dabbling ducks during spring migration by connectivity and complexity covariates using data collected during 2016 and 2017 spring migrations (15 Feb – mid-April) through Illinois, USA.

Model^a	K	AIC_c	ΔAIC_c	w_i	R^2
HI + ND + NWI + WC	25	-511.22	0.00	0.60	0.22
ED + HI + ND + NWI + WC	26	-509.81	1.41	0.30	0.22
HI + ND + NWI + RC + WC	32	-505.03	6.20	0.03	0.23
ED + HI + ND + NWI + RC + WC (Global)	33	-503.56	7.66	0.01	0.23
HI + ND + NWI	24	-501.9	9.37	0.01	0.20

^aCovariates include: ED – Edge density (vegetation to open water metric associated with edge complexity; total edge in meters/ha); HI – Horizontal interspersion (ordinal complexity variable; ranging from 0=none/monotypic to 4=highly complex/diverse); ND – Natural Division (distinct ecoregions within Illinois); NWI – NWI class (dominant vegetation class or deepwater habitat; Cowardin et al. 1979); RC – River connectivity (ordinal variable representing connectivity to rivers or streams; 0=isolated/no connection apparent to 7=highly connected, permanent connection); WC – Wetland complex (presence/absence variable whether the NWI class in proximity to other wetland types)

Table 2.9. Coefficients, beta estimates, and confidence intervals ($\pm 85\%$) for the best supported beta regression model predicting shallow inundation (i.e., <45cm) cover for dabbling ducks during spring migration by connectivity and complexity covariates, and base variables Natural Division (ND)^a and NWI class. Data were collected during 2016 and 2017 spring migrations (15 Feb – mid-April) through Illinois, USA.

Coefficients	Beta regression	
	Beta	Confidence interval
(Intercept)	0.233	(0.165, 0.318)
Horizontal interspersion (1)	0.283	(0.229, 0.344)
Horizontal interspersion (2)	0.350	(0.289, 0.417)
Horizontal interspersion (3)	0.386	(0.318, 0.458)
Horizontal interspersion (4)	0.435	(0.359, 0.514)
ND (GP)	0.138	(0.095, 0.196)
ND (IRMRSA)	0.231	(0.167, 0.312)
ND (LMRB)	0.168	(0.119, 0.232)
ND (MMRB)	0.228	(0.151, 0.328)
ND (MWB)	0.095	(0.068, 0.132)
ND (NM)	0.262	(0.187, 0.353)
ND (O)	0.215	(0.140, 0.315)
ND (RRHC)	0.132	(0.082, 0.206)
ND (SH)	0.174	(0.114, 0.257)
ND (STP)	0.207	(0.151, 0.278)
ND (UMRIRB)	0.144	(0.105, 0.194)
ND (WD)	0.177	(0.104, 0.286)
ND (WFP)	0.160	(0.103, 0.240)
ND (WRB)	0.173	(0.121, 0.241)
NWI class (FO)	0.183	(0.156, 0.212)
NWI class (LAKE)	0.224	(0.185, 0.269)
NWI class (POND)	0.243	(0.201, 0.290)
NWI class (RIVERINE)	0.190	(0.151, 0.235)
Wetland complex (+)	0.305	(0.273, 0.338)

^aSee Table 2.27 for Natural Divisions.

Table 2.10. Scaled up suitable metric estimates (hectares) by NWI class from 2,000 parametric bootstrap replicates from data collected in spring, summer, and autumn in Illinois, USA by INHS and SIU. Both PEM and NPE are inundated. Percent shorebird refers to mudflats and shallow inundation <10cm. The 85% confidence intervals are shown in parentheses. The total area across NWI classes can be viewed in Table 2.12. Consolidated NWI information can be viewed in Table 2.1.

Season	Response Variable	Emergent	Forest	Lake	Other	Pond	Riverine
Spring	Shallow inundation (i.e., <45cm)	26,560	81,508	45,548	64	12,747	15,456
		(26,479, 26,641)	(81,185, 81,828)	(43,944, 47,180)	(63, 67)	(12,721, 12,773)	(14,172, 16,796)
	Inundated forest	11,017	81,912	18,634	26	5,310	8,932
		(10,874, 11,161)	(81,047, 82,769)	(14,014, 23,270)	(22, 30)	(5,277, 5,344)	(4,254, 13,753)
Summer	Dense persistent emergent (PEM)	11,179	6,600	6,825	167	3,241	1,839
		(10,950.6, 11,396)	(6,197, 7,010)	(4,348, 10,339)	(12, 21)	(3,185, 3,296)	(606, 3,901)
	Non-persistent emergent (NPE)	10,468	26,421	11,948	26	3,866	3412
		(10,275, 10,656)	(25,871, 26,953)	(8,194, 16,309)	(22, 29)	(3,825, 3,908)	(1,340, 7,349)
	Mudflats	2,391	11,033	14,965	7	1,776	6,640
		(2,320, 2,469)	(10,686, 11,394)	(9,747, 20,446)	(5, 9)	(1,745, 1,807)	(2,893, 10,257)
Percent shorebird	576	1,565	3,221	2	273	627	
	(542, 613)	(1,424, 1,724)	(1,261.2, 6,161.2)	(1, 2)	(259, 288)	(89, 1,805)	
Autumn	Mudflats	1,619	7,424	10,482	5	1,216	4,580
		(1,577, 1,654)	(7,221, 7,598)	(9,357, 10,623)	(4, 5)	(1,200, 1,229)	(2,380, 6,511)
	Percent shorebird	1,967	6,160	10,375	5	1089	2,149
		(1,912, 2,012)	(5,936, 6,362)	(9,490, 10,546)	(4, 6)	(1,067, 1,107)	(855, 3,285)

Table 2.11. Scaled up suitable metric estimates percent of total area by NWI class from 2,000 parametric bootstrap replicates from data collected in spring, summer, and autumn in Illinois, USA by INHS and SIU. Both PEM and NPE are inundated. Percent shorebird refers to mudflats and shallow inundation <10cm. The total percent area across NWI classes can be viewed in Table 2.12. Consolidated NWI information can be viewed in Table 2.1.

Season	Response Variable	Emergent	Forest	Lake	Other	Pond	Riverine
Spring	Shallow inundation (i.e., <45cm)	0.37	0.25	0.31	0.33	0.33	0.30
	Inundated forest	0.15	0.25	0.13	0.13	0.14	0.17
Summer	Dense persistent emergent (PEM)	0.16	0.02	0.05	0.09	0.08	0.04
	Non-persistent emergent (NPE)	0.15	0.08	0.08	0.13	0.10	0.07
	Mudflats	0.03	0.03	0.10	0.03	0.05	0.13
	Percent shorebird	0.01	0.00	0.02	0.01	0.01	0.01
Autumn	Mudflats	0.02	0.02	0.07	0.02	0.03	0.09
	Percent shorebird	0.03	0.02	0.07	0.03	0.03	0.04

Table 2.12. Scaled up suitable metric estimates (hectares) and percent of total area across NWI class from 2,000 parametric bootstrap replicates from data collected in spring, summer, and autumn in Illinois, USA by INHS and SIU. Both PEM and NPE are inundated. Percent shorebird refers to mudflats and shallow inundation <10cm. Consolidated NWI information can be viewed in Table 2.1.

Season	Response Variable	Total
Spring	Shallow inundation (i.e., <45cm)	181,882 0.23
	Inundated forest	125,832 0.20
Summer	Dense persistent emergent (PEM)	29,702 0.05
	Non-persistent emergent (NPE)	56,141 0.09
	Mudflats	36,812 0.06
	Percent shorebird	6,262 0.01
Autumn	Mudflats	25,324 0.04
	Percent shorebird	21,744 0.03

Table 2.13. Scaled up suitable metric estimates (hectares) by ND^a from 2,000 parametric bootstrap replicates from data collected in spring, summer, and autumn in Illinois, USA by INHS and SIU. Both PEM and NPE are inundated. Percent shorebird refers to mudflats and shallow inundation <10cm. The 85% confidence intervals are shown in parentheses. Natural Divisions can be viewed in Figure 2.1.

Season	Response Variable	CP	GP	IRMRSA	LMRB	MMRB
Spring	Shallow inundation (i.e., <45cm)	8,092 (7,987, 8,198)	19,816 (19,423, 20,221)	1,707 (1,692, 1,721)	8,059 (7,936, 8,189)	2,136 (2,082, 2,192)
	Inundated forest	8,869 (8,599, 9,117)	11,174 (10,491, 11,815)	709 (669,748)	6,267 (5,846, 6,634)	659 (548, 773)
Summer	Dense persistent emergent (PEM)	570 (475, 678)	4,183 (3,401, 5,203)	307 (279, 335)	1,537 (1,324, 1,776)	31 (22, 47)
	Non-persistent emergent (NPE)	3,046 (2,905, 3,293)	7,269 (6,834, 7,857)	661 (642, 692)	4,125 (3,896, 4,460)	511 (458, 604)
	Mudflats	955 (882, 1,017)	5,596 (4,846, 6,115)	120 (105, 136)	1,181 (1,054, 1,267)	198 (169, 229)
	Percent shorebird	26 (12, 52)	278 (158, 583)	55 (46, 67)	393 (299, 530)	65 (51, 85)
Autumn	Mudflats	646 (623, 711)	3,856 (3,562, 4,441)	82 (78, 98)	798 (756, 919)	135 (124, 166)
	Percent shorebird	125 (105, 170)	1,217 (999, 1,637)	191 (185, 204)	1,442 (1,363, 1,613)	210 (201, 232)

^aSee Table 2.30 for Natural Divisions.

Table 2.13. Continued

Season	Response Variable	MWB	NM	O	RRHC	SH
Spring	Shallow inundation (i.e., <45cm)	28,076 (26,241, 30,053)	24,568 (24,346, 24,801)	1,097 (1,075, 1,118)	2,475 (2,329, 2,636)	2,723 (2,590, 2,856)
	Inundated forest	14,37 (7,918, 21,051)	12,779 (12,417, 13,064)	476 (439, 511)	1,430 (1,177, 1,700)	1,440 (1,195, 1,692)
Summer	Dense persistent emergent (PEM)	2,274 (116, 6,977)	12,644 (11,939, 13,334)	80 (70, 93)	73 (46, 105)	543 (324, 901)
	Non-persistent emergent (NPE)	7,171 (2,569, 16,233)	6,561 (6,177, 7,184)	284 (265, 317)	1,104 (920, 1,466)	898 (739, 1,147)
	Mudflats	11,856 (8,995, 16,394)	2,209 (1,971, 2,416)	229 (211, 247)	301 (150, 481)	639 (468, 803)
	Percent shorebird	2,496 (385, 2,239)	235 (176, 370)	5 (2, 9)	228 (159, 370)	113 (48, 247)
Autumn	Mudflats	8,265 (5,754, 10,448)	1,513 (1,412, 1,721)	157 (152, 174)	208 (140, 377)	444 (392, 601)
	Percent shorebird	8,041 (6,008, 9,347)	1,045 (946, 1,271)	22 (19, 28)	296 (237, 407)	389 (326, 531)

^aSee Table 2.30 for Natural Divisions.

Table 2.13. Continued

Season	Response Variable	STP	UMRIRB	WD	WFP	WRB
Spring	Shallow inundation (i.e., <45cm)	42,225 (41,583, 42,883)	20,096 (19,934, 20,253)	191 (169, 215)	6,037 (5,961, 6,121)	14,586 (14,443, 14,729)
	Inundated forest	30,472 (29,424, 31,308)	15,909 (15,281, 16,533)	61 (36, 89)	3,826 (3,685, 3,967)	17,426 (17,079, 17,736)
Summer	Dense persistent emergent (PEM)	4,230 (3,345, 5,627)	2,616 (2,284, 2,992)	33 (17, 62)	253 (222, 288)	329 (261, 422)
	Non-persistent emergent (NPE)	10,886 (10,201, 12,206)	7,601 (7,193, 8,288)	81 (53, 122)	1,460 (1,402, 1,562)	4,482 (4,258, 4,870)
	Mudflats	7,116 (5,895, 8,008)	3,434 (3,095, 3,686)	7 (2, 13)	918 (848, 985)	2,054 (1,930, 2,136)
	Percent shorebird	825 (542, 1,494)	1,139 (951, 1,399)	18 (7, 42)	115 (96, 143)	274 (223, 351)
Autumn	Mudflats	4,873 (4,433, 5,866)	2,341 (2,231, 2,601)	5 (2, 19)	620 (595, 688)	1,382 (1,343, 1,470)
	Percent shorebird	3,371 (3,019, 4,136)	3,786 (3,680, 4,048)	21 (12, 40)	423 (405, 463)	1,165 (1,122, 1,268)

^aSee Table 2.30 for Natural Divisions.

Table 2.14a. Scaled up suitable metric estimates by ND^a from 2,000 parametric bootstrap replicates from data collected in spring, summer, and autumn in Illinois, USA by INHS and SIU. Both PEM and NPE are inundated. Percent shorebird refers to mudflats and shallow inundation <10cm. The 85% confidence intervals are shown in parentheses. Natural Divisions can be viewed in Figure 2.1. The percent of the total area is shown in red.

Season	Response Variable	CP	GP	IRMRSA	LMRB	MMRB
Spring	Shallow inundation (i.e., <45cm)	0.33	0.24	0.30	0.26	0.32
	Inundated forest	0.36	0.14	0.13	0.20	0.10
Summer	Dense persistent emergent (PEM)	0.02	0.05	0.05	0.05	0.00
	Non-persistent emergent (NPE)	0.12	0.09	0.12	0.131	0.077
	Mudflats	0.04	0.07	0.02	0.04	0.03
	Percent shorebird	0.00	0.00	0.01	0.01	0.01
Autumn	Mudflats	0.03	0.05	0.01	0.03	0.02
	Percent shorebird	0.01	0.01	0.03	0.05	0.03

^aSee Table 2.30 for Natural Divisions.

Table 2.14b. Continued

Season	Response Variable	MWB	NM	O	RRHC	SH
Spring	Shallow inundation (i.e., <45cm)	0.34	0.44	0.25	0.28	0.25
	Inundated forest	0.17	0.23	0.11	0.16	0.13
Summer	Dense persistent emergent (PEM)	0.03	0.23	0.02	0.01	0.05
	Non-persistent emergent (NPE)	0.09	0.12	0.07	0.13	0.08
	Mudflats	0.14	0.04	0.05	0.03	0.06
	Percent shorebird	0.03	0.00	0.00	0.03	0.01
Autumn	Mudflats	0.10	0.03	0.04	0.02	0.04
	Percent shorebird	0.10	0.02	0.01	0.03	0.04

^aSee Table 2.30 for Natural Divisions.

Table 2.14c. Continued

Season	Response Variable	STP	UMRIRB	WD	WFP	WRB
Spring	Shallow inundation (i.e., <45cm)	0.30	0.21	0.28	0.24	0.25
	Inundated forest	0.21	0.17	0.09	0.15	0.30
Summer	Dense persistent emergent (PEM)	0.03	0.03	0.05	0.01	0.01
	Non-persistent emergent (NPE)	0.08	0.08	0.12	0.06	0.08
	Mudflats	0.05	0.04	0.01	0.04	0.04
	Percent shorebird	0.01	0.01	0.03	0.00	0.00
Autumn	Mudflats	0.03	0.02	0.01	0.02	0.02
	Percent shorebird	0.02	0.04	0.03	0.02	0.02

^aSee Table 2.30 for Natural Divisions.

Table 2.15. Rankings of top models (ΔAIC_c) from the four model sets for logistic and beta regression according to Akaike’s Information Criterion for small sample sizes (AIC_c) with the number of parameters (K) and model weights (w_i) for predicting inundated forest for dabbling ducks during spring migration using data collected during 2016 and 2017 spring migrations (15 Feb – mid-April) through Illinois, USA.

Model Type	Model Sets^a	K	AIC_c	ΔAIC_c	w_i
Logistic	Connectivity/complexity	24	484.89	0.00	0.99
	Land use/disturbance	21	496.39	11.50	0.00
	Water regime/climate	23	518.60	33.23	0.00
	Management	20	518.12	33.71	0.00
Beta Regression	Connectivity/complexity	22	-447.00	0.00	0.95
	Land use/disturbance	22	-441.29	5.71	0.05
	Water regime/climate	21	-429.25	17.75	0.00
	Management	21	-429.05	17.95	0.00

^aCovariates outlined in Table 2.4.

Table 2.16. Rankings of competing models ($\Delta AIC_c < 10$) in beta and logistic regression according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K) and model weights (w_i) for predicting inundated forest for dabbling ducks during spring migration by connectivity/complexity covariates using data collected during 2016 and 2017 spring migrations (15 Feb – mid-April) through Illinois, USA.

Model Type	Model ^a	K	AIC_c	ΔAIC_c	w_i	R^2
Logistic	HI + ND + NWI + WC	24	484.89	0.00	0.68	0.22
	ED + HI + ND + NWI + WC	25	486.48	1.60	0.31	0.22
	HI + ND + NWI + RC + WC	31	494.50	9.61	0.01	0.23
Beta Regression	ED + ND + NWI + WC	22	-447.00	0.00	0.75	0.21
	ED + HI + ND + NWI + WC	26	-442.79	4.20	0.09	0.22
	ED + ND + NWI + RC + WC	29	-442.48	4.52	0.08	0.24
	ND + NWI + WC	21	-442.10	4.90	0.06	0.19
	ED + HI + ND + NWI + RC + WC (Global)	33	-437.96	8.72	0.01	0.25
	HI + ND + NWI + WC	25	-438.16	9.04	0.01	0.21

^aCovariates include: ED – Edge density (vegetation to open water metric associated with edge complexity; total edge in meters/ha); HI – Horizontal interspersion (ordinal complexity variable; ranging from 0=none/monotypic to 4=highly complex/diverse); ND – Natural Division (distinct ecoregions within Illinois); NWI – NWI class (dominant vegetation class or deepwater habitat; Cowardin et al. 1979); RC – River connectivity (ordinal variable representing connectivity to rivers or streams; 0=isolated/no connection apparent to 7=highly connected, permanent connection); WC – Wetland complex (presence/absence variable whether the NWI class in proximity to other wetland types)

Table 2.17. Coefficients, beta estimates, and confidence intervals ($\pm 85\%$) for best supported models in both logistic and beta regression predicting the probability of and cover of inundated forest by connectivity and complexity covariates and base variables, Natural Division (ND)^a and NWI class. Odds ratios and confidence intervals reported for logistic regression. Inundated forest was always present in the Middle Mississippi River Border (n=15), Shawnee Hills (n=16), and Wisconsin Driftless (n=8) Natural Division, precluding confidence intervals and odds ratio calculations. Data were collected during 2016 and 2017 spring migrations (15 Feb – mid-April) through Illinois, USA.

Coefficients	Logistic regression				Beta regression	
	Beta	Confidence interval	Odds ratio	Confidence interval	Beta	Confidence interval
(Null)	0.522	(0.234, 0.796)	0.092	(-2.909, 0.695)	0.249	(0.187, 0.323)
Edge density	0.2486	(0.2485, 0.2487)
Horizontal interspersions (1)	0.746	(0.606, 0.849)	1.692	(0.410, 4.142)	.	.
Horizontal interspersions (2)	0.867	(0.767, 0.928)	4.953	(2.009, 10.779)	.	.
Horizontal interspersions (3)	0.876	(0.774, 0.935)	5.444	(2.131, 12.262)	.	.
Horizontal interspersions (4)	0.786	(0.638, 0.884)	2.355	(0.612, 5.986)	.	.
ND (GP)	0.224	(0.073, 0.515)	0.735	(0.027, 0.928)	0.118	(0.079, 0.172)
ND (IRMRSA)	0.123	(0.038, 0.331)	0.871	(0.548, 0.963)	0.139	(0.094, 0.200)
ND (LMRB)	0.228	(0.077, 0.512)	0.730	(0.041, 0.924)	0.158	(0.110, 0.221)
ND (MMRB)	1.000	.	.	.	0.129	(0.083, 0.195)
ND (MWB)	0.101	(0.033, 0.269)	0.897	(0.662, 0.969)	0.117	(0.084, 0.162)
ND (NM)	0.551	(0.204, 0.855)	0.124	(-0.765, 4.377)	0.206	(0.145, 0.285)
ND (O)	0.271	(0.080, 0.615)	0.660	(-0.461, 0.921)	0.125	(0.075, 0.202)
ND (RRHC)	0.106	(0.027, 0.338)	0.891	(0.532, 0.975)	0.127	(0.075, 0.205)
ND (SH)	1.000	.	.	.	0.207	(0.141, 0.294)
ND (STP)	0.569	(0.250, 0.839)	0.208	(-0.695, 3.776)	0.159	(0.114, 0.216)
ND (UMRIRB)	0.160	(0.056, 0.381)	0.826	(0.436, 0.946)	0.150	(0.110, 0.202)
ND (WD)	1.000	.	.	.	0.138	(0.081, 0.226)
ND (WFP)	0.301	(0.092, 0.645)	0.606	(-0.667, 0.907)	0.150	(0.094, 0.232)
ND (WRB)	0.704	(0.327, 0.921)	1.178	(-0.554, 9.649)	0.189	(0.135, 0.258)

^aSee Table 2.30 for Natural Divisions.

Table 2.17. Continued.

Coefficients	Logistic regression				Beta regression	
	Beta	Confidence interval	Odds ratio	Confidence interval	Beta	Confidence interval
NWI class (FO)	0.306	(0.220, 0.408)	0.596	(0.370, 0.742)	0.373	(0.326, 0.422)
NWI class (LAKE)	0.456	(0.315, 0.605)	0.231	(-0.404, 0.579)	0.224	(0.179, 0.276)
NWI class (POND)	0.461	(0.329, 0.600)	0.216	(-0.372, 0.552)	0.243	(0.196, 0.298)
NWI class (RIVERINE)	0.352	(0.138, 0.648)	0.503	(-0.683, 0.853)	0.307	(0.241, 0.381)
Wetland complex (+)	0.760	(0.681, 0.824)	1.897	(0.952, 3.299)	0.354	(0.316, 0.393)

^aSee Table 2.30 for Natural Divisions.

Table 2.18. Rankings of competing top models (ΔAIC_c) from the four model sets in the logistic and beta regression according to Akaike’s Information Criterion for small sample sizes (AIC_c) with the number of parameters (K), model weights (w_i) for predicting inundated dense persistent emergent vegetation (PEM) for marsh birds during summer (mid-April – mid-June) using data collected during 2016 and 2017 throughout Illinois, USA. The null model with ND and NWI class was the top model for all four-model sets in beta regression.

Model Type	Model Sets^a	K	AIC_c	ΔAIC_c	w_i
Logistic	Connectivity/complexity	31	480.75	0.00	0.99
	Land use/disturbance	20	521.24	40.49	0.00
	Water regime/climate	28	527.03	46.28	0.00
	Management	19	530.34	49.59	0.00
Beta Regression	Land use/disturbance (Null)	20	-157.73	-	-
	Management (Null)	20	-157.73	-	-
	Connectivity/complexity (Null)	20	-157.73	-	-
	Water regime/climate (Null)	20	-157.73	-	-

^aCovariates outlined in Table 2.4.

Table 2.19. Rankings of competing models ($\Delta AIC_c < 10$) in logistic and beta regression according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K) and model weights (w_i) for predicting inundated dense persistent emergent vegetation (PEM) for migrating and breeding marsh birds during summer (mid-April – mid-June) by connectivity/complexity covariates using data collected during 2016 and 2017 throughout Illinois, USA.

Model Type	Model ^a	K	AIC_c	ΔAIC_c	w_i	R^2
Logistic	HI + ND + NWI + RC + WC	31	480.75	0.00	0.57	0.27
	ED + HI + ND + NWI + RC + WC (Global)	32	482.55	1.80	0.23	0.27
	HI + ND + NWI + RC	30	483.51	2.76	0.14	0.26
	ED + HI + ND + NWI + RC	31	485.49	4.74	0.05	0.27
Beta Regression	ND + NWI (Null)	20	-157.73	0.00	0.47	0.34
	ED + ND + NWI	21	-156.52	1.21	0.25	0.35
	ND + NWI + WC	21	-154.88	2.85	0.11	0.34
	ED + HI + ND + NWI	25	-153.66	4.08	0.06	0.38
	ED + ND + NWI + WC	22	-153.56	4.18	0.06	0.35
	HI + ND + NWI	24	-152.29	5.44	0.03	0.37
	ED + HI + ND + NWI + WC	26	-150.40	7.33	0.01	0.38
HI + ND + NWI + WC	25	-149.13	8.60	0.01	0.37	

^aCovariates include: ED – Edge density (vegetation to open water metric associated with edge complexity; total edge in meters/ha); HI – Horizontal interspersion (ordinal complexity variable; ranging from 0=none/monotypic to 4=highly complex/diverse); ND – Natural Division (distinct ecoregions within Illinois); NWI – NWI class (dominant vegetation class or deepwater habitat; Cowardin et al. 1979); WC – Wetland complex (presence/absence variable whether the NWI class in proximity to other wetland types).

Table 2.20. Coefficients, beta estimates, and confidence intervals ($\pm 85\%$) for both logistic and beta regression predicting the probability of and cover of inundated dense persistent emergent vegetation (e.g., PEM; *Typha* spp.) by connectivity and complexity covariates and base variables, Natural Division (ND)^a and NWI class. Odds ratios and confidence intervals reported for logistic regression. Data were collected during 2016 and 2017 breeding and migration for marsh birds (mid-April – mid-June) through Illinois, USA.

Coefficients	Logistic regression				Beta regression	
	Beta	Confidence interval	Odds ratio	Confidence interval	Beta	Confidence interval
(Null)	0.350	(0.135, 0.650)	0.461	(-0.859, 0.844)	0.102	(0.058, 0.173)
Horizontal interspersion (1)	0.516	(0.274, 0.750)	0.976	(-0.300, 4.575)		
Horizontal interspersion (2)	0.707	(0.463, 0.871)	3.477	(0.599, 11.538)		
Horizontal interspersion (3)	0.854	(0.678, 0.942)	9.871	(2.898, 29.319)		
Horizontal interspersion (4)	0.893	(0.746, 0.959)	14.406	(4.458, 42.486)		
ND (GP)	0.410	(0.184, 0.682)	0.291	(-0.581, 2.974)	0.536	(0.342, 0.720)
ND (IRM RSA)	0.615	(0.379, 0.807)	1.963	(0.133, 6.752)	0.537	(0.367, 0.700)
ND (LMRB)	0.422	(0.223, 0.650)	0.353	(-0.468, 2.439)	0.558	(0.389, 0.714)
ND (MMRB)	0.069	(0.013, 0.296)	0.863	(0.219, 0.976)	0.232	(0.058, 0.596)
ND (MWB)	0.515	(0.284, 0.740)	0.968	(-0.265, 4.270)	0.418	(0.254, 0.601)
ND (NM)	0.742	(0.498, 0.893)	4.332	(0.842, 14.437)	0.658	(0.496, 0.790)
ND (O)	0.264	(0.091, 0.562)	0.333	(-1.382, 0.813)	0.317	(0.154, 0.541)
ND (RRHC)	0.218	(0.038, 0.664)	0.483	(-2.663, 0.927)	0.202	(0.049, 0.553)
ND (SH)	0.322	(0.112, 0.642)	0.117	(-2.322, 0.765)	0.742	(0.528, 0.881)
ND (STP)	0.364	(0.183, 0.593)	0.060	(-0.585, 7.706)	0.523	(0.353, 0.687)
ND (UMRIRB)	0.560	(0.348, 0.752)	1.364	(-0.008, 4.636)	0.458	(0.308, 0.616)
ND (WD)	0.569	(0.230, 0.854)	1.453	(-0.447, 9.880)	0.445	(0.228, 0.685)
ND (WFP)	0.162	(0.038, 0.490)	0.640	(-0.784, 0.927)	0.197	(0.069, 0.446)
ND (WRB)	0.174	(0.058, 0.416)	0.610	(-0.322, 0.885)	0.336	(0.155, 0.584)

^aSee Table 2.30 for Natural Divisions.

Table 2.20. Continued.

Coefficients	Logistic regression				Beta regression	
	Beta	Confidence interval	Odds ratio	Confidence interval	Beta	Confidence interval
NWI class (FO)	0.179	(0.122, 0.254)	0.596	(0.370, 0.742)	0.299	(0.233, 0.375)
NWI class (LAKE)	0.293	(0.185, 0.431)	0.231	(-0.404, 0.579)	0.426	(0.325, 0.533)
NWI class (POND)	0.297	(0.195, 0.425)	0.216	(-0.372, 0.552)	0.466	(0.375, 0.559)
NWI class (RIVERINE)	0.211	(0.073, 0.476)	0.503	(-0.683, 0.853)	0.671	(0.439, 0.842)
River connectivity (1)	0.067	(0.030, 0.143)	0.866	(0.689, 0.942)		
River connectivity (2)	0.042	(0.018, 0.095)	0.918	(0.804, 0.965)		
River connectivity (3)	0.088	(0.041, 0.179)	0.822	(0.595, 0.921)		
River connectivity (4)	0.028	(0.012, 0.068)	0.946	(0.864, 0.978)		
River connectivity (5)	0.038	(0.016, 0.088)	0.927	(0.821, 0.970)		
River connectivity (6)	0.022	(0.009, 0.052)	0.959	(0.898, 0.983)		
River connectivity (7)	0.012	(0.005, 0.029)	0.978	(0.945, 0.991)		
Wetland complex (+)	0.515	(0.405, 0.624)	0.974	(0.264, 2.081)		

Table 2.21. Rankings of top models (ΔAIC_c) from the four model sets in the logistic and beta regression portion according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K), model weights (w_i) for predicting inundated non-persistent emergent vegetation (NPE) for migrating and breeding marsh birds during summer (mid-April – mid-June) using data collected during 2016 and 2017 throughout Illinois, USA.

Model Type	Model Set^a	<i>K</i>	<i>AIC_c</i>	ΔAIC_c	<i>w_i</i>
Logistic	Connectivity/complexity	25	590.25	0	1.00
	Land use/disturbance	21	642.54	52.29	0.00
	Management	24	646.15	55.90	0.00
	Water regime/climate	23	648.61	58.36	0.00
Beta Regression	Connectivity/complexity	25	-633.53	0.00	1.00
	Water regime/climate	21	-607.99	25.54	0.00
	Land use/disturbance (Null)	20	-602.57	30.96	0.00
	Management (Null)	20	-602.57	30.96	0.00

^aCovariates outlined in Table 2.4.

Table 2.22. Rankings of competing models ($\Delta AIC_c < 10$) in logistic and beta regression according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K) and model weights (w_i) for predicting inundated non-persistent emergent vegetation (NPE) for migrating and breeding marsh birds during summer (mid-April – mid-June) by connectivity/complexity covariates using data collected during 2016 and 2017 throughout Illinois, USA.

Model Type	Model ^a	K	AIC_c	ΔAIC_c	w_i	R^2
Logistic	ED + HI + ND + NWI + WC	25	590.25	0.00	0.31	0.22
	ED + HI + ND + NWI	24	590.54	0.29	0.27	0.22
	HI + ND + NWI + WC	24	591.09	0.84	0.21	0.22
	HI + ND + NWI	23	591.39	1.14	0.18	0.21
	ED + HI + ND + NWI + RC + WC (Global)	32	597.11	6.86	0.01	0.23
	HI + ND + NWI + RC + WC	31	597.25	7.00	0.01	0.23
	ED + HI + ND + NWI + RC	31	598.11	7.85	0.01	0.23
	HI + ND + NWI + RC	30	598.29	8.04	0.01	0.22
Beta Regression	HI + ND + NWI + WC	25	-633.53	0.00	0.40	0.27
	HI + ND + NWI	24	-632.78	0.75	0.27	0.26
	ED + HI + ND + NWI + WC	26	-632.22	1.31	0.21	0.27
	ED + HI + ND + NWI	25	-631.09	2.44	0.12	0.26

^aCovariates include: ED – Edge density (vegetation to open water metric associated with edge complexity; total edge in meters/ha); HI – Horizontal interspersions (ordinal complexity variable; ranging from 0=none/monotypic to 4=highly complex/diverse); ND – Natural Division (distinct ecoregions within Illinois); NWI – NWI class (dominant vegetation class or deepwater habitat; Cowardin et al. 1979); RC – River connectivity (ordinal variable representing connectivity to rivers or streams; 0=isolated/no connection apparent to 7=highly connected, permanent connection); WC – Wetland complex (presence/absence variable whether the NWI class in proximity to other wetland types).

Table 2.23. Coefficients, beta estimates, and confidence intervals ($\pm 85\%$) for both logistic and beta regression predicting the probability of and cover of inundated non-persistent emergent vegetation (e.g., NPE; moist-soil) by connectivity and complexity covariates and base variables, Natural Division (ND)^a and NWI class. Odds ratios and confidence intervals reported for logistic regression. Data were collected during 2016 and 2017 breeding and migration for marsh birds (mid-April – mid-June) through Illinois, USA. The Northeastern Morainal Natural Division always had inundated non-persistent emergent vegetation present (n=22), precluding confidence interval and odds ratio calculations.

Coefficients	Logistic regression				Beta regression	
	Beta	Confidence interval	Odds ratio	Confidence interval	Beta	Confidence interval
(Null)	0.322	(0.162, 0.537)	0.526	(-0.806, 0.161)	0.102	(0.064, 0.156)
Edge density	0.3221	(0.3218, 0.3224)	0.001	(0.0002, 0.003)		
Horizontal interspersions (1)	0.600	(0.444, 0.738)	2.166	(0.685, 4.951)	0.152	(0.101, 0.222)
Horizontal interspersions (2)	0.721	(0.581, 0.829)	4.452	(1.919, 9.185)	0.210	(0.145, 0.295)
Horizontal interspersions (3)	0.851	(0.750, 0.916)	11.057	(5.335, 21.949)	0.212	(0.147, 0.295)
Horizontal interspersions (4)	0.893	(0.804, 0.944)	16.580	(7.636, 34.790)	0.292	(0.209, 0.392)
ND (GP)	0.493	(0.251, 0.738)	1.050	(-0.294, 4.949)	0.066	(0.045, 0.096)
ND (IRM RSA)	0.408	(0.202, 0.652)	0.455	(-0.465, 2.956)	0.080	(0.058, 0.110)
ND (LMRB)	0.343	(0.171, 0.570)	0.101	(-0.566, 1.792)	0.096	(0.070, 0.131)
ND (MMRB)	0.193	(0.079, 0.400)	0.496	(-0.819, 0.403)	0.071	(0.046, 0.107)
ND (MWB)	0.079	(0.037, 0.159)	0.820	(0.601, 0.919)	0.075	(0.053, 0.104)
ND (NM)	1.000	.	.	.	0.106	(0.075, 0.148)
ND (O)	0.248	(0.101, 0.492)	0.307	(-0.764, 1.039)	0.067	(0.042, 0.105)
ND (RRHC)	0.294	(0.114, 0.573)	0.123	(-0.728, 1.829)	0.103	(0.063, 0.162)
ND (SH)	0.212	(0.084, 0.438)	0.435	(-0.806, 0.645)	0.082	(0.053, 0.124)
ND (STP)	0.225	(0.112, 0.400)	0.390	(-0.735, 0.403)	0.086	(0.062, 0.117)
ND (UMRIRB)	0.123	(0.060, 0.235)	0.704	(0.865, 0.352)	0.074	(0.055, 0.099)
ND (WD)	0.222	(0.065, 0.539)	0.400	(-0.854, 1.462)	0.108	(0.063, 0.177)
ND (WFP)	0.306	(0.122, 0.584)	0.071	(-0.708, 1.955)	0.039	(0.023, 0.064)
ND (WRB)	0.207	(0.097, 0.387)	0.451	(-0.773, 0.331)	0.074	(0.051, 0.106)

^aSee Table 2.30 for Natural Divisions.

Table 2.23. Continued.

Coefficients	Logistic regression				Beta regression	
	Beta	Confidence interval	Odds ratio	Confidence interval	Beta	Confidence interval
NWI class (FO)	0.304	(0.225, 0.397)	0.080	(-0.389, 0.387)	0.060	(0.051, 0.071)
NWI class (LAKE)	0.290	(0.196, 0.406)	0.138	(-0.485, 0.441)	0.067	(0.053, 0.084)
NWI class (POND)	0.233	(0.149, 0.347)	0.359	(-0.632, 0.118)	0.093	(0.075, 0.115)
NWI class (RIVERINE)	0.101	(0.053, 0.185)	0.762	(0.522, 0.882)	0.058	(0.041, 0.082)
Wetland complex (+)	0.416	(0.329, 0.508)	0.500	(0.035, 1.175)	0.124	(0.108, 0.142)

Table 2.24. Rankings of top models (ΔAIC_c) from the four model sets in the logistic regression portion according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K), model weights (w_i) for predicting mudflats for migrating shorebirds using data collected during 2016 and 2017 summer (mid-April – mid-June) and autumn (late-July – mid-Sept.) migrations through Illinois, USA.

Model Type	Model Set^a	<i>K</i>	<i>AIC_c</i>	ΔAIC_c	w_i
Logistic	Connectivity/complexity	31	1294.03	0.00	1.00
	Management	25	1379.65	85.62	0.00
	Land use/disturbance (Global)	25	1399.66	105.63	0.00
	Water regime/climate	21	1408.64	114.61	0.00
Beta Regression	Connectivity/complexity	22	-1439.07	0.00	0.38
	Land use/disturbance	22	-1438.91	0.16	0.35
	Management	21	-1437.04	2.03	0.14
	Water regime/climate	21	-1437.04	2.03	0.14

^aCovariates outlined in Table 2.4.

Table 2.25. Rankings of competing models ($\Delta AIC_c < 10$) in logistic and beta regression according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K) and model weights (w_i) for predicting mudflats for migrating shorebirds during summer (mid-April – mid-June) and autumn (late-July – mid-Sept.) migration by connectivity/complexity covariates using data collected during 2016 and 2017 in Illinois, USA.

Model Type	Model ^a	K	AIC_c	ΔAIC_c	w_i	R^2
Logistic	HI + ND + NWI + RC + SEAS	31	1294.03	0	0.48	0.18
	ED + HI + ND + NWI + RC + SEAS	32	1295.41	1.38	0.24	0.20
	HI + ND + NWI + RC + SEAS + WC	32	1296.00	1.98	0.18	0.18
	ED + HI + ND + NWI + RC + SEAS + WC (Global)	33	1297.41	3.38	0.09	0.2
Beta Regression	ND + NWI + SEAS + WC	22	-1439.07	0.00	0.44	0.31
	ED + ND + NWI + SEAS + WC	23	-1437.77	1.30	0.23	0.33
	ND + NWI + SEAS (Null)	21	-1437.04	2.03	0.16	0.31
	ED + ND + NWI + SEAS	22	-1435.94	3.13	0.09	0.32
	HI + ND + NWI + SEAS + WC	26	-1433.40	5.67	0.03	0.33
	HI + ND + NWI + SEAS	25	-1432.20	6.86	0.01	0.33
	ED + HI + ND + NWI + SEAS + WC	27	-1431.75	7.31	0.01	0.34
	ND + NWI + RC + SEAS + WC	29	-1431.51	7.55	0.01	0.35
	ED + HI + ND + NWI + SEAS	26	-1430.68	8.39	0.01	0.34
	ND + NWI + RC + SEAS	28	-1430.63	8.44	0.01	0.34
	ED + ND + NWI + RC + SEAS + WC	30	-1429.89	9.18	0.00	0.35

^aCovariates include: ED – Edge density (vegetation to open water metric associated with edge complexity; total edge in meters/ha); HI – Horizontal interspersion (ordinal complexity variable; ranging from 0=none/monotypic to 4=highly complex/diverse); ND – Natural Division (distinct ecoregions within Illinois); NWI – NWI class (dominant vegetation class or deepwater habitat; Cowardin et al. 1979); RC – River connectivity (ordinal variable representing connectivity to rivers or streams; 0=isolated/no connection apparent to 7=highly connected, permanent connection); WC – Wetland complex (presence/absence variable whether the NWI class in proximity to other wetland types); SEAS – Season (summer or autumn).

Table 2.26. Coefficients, beta estimates, and confidence intervals ($\pm 85\%$) for both logistic and beta regression predicting the probability of and cover of mudflats by connectivity and complexity covariates and base variables, Natural Division (ND)^a and NWI class. Odds ratios and confidence intervals reported for logistic regression. Data were collected during 2016 and 2017 summer (mid-April – mid-June) and autumn (late-July – mid-Sept.) migrations through Illinois, USA.

Coefficients	Logistic regression				Beta regression	
	Beta	Confidence interval	Odds ratio	Confidence interval	Beta	Confidence interval
(Null)	0.055	(0.029, 0.104)	0.942	(0.884, 0.971)	0.056	(0.041, 0.077)
Horizontal interspersion (1)	0.101	(0.066, 0.152)	0.926	(0.213, 2.057)	.	.
Horizontal interspersion (2)	0.161	(0.109, 0.231)	2.288	(1.100, 4.149)	.	.
Horizontal interspersion (3)	0.299	(0.214, 0.401)	6.301	(3.657, 10.444)	.	.
Horizontal interspersion (4)	0.333	(0.239, 0.444)	7.556	(4.361, 12.656)	.	.
ND (GP)	0.087	(0.048, 0.155)	0.635	(-0.146, 2.130)	0.064	(0.044, 0.093)
ND (IRM RSA)	0.013	(0.007, 0.024)	0.771	(0.587, 0.873)	0.068	(0.045, 0.101)
ND (LMRB)	0.060	(0.036, 0.098)	0.084	(-0.365, 0.850)	0.052	(0.037, 0.074)
ND (MMRB)	0.038	(0.020, 0.072)	0.321	(-0.654, 0.333)	0.057	(0.036, 0.089)
ND (MWB)	0.033	(0.020, 0.053)	0.422	(0.043, 0.6515)	0.054	(0.039, 0.073)
ND (NM)	0.034	(0.018, 0.062)	0.403	(-0.685, 0.132)	0.052	(0.034, 0.079)
ND (O)	0.094	(0.049, 0.174)	0.777	(-0.127, 2.617)	0.071	(0.046, 0.108)
ND (RRHC)	0.018	(0.008, 0.042)	0.679	(0.249, 0.863)	0.053	(0.029, 0.097)
ND (SH)	0.070	(0.037, 0.127)	0.286	(-0.338, 1.499)	0.059	(0.039, 0.088)
ND (STP)	0.053	(0.032, 0.085)	0.049	(-0.433, 0.595)	0.066	(0.048, 0.091)
ND (UMRIRB)	0.027	(0.017, 0.043)	0.525	(0.232, 0.706)	0.056	(0.041, 0.076)
ND (WD)	0.002	(0.000, 0.011)	0.960	(0.805, 0.992)	0.037	(0.009, 0.140)
ND (WFP)	0.061	(0.031, 0.118)	0.115	(-0.456, 1.282)	0.041	(0.026, 0.065)
ND (WRB)	0.051	(0.030, 0.086)	0.082	(-0.479, 0.617)	0.052	(0.036, 0.075)
NWI class (FO)	0.090	(0.070, 0.115)	0.691	(0.290, 1.218)	0.044	(0.037, 0.053)
NWI class (LAKE)	0.107	(0.078, 0.146)	1.061	(0.447, 1.935)	0.106	(0.085, 0.130)
NWI class (POND)	0.078	(0.056, 0.108)	0.451	(0.015, 1.075)	0.075	(0.058, 0.095)
NWI class (RIVERINE)	0.160	(0.104, 0.237)	2.256	(0.995, 4.314)	0.080	(0.062, 0.102)

^aSee Table 2.30 for Natural Divisions.

Table 2.26. Continued.

Coefficients	Logistic regression				Beta regression	
	Beta	Confidence interval	Odds ratio	Confidence interval	Beta	Confidence interval
River connectivity (1)	0.095	(0.060, 0.146)	0.786	(0.092, 1.923)	.	.
River connectivity (2)	0.134	(0.085, 0.205)	1.643	(0.581, 3.419)	.	.
River connectivity (3)	0.143	(0.091, 0.219)	1.859	(0.706, 3.791)	.	.
River connectivity (4)	0.188	(0.124, 0.275)	2.964	(1.426, 5.479)	.	.
River connectivity (5)	0.140	(0.091, 0.211)	1.795	(0.712, 3.563)	.	.
River connectivity (6)	0.106	(0.067, 0.164)	1.026	(0.221, 2.363)	.	.
River connectivity (7)	0.183	(0.121, 0.267)	2.834	(1.358, 5.234)	.	.
Season (Summer)	0.048	(0.040, 0.058)	0.135	(-0.060, 0.294)	0.083	(0.073, 0.094)
Wetland complex (+)	0.069	(0.060, 0.080)

^aSee Table 2.30 for Natural Divisions.

Table 2.27. Rankings of top models (ΔAIC_c) from the four model sets in the logistic and beta regression portion according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K), model weights (w_i) for predicting proportion shorebird habitat (i.e., mudflats and shallow inundation <10cm) for migrating shorebirds using data collected during 2016 and 2017 summer (mid-April – mid-June) and autumn (late-July – mid-Sept.) migrations through Illinois, USA.

Model Type	Model Set^a	K	AIC_c	ΔAIC_c	w_i
Logistic	Connectivity/complexity	26	501.82	0.00	0.996
	Management	22	514.36	12.54	1.89E-03
	Water regime/climate (Null)	20	515.91	14.09	8.69E-04
	Land use/disturbance (Null)	20	515.91	14.09	8.69E-04
Beta Regression	Land use/disturbance	20	-391.86	0.00	0.82
	Connectivity/complexity	20	-388.20	3.66	0.13
	Water regime/climate	23	-385.62	6.24	0.04
	Management	19	-382.50	9.36	0.01

^aCovariates outlined in Table 2.4.

Table 2.28. Rankings of competing models ($\Delta AIC_c < 10$) in logistic and beta regression according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K) and model weights (w_i) for predicting proportion shorebird habitat (i.e., mudflats and shallow inundation <10cm) for migrating shorebirds during summer (mid-April – mid-June) and autumn (late-July – mid-Sept.) migration by connectivity/complexity covariates using data collected during 2016 and 2017 in Illinois, USA.

Model Type	Model ^a	K	AIC_c	ΔAIC_c	w_i	R^2
Logistic	ED + HI + ND + NWI + SEAS + WC	26	501.82	0.00	0.28	0.22
	HI + ND + NWI + SEAS + WC	25	501.92	0.09	0.27	0.22
	ED + HI + ND + NWI + SEAS	25	502.31	0.49	0.22	0.21
	HI + ND + NWI + SEAS	24	502.58	0.76	0.19	0.21
	HI + ND + NWI + RC + SEAS + WC	32	507.30	5.48	0.02	0.23
	ED + HI + ND + NWI + RC + SEAS + WC (Global)	33	507.94	6.12	0.01	0.24
	HI + ND + NWI + RC + SEAS	31	508.89	7.07	0.01	0.23
	ED + HI + ND + NWI + RC + SEAS	32	509.31	7.49	0.01	0.23
	ED + ND + NWI + SEAS	21	511.64	9.82	0.00	0.18
Beta Regression	ND + NWI + Pwet_5km + SEAS	20	-391.86	0.00	0.59	0.32
	ND + NWI + ORAM + Pwet_5km + SEAS	21	-389.24	2.62	0.16	0.33
	LDI_5km + Pwet_5km + SEAS + ND + NWI	21	-389.05	2.81	0.14	0.32
	LDI_5km + ND + NWI + Pwet_5km + ORAM + SEAS	22	-386.35	5.51	0.04	0.33
	LDI_100 + LDI_5km + ND + NWI + ORAM + Pwet_5km + Pwet_100 + SEAS (Global)	24	-386.26	5.60	0.04	0.35
	LDI_100 + ND + NWI + SEAS	20	-383.92	7.94	0.01	0.30
	ND + NWI + SEAS (Null)	19	-382.5	9.36	0.01	0.26
	ND + NWI + Pwet_100 + SEAS	20	-382.41	9.45	0.01	0.28
	LDI_100m + ND + NWI + Pwet_100 + SEAS	21	-382.36	9.50	0.01	0.30

^aCovariates for logistic regression include: ED – Edge density (vegetation to open water metric associated with edge complexity; total edge in meters/ha); HI – Horizontal interspersion (ordinal complexity variable; ranging from 0=none/monotypic to 4=highly complex/diverse); ND – Natural Division (distinct ecoregions within Illinois); NWI – NWI class (dominant vegetation class or deepwater habitat; Cowardin et al. 1979); WC – Wetland complex (presence/absence variable whether the NWI class in proximity to other wetland types); SEAS – Season (summer or autumn). Beta regression covariates include: LDI_100m – Landscape Development Index for the surrounding 100-meter buffer; LDI_5km – Landscape Development Index for surrounding 5-km buffer; ORAM – Ohio Rapid Assessment Method score representing anthropogenic disturbance; Pwet_5km – Proportion of surrounding 5-km buffer comprised of wetlands; Pwet_100m – proportion of surrounding 100-meter buffer comprised of wetlands.

Table 2.29. Coefficients, beta estimates, and confidence intervals ($\pm 85\%$) for both logistic and beta regression predicting the probability of and cover of the probability of and cover of shorebird foraging habitat (i.e., mudflats and shallow inundation <10 cm) by connectivity and complexity covariates and base variables, Natural Division (ND)^a and NWI class. Odds ratios and confidence intervals reported for logistic regression. Data were collected during 2016 and 2017 summer (mid-April – mid-June) and autumn (late-July – mid-Sept.) migrations through Illinois, USA.

Coefficients	Logistic regression				Beta regression	
	Beta	Confidence Interval	Odds ratio	Confidence interval	Beta	Confidence Interval
(Null)	0.025	(0.005, 0.116)	0.974	(0.869, 0.995)	0.027	(0.007, 0.097)
Edge density	0.02551	(0.02548, 0.02554)	0.001	(0.000, 0.002)	.	.
Horizontal interspersions (1)	0.087	(0.049, 0.151)	2.651	(0.962, 5.793)	.	.
Horizontal interspersions (2)	0.103	(0.056, 0.183)	3.402	(1.255, 7.591)	.	.
Horizontal interspersions (3)	0.149	(0.078, 0.265)	5.706	(2.257, 12.807)	.	.
Horizontal interspersions (4)	0.175	(0.089, 0.316)	7.109	(2.716, 16.696)	.	.
ND (GP)	0.103	(0.019, 0.407)	3.406	(-0.260, 25.237)	0.040	(0.008, 0.173)
ND (IRM RSA)	0.218	(0.052, 0.586)	9.633	(1.092, 53.042)	0.073	(0.019, 0.245)
ND (LM RB)	0.259	(0.065, 0.637)	12.370	(1.669, 65.978)	0.081	(0.021, 0.264)
ND (MM RB)	0.356	(0.097, 0.741)	20.136	(3.088, 108.294)	0.048	(0.012, 0.173)
ND (MW B)	0.262	(0.067, 0.636)	12.558	(1.750, 65.855)	0.041	(0.011, 0.149)
ND (NM)	0.047	(0.007, 0.247)	0.874	(-0.720, 11.559)	0.063	(0.013, 0.257)
ND (O)	0.035	(0.004, 0.234)	0.367	(-0.840, 10.652)	0.028	(0.004, 0.175)
ND (RR HC)
ND (SH)	0.190	(0.038, 0.580)	7.962	(0.520, 51.830)	0.056	(0.013, 0.213)
ND (ST P)	0.183	(0.044, 1.143)	7.568	(0.679, 42.732)	0.065	(0.016, 0.224)
ND (UM RIR B)	0.312	(0.086, 0.686)	16.357	(2.602, 82.636)	0.040	(0.010, 0.143)
ND (WD)
ND (WFP)	0.169	(0.032, 0.560)	6.799	(0.251, 47.630)	0.033	(0.007, 0.146)
ND (WR B)	0.207	(0.048, 0.574)	8.974	(0.934, 50.427)	0.065	(0.016, 0.229)

^aSee Table 2.30 for Natural Divisions.

Table 2.29. Continued.

Coefficients	Logistic regression				Beta regression	
	Beta	Confidence Interval	Odds ratio	Confidence interval	Beta	Confidence Interval
NWI class (FO)	0.017	(0.011, 0.026)	0.355	(-0.015, 0.590)	0.023	(0.017, 0.031)
NWI class (LAKE)	0.026	(0.015, 0.045)	0.031	(-0.406, 0.790)	0.044	(0.023, 0.060)
NWI class (POND)	0.009	(0.005, 0.017)	0.660	(0.350, 0.822)	0.058	(0.039, 0.086)
NWI class (RIVERINE)	0.023	(0.011, 0.050)	0.100	(-0.999, 0.595)	0.020	(0.012, 0.034)
Proportion wetlands (5-km)	0.739	(0.345, 0.938)
Season (Summer)	0.005	(0.004, 0.007)	0.805	(0.720, 0.864)	0.021	(0.016, 0.027)
Wetland complex (+)	0.015	(0.009, 0.024)	0.420	(0.063, 0.641)	0.041	(0.030, 0.056)

^aSee Table 2.30 for Natural Divisions.

Table 2.30. The Natural Divisions of Illinois, Lake Michigan excepted (Schwegman 1973) and their acronyms used in analysis.

Natural Division	Acronym
Coastal Plain	CP
Grand Prairie	GP
Illinois River and Mississippi River Sands Areas	IRMRSA
Lower Mississippi River Bottomlands	LMRB
Middle Mississippi River Border	MMRB
Major Water Bodies	MWB
Northeastern Morainal	NM
Ozarks	O
Rock River Hill Country	RRHC
Shawnee Hills	SH
Southern Till Plain	STP
Upper Mississippi River and Illinois River Bottomlands	UMRIRB
Wisconsin Driftless	WD
Western Forest-Prairie	WFP
Wabash River Border	WRB

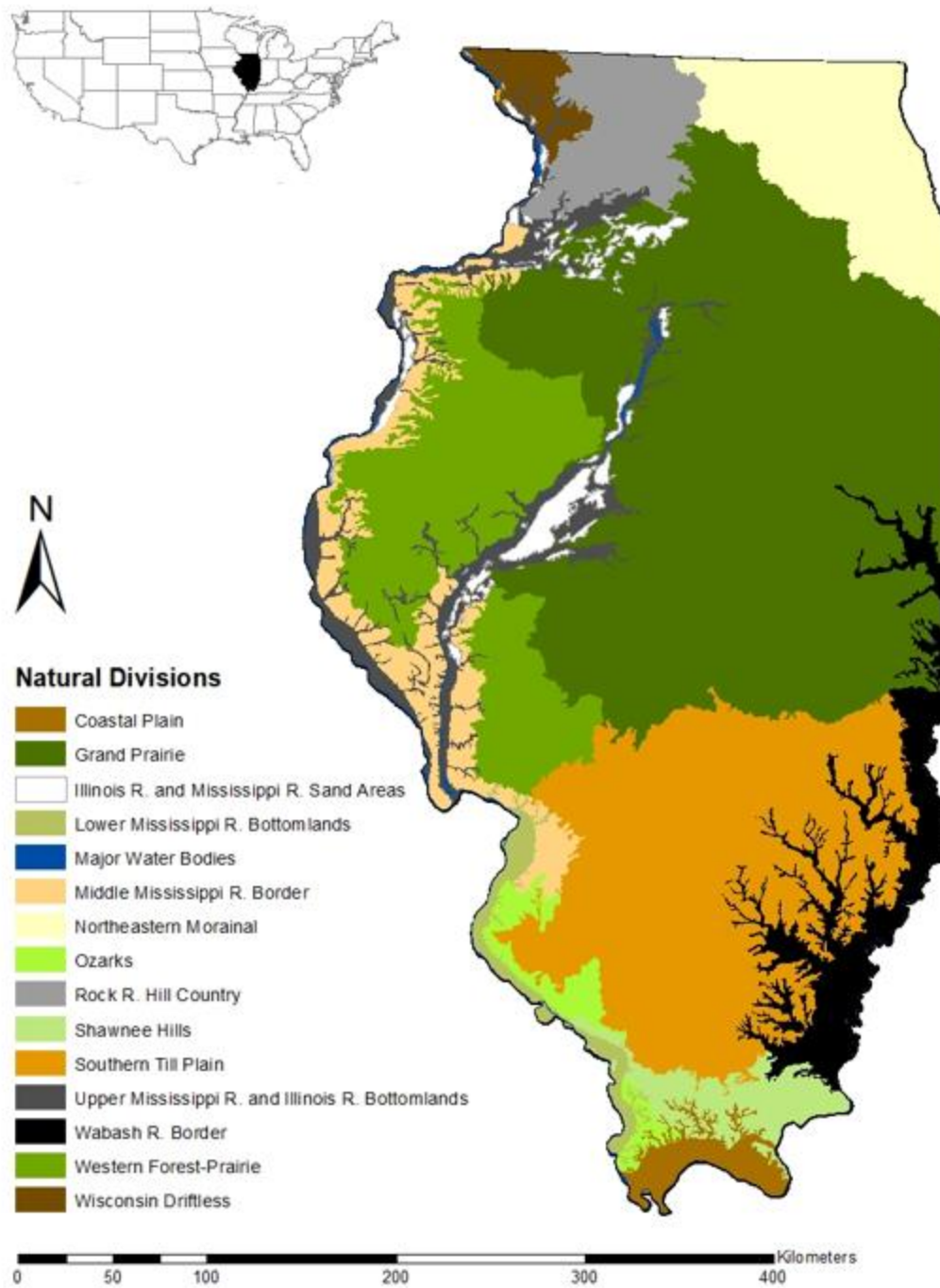


Figure 2.1. Illinois Natural Divisions (Schwegman 1973) used in plot selection to stratify the state by the distinct ecoregions and allocate survey effort relative to NWI wetland density. Wetland plots were surveyed by the Illinois Natural History Survey (INHS) and Southern Illinois University (SIU) in spring, summer, and autumn during 2016 – 2017 in Illinois, USA.

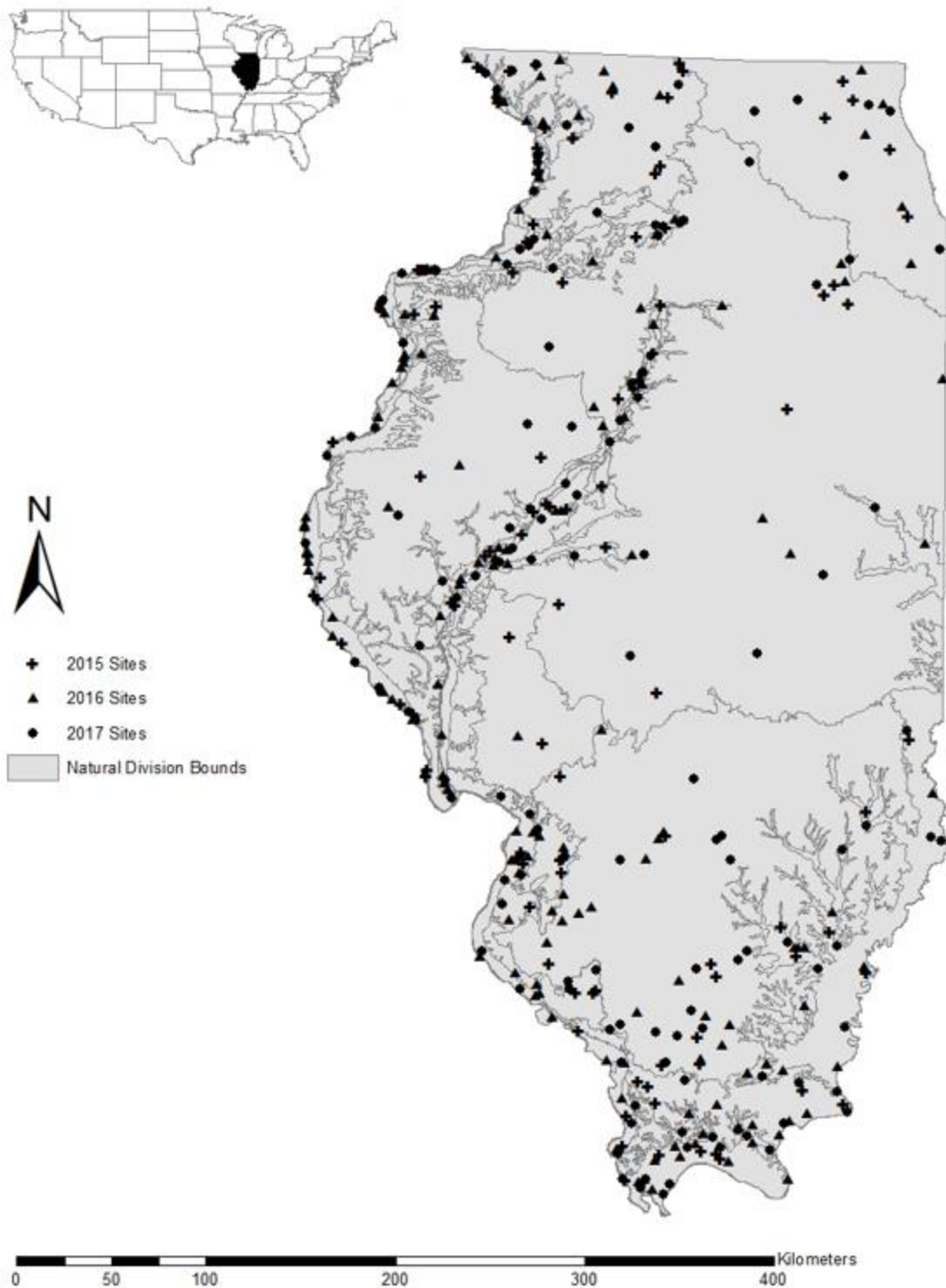


Figure 2.2. Locations of 25-ha wetland plots surveyed for suitable habitat in spring, summer, and autumn during 2015 – 2017 in Illinois, USA. Natural Division boundaries shown in gray.

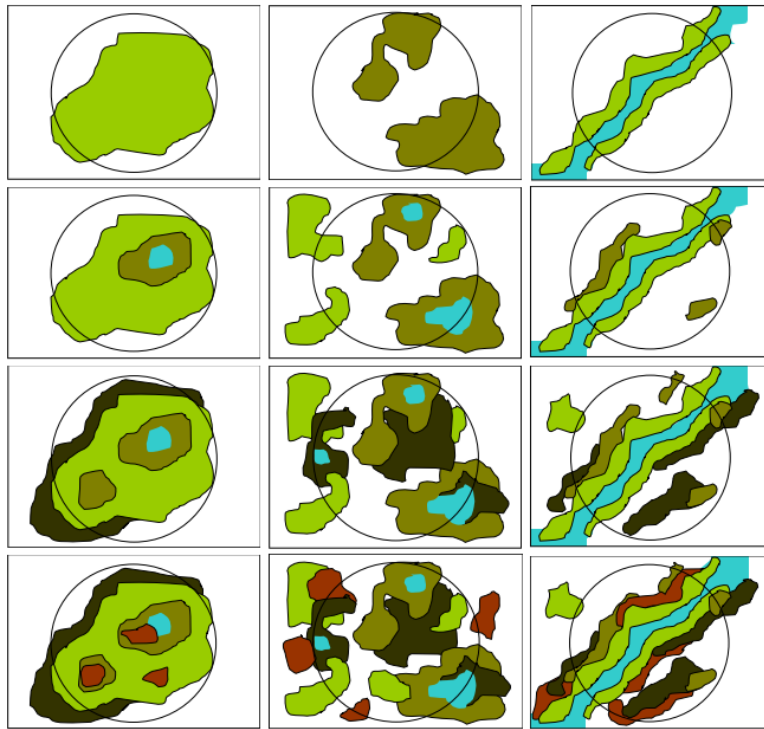


Figure 2.3. Graphic representation of horizontal interspersion (i.e., wetland complexity) factor. Columns reference alternative wetland shapes and rows represent low to high interspersion from top to bottom. I scored NWI classes during site visits during spring, summer, and autumn survey periods during 2016 – 2017 on horizontal interspersion ranging from 0= no interspersion (i.e., top of figure) to 4= highly interspersed and complex (i.e., bottom of figure). Retrieved from the field operations manual of 2011 National Wetland Condition Assessment (US EPA 2011).

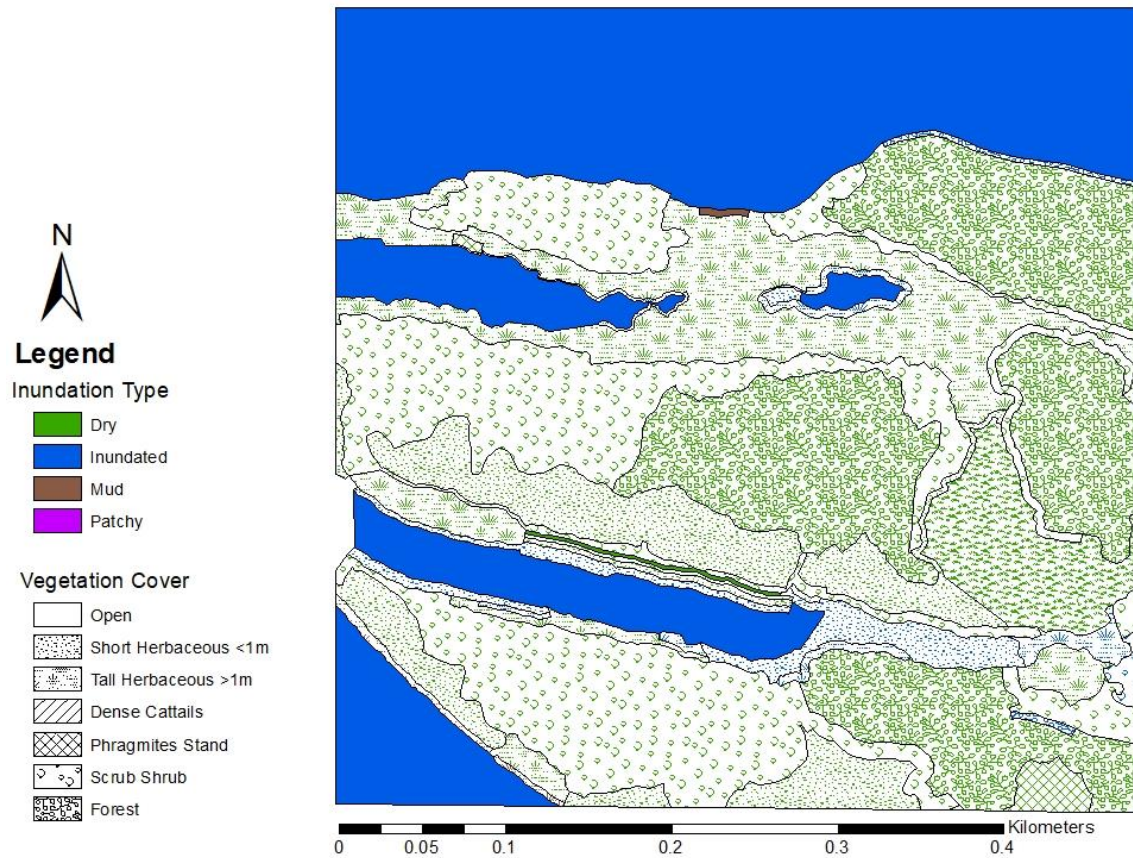
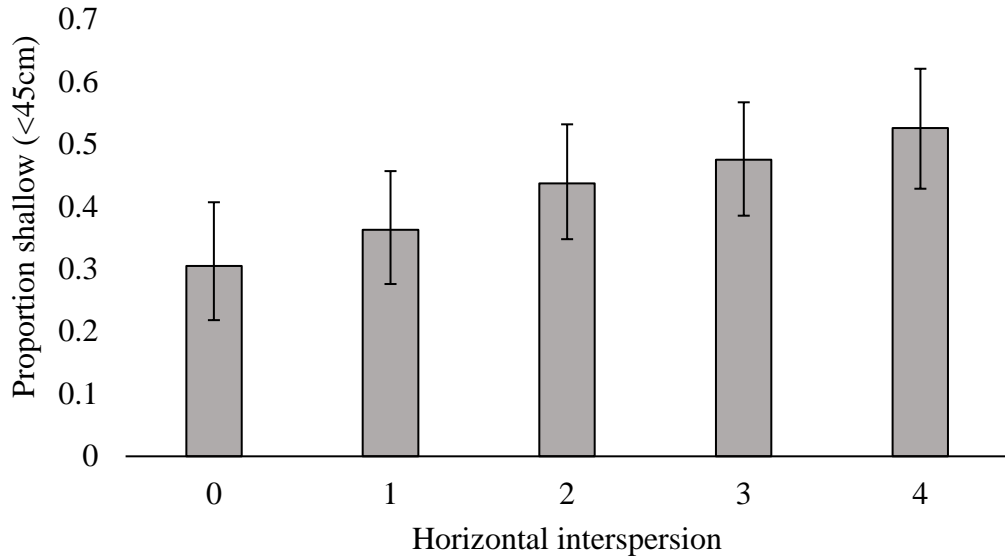
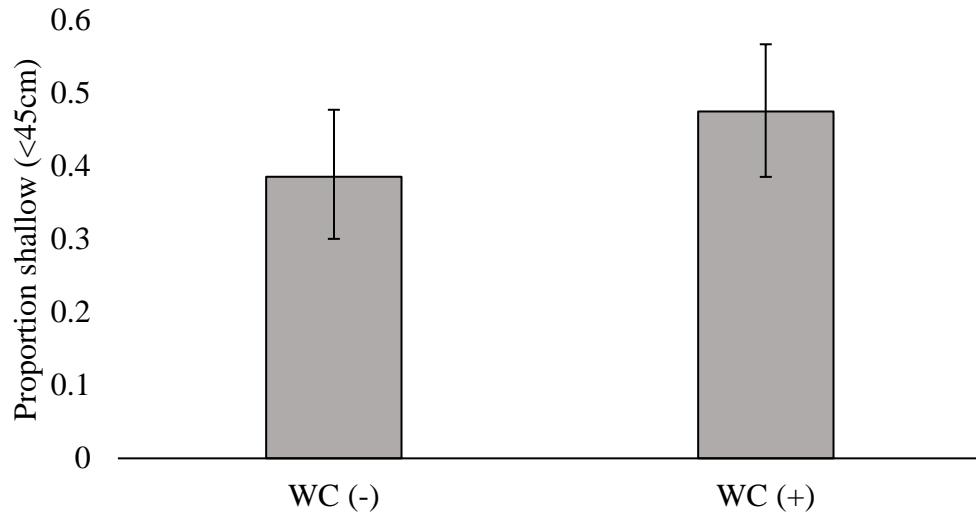


Figure 2.4. Example thematic map from a 25-ha plot surveyed in Spring 2017. I used survey tracks and observer notes to digitize inundation and vegetation boundaries in ArcMap 10.4 (ESRI 2011). Additional vegetation classes were mapped but not represented in this map (see Table 2.3).



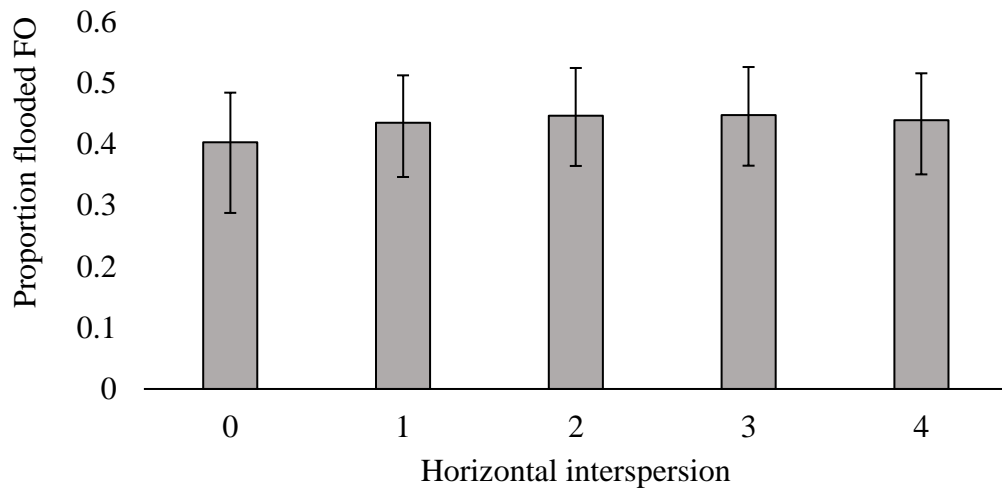
Natural Division (Coastal Plain)
 NWI Class (Freshwater Emergent)
 Wetland Complex (+)

Figure 2.5. Model predicted estimates (\pm 85% CI) from a beta regression of the proportion of shallow water (i.e., <45cm) available to dabbling ducks across horizontal interspersions classes during spring migrations (15 Feb – mid-April; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



Natural Division (Coastal Plain)
 NWI Class (Freshwater Emergent)
 Horizontal Interspersion (3)

Figure 2.6. Model predicted estimates (\pm 85% CI) for beta regression of proportion of shallow water (i.e., <45cm) available to dabbling ducks by wetland complex during spring migrations (15 Feb – mid-April; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



Natural Division (Coastal Plain)
 NWI class (Freshwater Forested/Scrub-Shrub)
 Wetland Complex (+)

Figure 2.7. Conditional model predicted estimates (\pm 85% CI) for proportion of inundated forest (FO) available to dabbling ducks by horizontal interspersion (i.e., ordinal variable representing complexity; ranging from none=0 to high=4) during spring migrations (15 Feb – mid-April; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.

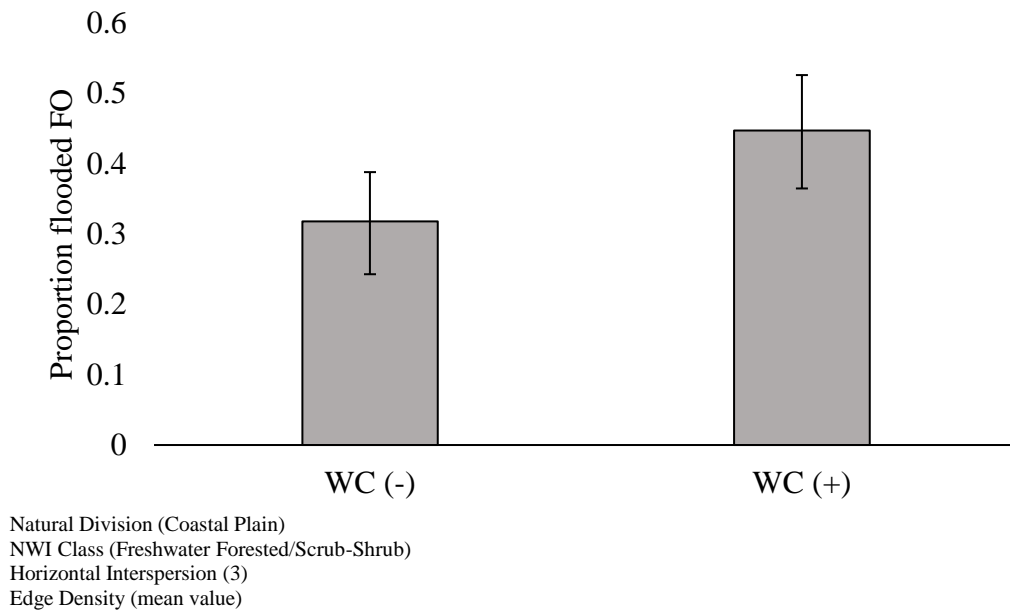
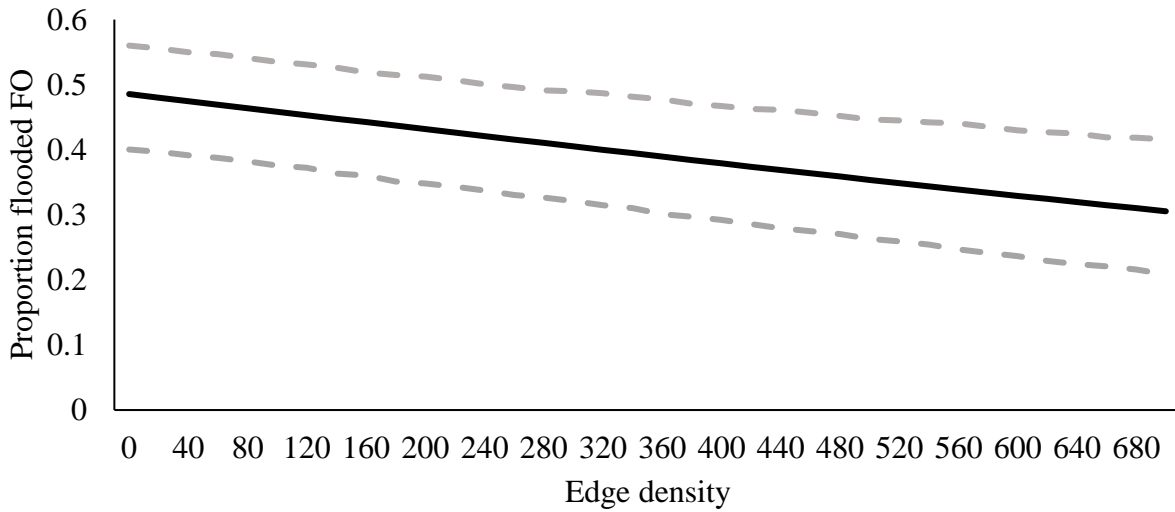


Figure 2.8. Conditional model predicted estimates (\pm 85% CI) for proportion of inundated forest (FO) available to dabbling ducks by the variable wetland complex (i.e., presence/absence of connectivity to other wetland types) during spring migrations (15 Feb – mid-April; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



Natural Division (Coastal Plain)
 NWI Class (Freshwater Forested/Scrub-Shrub)
 Wetland Complex (+)
 Horizontal Interspersion (3)

Figure 2.9. Conditional model predicted estimates (\pm 85% CI) for proportion of inundated forest (FO) available to dabbling ducks by edge density during spring migrations (15 Feb – mid-April; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.

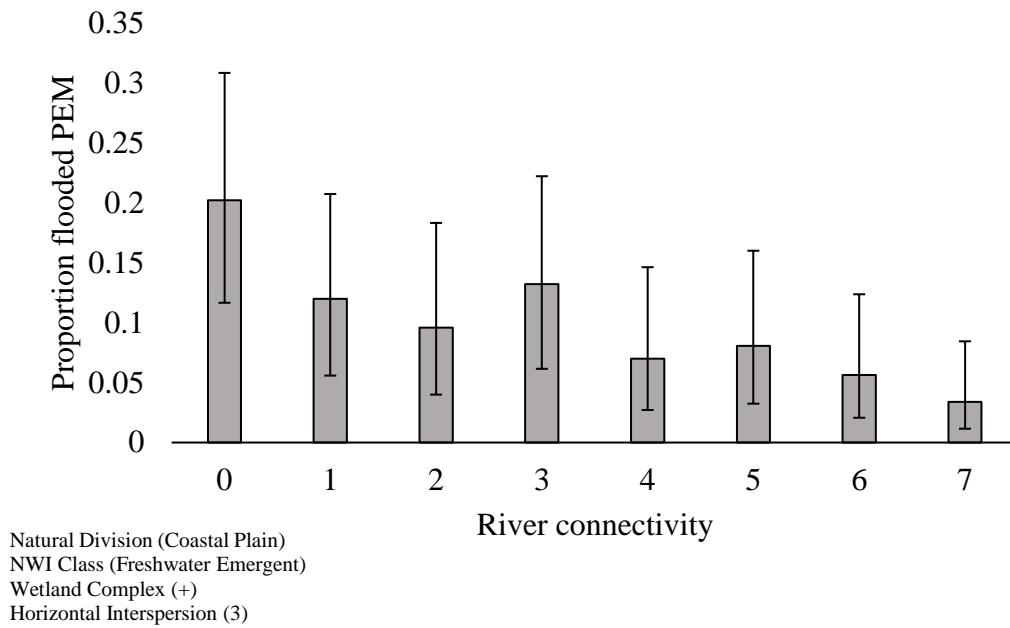


Figure 2.10. Conditional model predicted estimates (\pm 85% CI) for proportion of inundated dense persistent emergent vegetation (PEM) for migrating and breeding marsh birds by river connectivity (ordinal variable ranging from 0=not connected to rivers/streams to 7=permanent connection) during summer (mid-April – mid-June; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.

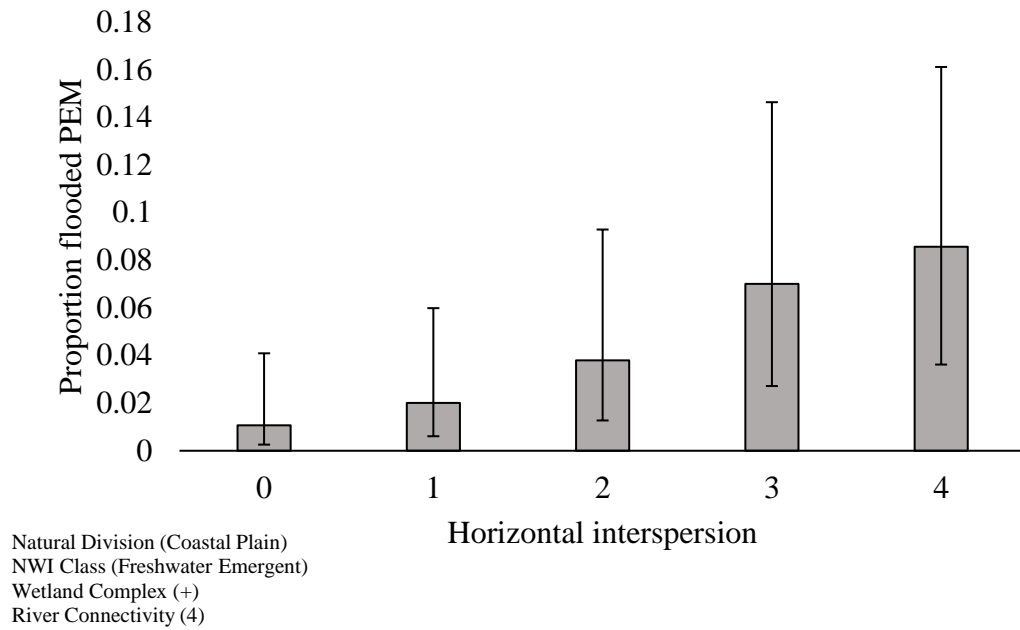


Figure 2.11. Conditional model predicted estimates (\pm 85% CI) for proportion of inundated dense persistent emergent vegetation (PEM) for migrating and breeding marsh birds by horizontal interspersions (i.e., ordinal variable representing complexity; ranging from none=0 to high=4) during summer (mid-April – mid-June; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.

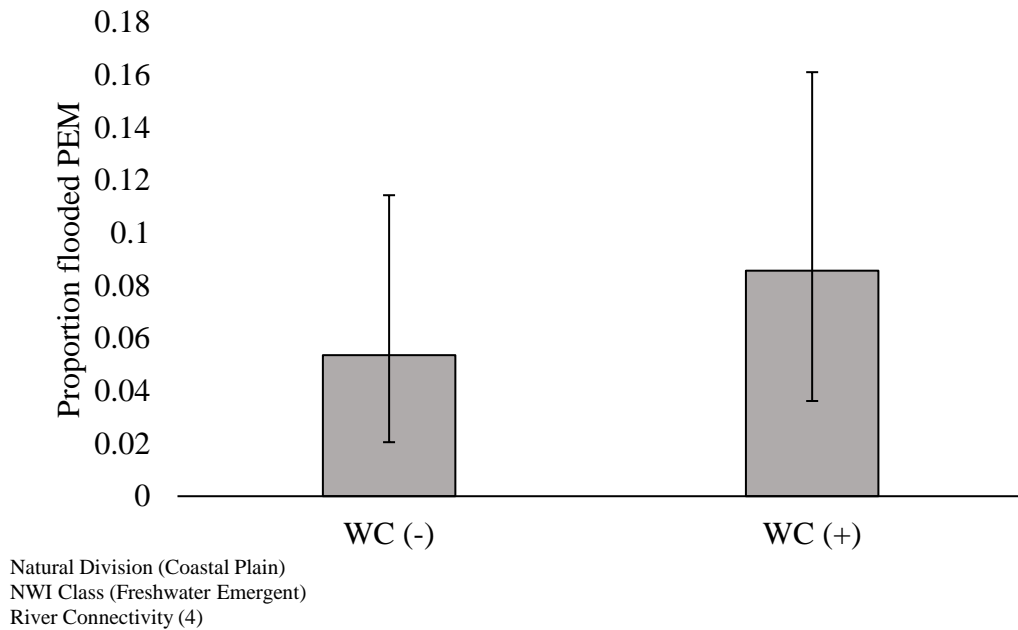


Figure 2.12. Conditional model predicted estimates (\pm 85% CI) for proportion of inundated dense persistent emergent vegetation (PEM) for migrating and breeding marsh birds by the variable wetland complex (i.e., presence/absence of connectivity to other wetland types) during summer (mid-April – mid-June; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.

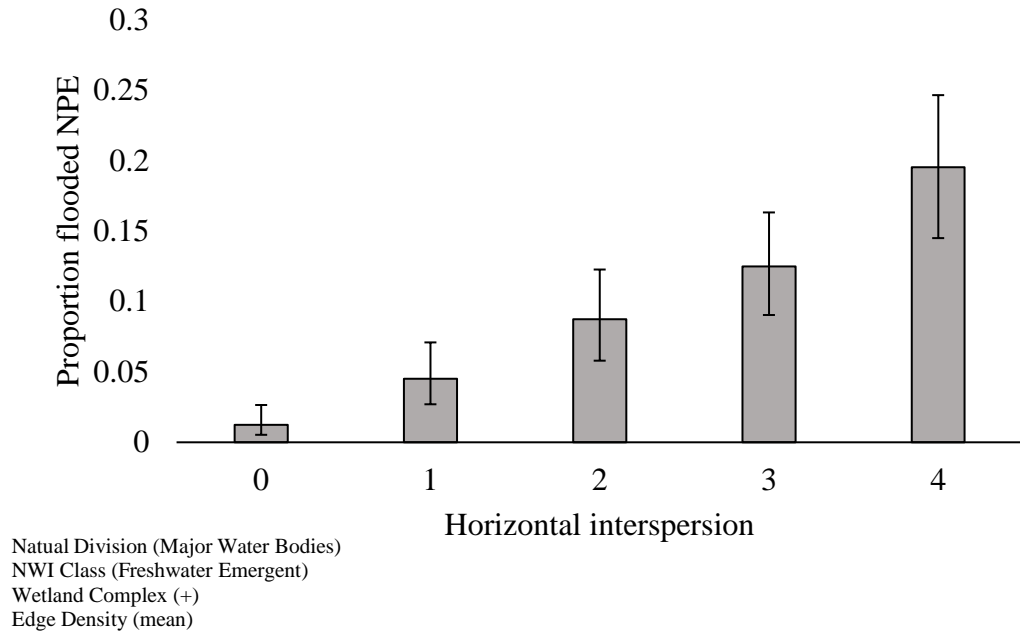
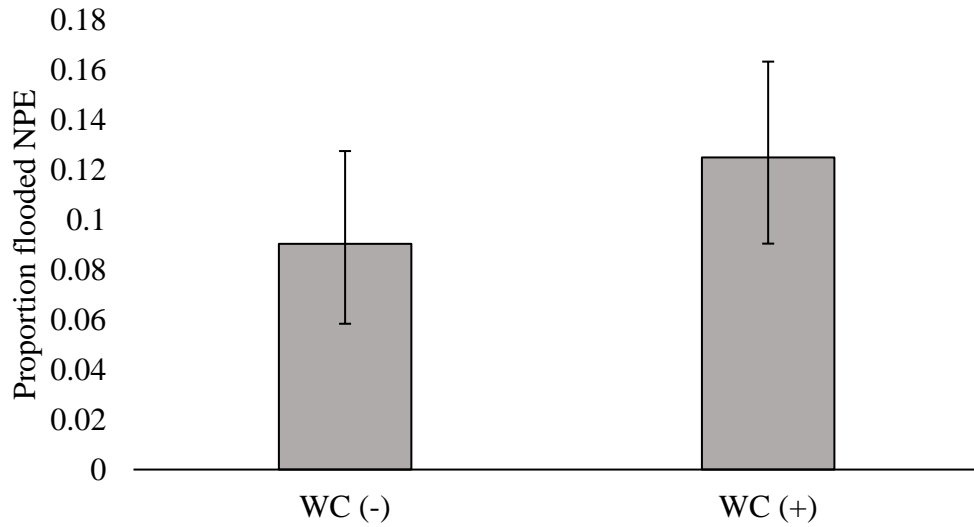
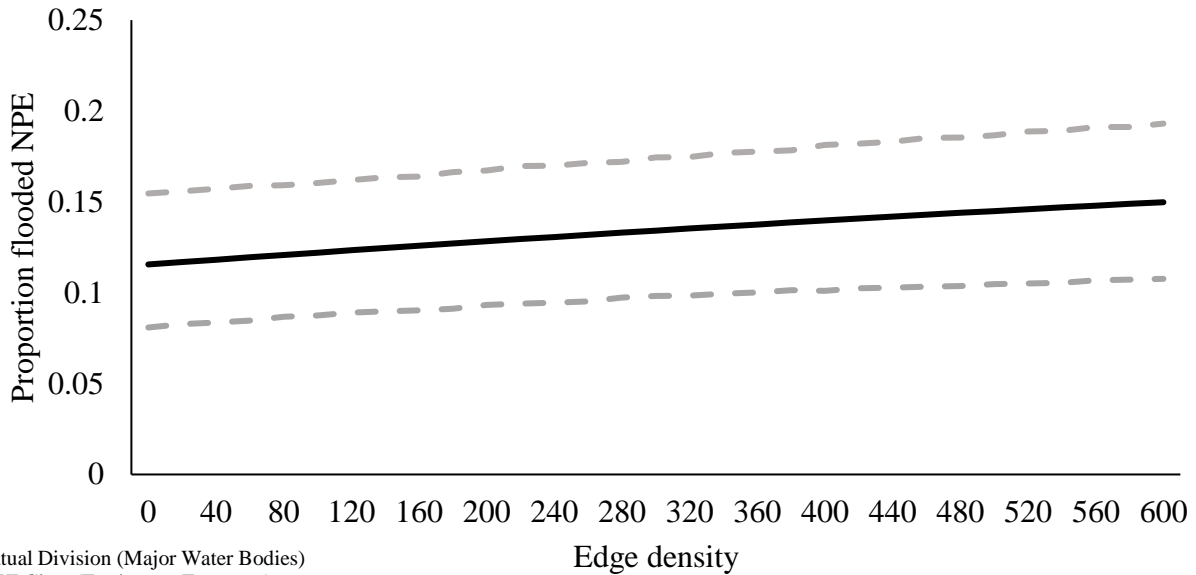


Figure 2.13. Conditional model predicted estimates (\pm 85% CI) for proportion of inundated non-persistent emergent vegetation (NPE) for migrating and breeding marsh birds by horizontal interspersions (i.e., ordinal variable representing complexity; ranging from none=0 to high=4) during summer (mid-April – mid-June; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



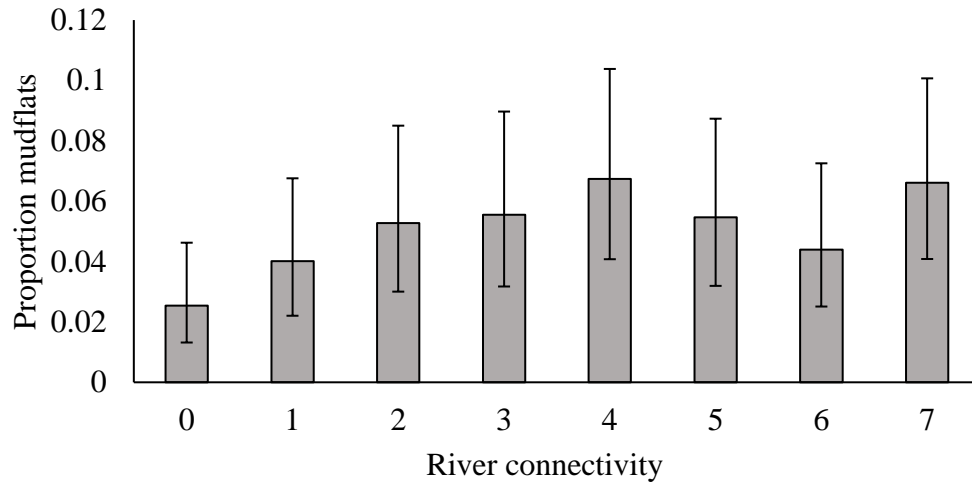
Natural Division (Major Water Bodies)
 NWI Class (Freshwater Emergent)
 Edge Density (mean)
 Horizontal Interspersion (3)

Figure 2.14. Conditional model predicted estimates (\pm 85% CI) for proportion of inundated non-persistent emergent vegetation (NPE) for migrating and breeding marsh birds by the variable wetland complex (i.e., presence/absence of connectivity to other wetland types) during summer (mid-April – mid-June; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



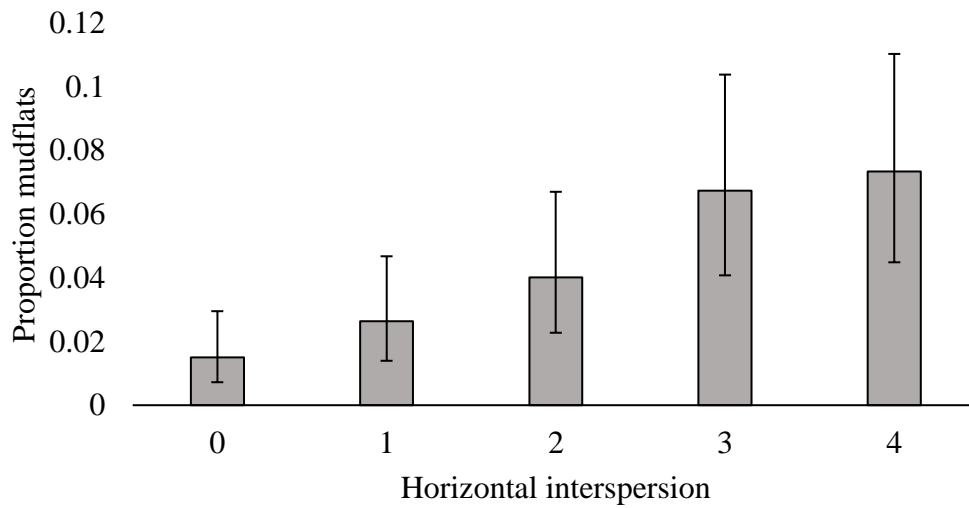
Natural Division (Major Water Bodies)
 NWI Class (Freshwater Emergent)
 Edge Density (mean)
 Wetland Complex (+)

Figure 2.15. Conditional model predicted estimates (\pm 85% CI) for proportion of inundated non-persistent emergent vegetation (NPE) for migrating and breeding marsh birds by edge density (total edge/ha) during summer (mid-April – mid-June; 2016 and 2017) in Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



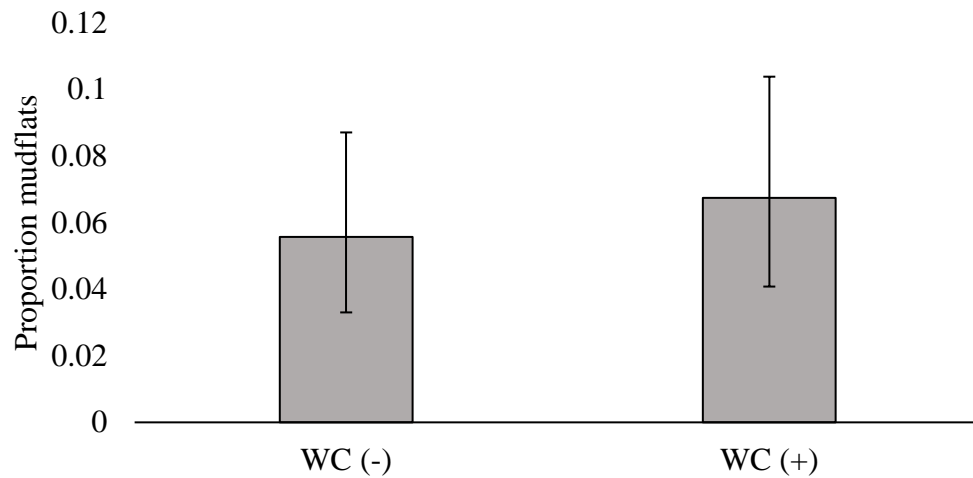
Natural Division (Illinois & Mississippi River Sands Area)
 NWI Class (Lake)
 Wetland Complex (+)
 Horizontal Interspersion (3)
 Season (Autumn)

Figure 2.16. Conditional model predicted estimates (\pm 85% CI) for proportion of mudflat for migrating shorebirds by river connectivity (ordinal variable ranging from 0=disconnected from rivers/streams to 7=permanent connection) during summer (mid-April – mid-June) and autumn (late-July – mid-Sept.) migrations in 2016 and 2017 throughout Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



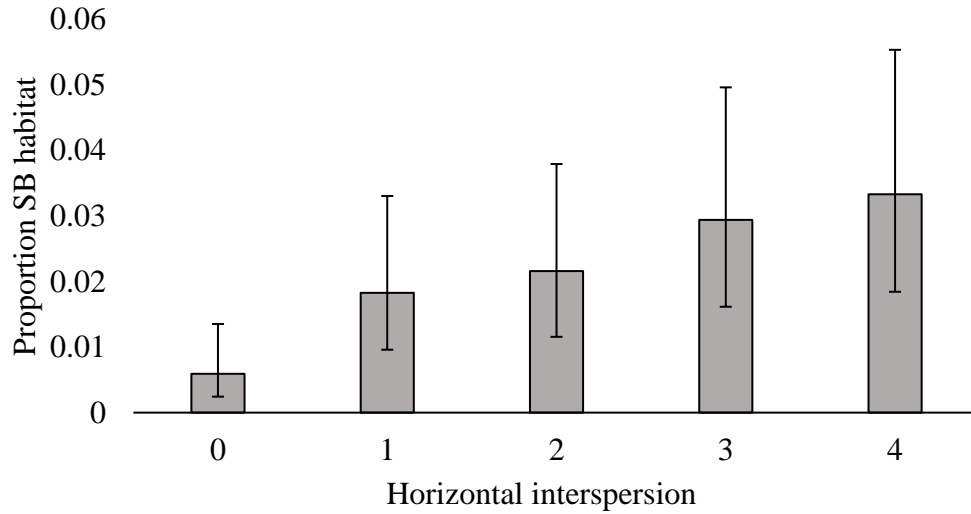
Natural Division (Illinois & Mississippi River Sands Area)
 NWI Class (Lake)
 Wetland Complex (+)
 Season (Autumn)

Figure 2.17. Conditional model predicted estimates (\pm 85% CI) for proportion mudflats for migrating shorebirds by horizontal interspersion (i.e., ordinal variable representing complexity; ranging from none=0 to high=4) during summer (mid-April – mid-June) and autumn migration (late-July – mid-Sept.) during 2016 and 2017 throughout Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



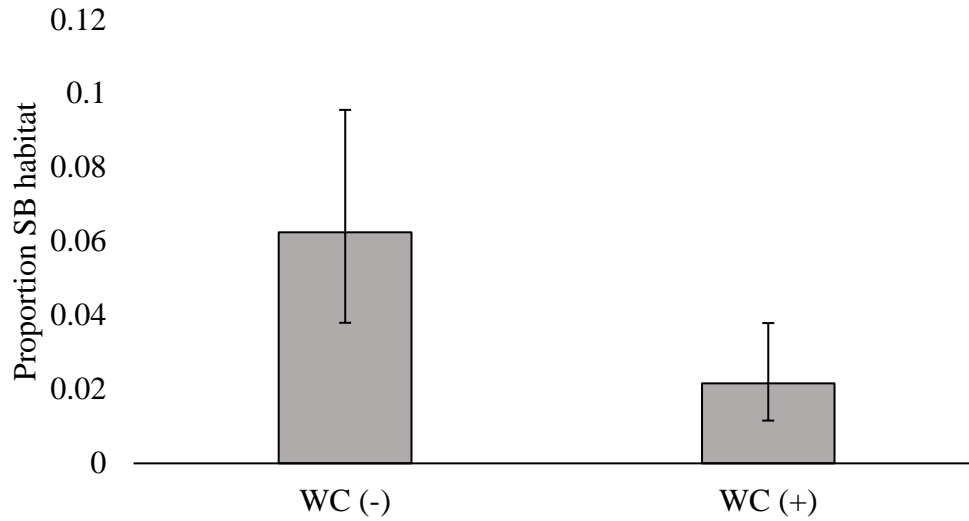
Natural Division (Illinois & Mississippi River Sands Area)
 NWI Class (Lake)
 Horizontal Interspersion (3)
 River Connectivity (4)
 Season (Autumn)

Figure 2.18. Conditional model predicted estimates (\pm 85% CI) for proportion mudflats for migrating shorebirds by the variable wetland complex (i.e., presence/absence of connectivity to other wetland types) during summer (mid-April – mid-June) and autumn migration (late-July – mid-Sept.) during 2016 and 2017 throughout Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



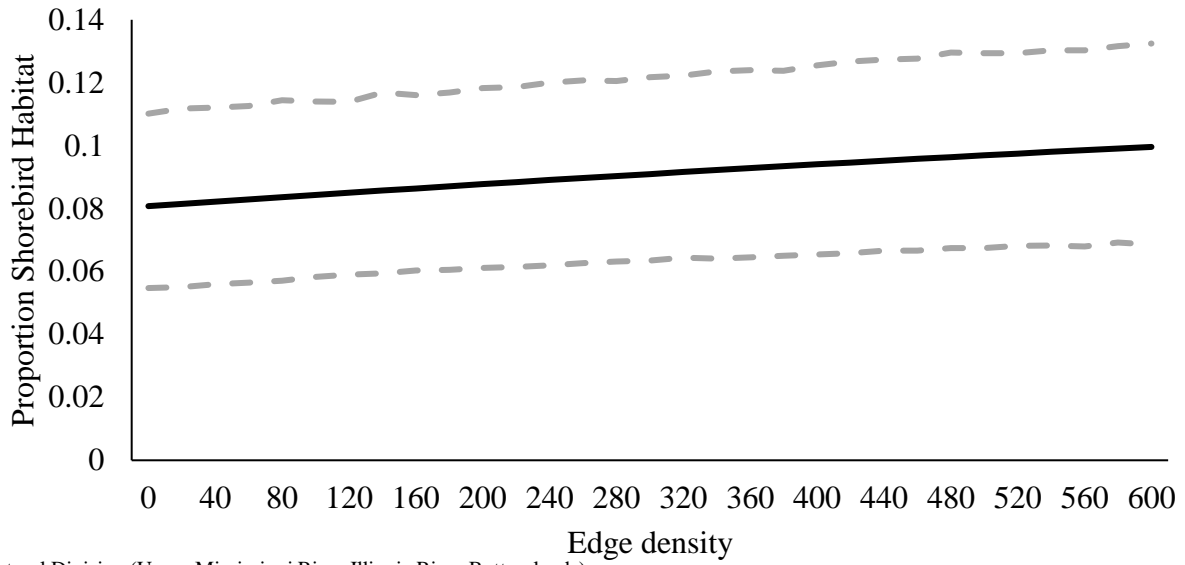
Natural Division (Upper Mississippi & Illinois River Bottomlands)
 NWI Class (Lake)
 Wetland Complex (+)
 Edge Density (mean)
 Season (Summer)

Figure 2.19. Conditional model predicted estimates (\pm 85% CI) for proportion shorebird (SB) habitat (i.e., mudflats and shallow inundation <10cm) for migrating shorebirds by horizontal interspersions (i.e., ordinal variable representing complexity; ranging from none=0 to high=4) during summer (mid-April – mid-June) and autumn migration (late-July – mid-Sept.) during 2017 throughout Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



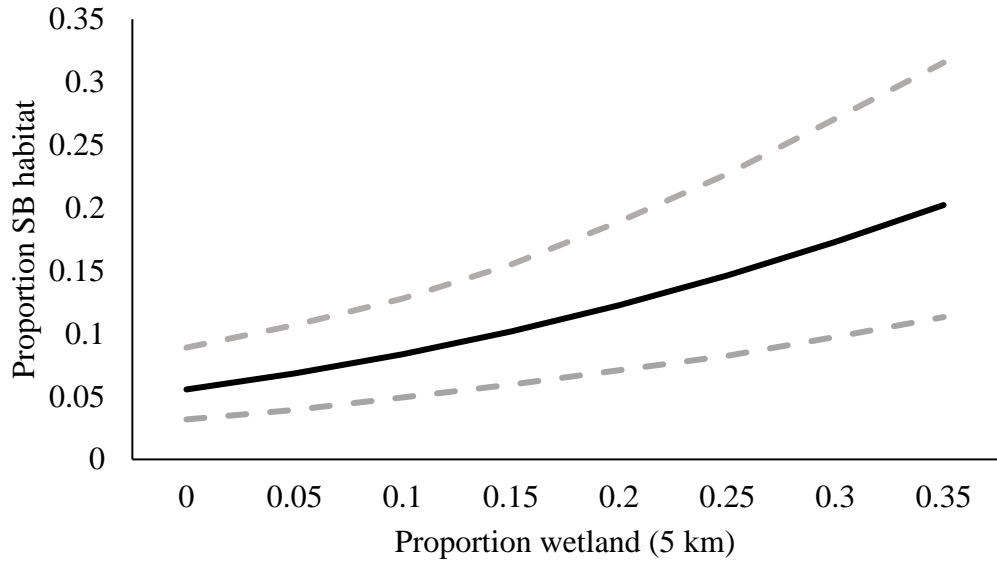
Natural Division (Upper Mississippi & Illinois River Bottomlands)
 NWI Class (Lake)
 Edge Density (mean)
 Season (Summer)

Figure 2.20. Conditional model predicted estimates (\pm 85% CI) for proportion shorebird (SB) habitat (i.e., mudflats and shallow inundation <10cm) for migrating shorebirds by the variable wetland complex (i.e., presence/absence of connectivity to other wetland types) summer (mid-April – mid-June) and autumn migration (late-July – mid-Sept.) during 2017 throughout Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



Natural Division (Upper Mississippi River Illinois River Bottomlands)
 NWI Class (Lake)
 Wetland Complex (+)
 Horizontal Interspersion (3)

Figure 2.21. Conditional model predicted estimates (\pm 85% CI) for proportion shorebird habitat (i.e., mudflats and shallow inundation <10cm) for migrating shorebirds by edge density during summer (mid-April – mid-June) and autumn migration (late-July – mid-Sept.) during 2017 throughout Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.



Natural Division (Illinois River and Mississippi River Sands Area)
 NWI Class (Lake)
 Season (Autumn)

Figure 2.22. Conditional model predicted estimates (\pm 85% CI) for proportion shorebird (SB) habitat (i.e., mudflats and shallow inundation <10cm) for migrating shorebirds by proportion wetland within the 5-km buffer during summer (mid-April – mid-June) and autumn migration (late-July – mid-Sept.) during 2017 throughout Illinois, USA. Predicted estimates were calculated with alternate variables held at constant levels shown in bottom left of figure.

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CHAPTER 3: LOCAL AND LANDSCAPE CHARACTERISTICS THAT INFLUENCE WATERBIRD USE OF WETLANDS DURING SPRING MIGRATION IN ILLINOIS

3.1 ABSTRACT

Habitat quality of wetlands used during migration stopovers influence survival and body condition of some waterbird species and may carry over to the breeding period. However, relatively little is known about relationships between habitat characteristics and waterbird use during spring migration, despite the implication that habitat during spring migration is likely limiting in the Midwest, USA. I enumerated waterfowl and other waterbirds by species using repeated aerial surveys of polygons included in the National Wetland Inventory in Illinois during spring migration 2016–2017 to identify species-habitat relationships. During a separate ground visit, I mapped surface water inundation, visually estimated vegetation cover, and assessed wetland management and anthropogenic stressors. Local wetland hydrological and vegetation cover were the most important predictors of dabbling duck and other waterbird (diving ducks, geese, herons, coots, etc.) densities. Dabbling duck density increased by 34.8% and other waterbird density increased by 44.6% for every 10% increase in the proportion of wetland areas inundated by surface water. Furthermore, dabbling duck density increased by 26.5% for every 10% increase in the proportion of shallow (<45 cm) surface water. Dabbling duck and other waterbird density was negatively related to woody and herbaceous vegetation cover, which could be due to multiple factors including perceived risk associated with more vegetated wetlands or potentially limited food availability. While many factors influence waterbird use and selection, this study emphasizes the importance of providing surface inundation and regions of shallow water during spring migration in Illinois. Restoration efforts should incorporate controllable hydrology and potentially avoid instances where hydrology is largely unregulated in large river floodplains.

3.2 INTRODUCTION

Population trends of wetland-dependent migratory birds often parallel the amount and suitability of habitat available during the breeding season (Weller and Spatcher 1965, Howe et al. 1989, Conway et al. 1994, Baldassarre 2014). The Midwest, USA, is among the regions with the greatest percentage loss of wetlands (Tiner 1984), and this often considered a primary reason for concomitant declines in several species of waterbirds (e.g., Gibbs et al. 1992, Meanley 1992, Lor

and Malecki 2002, Ward et al. 2010). For example, Illinois has lost >85% of its original 3.2 million ha of wetlands (Dahl 2006) and emergent wetlands that provide critical waterbird habitat are still decreasing (USFWS 2015). Additionally, some waterbirds may be impacted by habitat conditions during the non-breeding season (e.g., spring and fall migrations; Morrison 2006, Skagen et al. 2008), if carryover effects from migration stopovers manifest in breeding success. Recent studies have demonstrated that the suitability of stopover wetlands influences survival and body condition of some species during migration as they prepare for breeding (Anteau and Afton 2004, Morrison 2006, Devries et al. 2008). Furthermore, in a review of spring migration in Anatidae (i.e., waterfowl), Arzel et al. (2006) asserted that relatively little is known about stopover habitat and posited that habitat during spring migration is likely limiting. Others have documented that certain habitat resources may be limiting in some years or regions in the Midwest for some species (Straub et al. 2012, Hagy et al. 2017a), and these deficiencies could have implications for breeding populations (Anteau and Afton 2011). Consequently, conservation partnerships, such as the Upper Mississippi River and Great Lakes Region Joint Venture [UMRGLR-Joint Venture (Soulliere et al. 2017), consider spring migration to be a critical pre-breeding period for many species of wetland-dependent birds, including waterfowl.

Local wetland characteristics such as hydrology and vegetation cover impact wetland use by migratory birds. For instance, dabbling ducks (Anatini) feed primarily in shallow water (<45 cm; McGilvrey 1966, Sousa and Farmer 1983) where seeds and other benthic resources are accessible (Isola et al. 2000, Taft et al. 2002). Some researchers have shown that ducks respond to food density (Fredrickson and Heitmeyer 1988), but others have failed to show this relationship (Hagy and Kaminski 2015). However, habitat use may be regionally or seasonally variable as Yetter et al. (2018) found that mallards in the Illinois River Valley used open water most frequently, followed by row-crop agriculture, then herbaceous and woody wetlands during autumn migration. On the other hand, diving ducks (Aythyini), mergansers (Mergini), and other diving waterbirds select areas of deep, open water (Murken et al. 1997) to access mollusks (e.g., fingernail clams *Sphaerium transversum*), invertebrates, seeds, and other resources (Thompson 1973). During spring migration, canvasbacks (*Aythya valisineria*) select large water bodies characterized by extensive beds of submerged vegetation (Baldassarre 2014) such as the Keokuk Pool (i.e., Pool 19) on the Mississippi River, with an average water depth of 2.7 m and a

maximum of 11.0 m (Thompson 1973). Similarly, Anteau and Afton (2009) found that spring migrating lesser scaup (*Aythya affinis*) fed in wetlands with a large open water zone. Other divers such as ring-necked ducks (*Aythya collaris*) and hooded mergansers (*Lophodytes cucullatus*) selected shallow freshwater marshes and a variety of forested wetlands and riparian areas respectively (Baldassarre 2014). American coots (*Fulica americana*) migrating in spring often used ponds and small wetlands (Heitmeyer 1980), whereas coots migrating in autumn fed in shallow, vegetated aquatic habitats sympatrically with several waterfowl species (Eddleman 1983). While wetland use and selection vary between species and between seasons, selection of particular wetlands or vegetation types is likely related to a combination of factors (Murkin et al. 1997).

Wetland structure or complexity may impact wetland use by waterbirds. Waterbird diversity and abundance is often greatest when hemi-marsh conditions exist (i.e., approximate 50:50 ratio of open water and emergent vegetation; Weller and Spatcher 1965, Kaminski and Prince 1981, Smith et al. 2004). In northeast Illinois, Ward et al (2010) found that the decline in hemi-marsh vegetation was likely responsible for seven waterbird population declines, including blue-winged teal (*Anas discors*), ruddy duck (*Oxyura jamaicensis*), and American coot. Both hemi-marsh and patches of other vegetation types are important for isolating waterfowl pairs during spring (Murkin et al. 1982). Likewise, vegetation and open water interfaces are important to diving duck habitat selection (Murkin et al. 1997) and are associated with increased waterbird diversity (e.g., edge density; Rehm and Baldassarre 2007).

Due to decreases in wetland cover across North America (Dahl 2006), wetlands are often managed to provide suitable hydrology and vegetation cover for wildlife (Fredrickson and Taylor 1982). Waterbird use and density is often higher in wetlands with active hydrological and/or vegetation management (Kaminski et al. 2009). For instance, Pankau (2008) and O'Neal et al. (2008) found migrant waterbird use-days were significantly greater in restored wetlands with actively managed hydrology compared to wetlands with passive management or those that were unmanaged for hydrology.

In addition to local impacts from vegetation cover, structure, or wetland management, landscape characteristics such as connectivity to other wetland types or to rivers and streams may

impact waterbird use or selection. Furthermore, highly connected wetlands or groups of wetlands in close proximity, known as wetland complexes, support various waterbird species and increase waterbird diversity by supplying varied microhabitats and areas for forage or other physical requirements (Weller 1988, Fairbairn and Dinsmore 2001, Beatty et al. 2014a). In a habitat selection study, Beatty et al. (2014a) found that mallards selected resource units adjacent to emergent wetlands, open water, and woody wetlands during spring migration. Wetland landscapes are an essential aspect of staging, stopover, and wintering habitat for migratory waterbirds (Webb et al. 2010, Pearse et al. 2012, Beatty et al. 2014a). For instance, Beatty et al. (2014a) found that large-scale relocation movements in mallard habitat selection focused on wetlands at a landscape-scale during the non-breeding period (i.e., both spring and autumn migration), and Webb et al. (2010) found that area of semi-permanent wetlands within 10 km was important in models predicting dabbling duck abundance during migration. Additionally, large river systems, including associated lakes and floodplain wetlands, rivers and creeks, provide crucial habitat for waterbirds (Havera 1999, Connor and Gabor 2006). However, wetland connectivity to river systems may have variable impacts on waterbird use (Stafford et al. 2007). Xia et al. (2016) explored differences in waterbird abundance and diversity among isolated, freely connected, and controlled hydrological connections to a large river system. Their results indicated alteration to hydrological connectivity clearly affected wintering waterbirds: the abundance and species richness of wintering waterbirds were significantly lower in isolated lakes than in controlled or freely connected ones (Xia et al. 2016). Furthermore, lakes with moderate control and connectivity to rivers had the highest waterbird abundance and species richness.

Other landscape-level factors such as surrounding land use and disturbance (e.g., hunting, or disrupted wetland hydrology) impact waterbird use of wetlands (Havera et al. 1992, Peterson and Niemi 2007). For instance, Beatty et al. (2014b) demonstrated that mallards shift wetland use in response to anthropogenic disturbances and site proximity to protected areas and sanctuaries, and Havera et al. (1992) found that boating activities were the primary source of disturbance in spring and autumn on diving ducks in the major staging area of Pool 19 in Illinois. Further, Peterson and Niemi (2007) demonstrated that wetland birds responded to vegetative structure and the degree of anthropogenic disturbance within and surrounding the wetlands, specifically with intensity of surrounding land use.

To facilitate effective wetland management and prioritize wetland restoration for waterbirds, it is prudent to understand local and landscape factors associated with wetland use during periods critical for annual cycle events. During spring migration of 2016–2017, I examined waterbird density in NWI polygons across Illinois relative to wetland hydrology and vegetation cover, wetland complexity and connectivity, wetland management practices, anthropogenic disturbance, and surrounding land use. Factors associated with waterbird use may be used to verify habitat suitability indicators (Chapter 2) useful in evaluating wetlands quality and tracking restoration success (Hagy et al. 2017). Moreover, my results will inform strategies of the Illinois Wetlands Campaign of the Illinois Comprehensive Wildlife Conservation Plan and Strategy (IDNR 2005) and UMRGLR Joint Venture for strategic management and conservation of wetlands for waterbirds.

3.3 MATERIALS AND METHODS

Study area

I studied wetlands identified by the National Wetlands Inventory (NWI) across Illinois on both public and private land. The study spanned Illinois' 14 Natural Divisions which are distinct ecoregions differentiated by topography, soils, glacial history, flora, and fauna (Figure 2.1; Schwegman 1973), with the Lake Michigan Natural Division excepted. I excluded the Lake Michigan Natural Division based on assumptions that changes in inundation and habitat suitability over time were minimal and habitat availability was likely captured by NWI. Despite human alteration and loss of ~90% of its wetlands (Dahl 1990, 2006), Illinois continues to support large populations of migratory waterbirds as well as some breeding populations. Illinois is classified as a stopover state by the UMRGLR Joint Venture that lies within the heart of the Mississippi Flyway, with breeding grounds primarily to the north and wintering grounds to the south for most species of wetland-dependent waterbirds (Havera 1999).

In Illinois, average monthly precipitation between February and April 2016–2017, the spring survey period, was 6.8 cm and 9.4 cm respectively, and the Palmer Hydrological Drought Index (PHDI) values ranged from 1.37–3.13 indicating wetter conditions than average (NCDC 2010). Average temperature, measured in Celsius (C°), for the month of February was 1.0° in

2016 and 5.1° in 2017, whereas average temperature during March was 8.5° in 2016 and 6.2° in 2017. During April, average temperature was 11.7° in 2016 and 13.6° in 2017.

Determining Ground Sampling Plots

I surveyed wetland habitat conditions for waterbirds during spring migration in 2016–2017. I conducted surveys for northerly migrating waterfowl and other waterbirds (mid-February–mid-April). Each year, I visited 100–120 25-ha plots overlapping NWI-mapped wetlands (Figure 1.2). I consolidated NWI wetland polygons into 6 classes according to major wetland cover types (emergent, forested, lake, pond, riverine and other; Cowardin et al. 1979; Table 2.1), and I omitted any polygons < 0.5 ha. I generated plots randomly using a spatially-balanced stratification (Theobald et al. 2007) based on consolidated NWI wetland classes and Natural Divisions. To do this, I first created a wetland density map for the state using 10-m cell sizes and a search window of 0.25-km × 0.25-km. Sampleable NWI wetlands were greater than 0.35 km from a Natural Division border to ensure plots were placed within a distinct Natural Division. I then converted the density map to an inclusion probability map, where the highest density was one and the lowest density was zero. Using the Reversed Randomized Quadrant-Recursive Raster tool (RRQRR) in ArcGIS 10.4 (ESRI 2011), I created an overrun of two times the target number plots within consolidated NWI polygons in a spatially-balanced design using the Natural Divisions as strata (Theobald et al. 2007). I selected plots with equal inclusion probabilities for the six NWI wetland classes. Then, I used Neyman allocation (Neyman 1934) to allocate survey effort proportionally based on NWI wetland area within Natural Division with a minimum of three plots per Natural Division. This sampling design with unequal effort among strata allowed for unbiased estimation (Johnson et al. 2009) and was flexible to uncertainty in removing and adding sites (e.g., denied access by landowner; Theobald et al. 2007).

Within each plot, I mapped inundation and vegetation boundaries using a field PC (Juniper Systems, Archer) with a GPS on foot or by boat. Inundated areas, as well as areas classified as mud or patchy inundation were separated, and I mapped vegetation communities (e.g., emergent/herbaceous, woody; Table 1.3). Vegetation and inundation patches were mapped if they were at least 6 m wide or $\geq 36 \text{ m}^2$. I waded or boated inundated areas to assess depth profile in at least 10 spots to evaluate the proportion of the wetland polygon that was shallowly

inundated (<45cm). Inundated areas within the plot but outside of NWI-mapped polygons were also mapped and data collection occurred as described above if the inundated area $\geq 400 \text{ m}^2$. Since these newly mapped areas were mapped strictly relative to inundation boundaries, they likely did not encompass the entire area that could be classified as a wetland. Personnel lacked the expertise to delineate wetlands, so these data represented inundated areas omitted by the NWI or wetlands <0.5 ha, which were omitted in plot selection.

Using the recorded tracks and field notes, I created thematic cover maps by digitizing inundation and vegetation cover in ArcGIS. I differentiated inundation and cover polygons at a site (e.g., Figure 4.4). Next, I overlaid the cover maps with the NWI polygons at a given site, dissolved by cover (e.g., open, short herbaceous, etc.) and inundation type (e.g., inundated, dry, etc.) by NWI class.

I noted evidence of wetland management including indication of drawdowns, presence of impoundments, mowing or disking, moist-soil plant management for waterfowl, and water-control structures (e.g., culverts, stop-log structures). Based on evidence of management while searching the 25-ha plots, observers determined the intensity of management on an ordinal scale (e.g., none, low, medium, and medium/high; Table 2.1). This scale ranged from, for example, unmanaged bottomland forests to highly managed duck club areas with water control, impoundments, and at times, planting of food plots specifically for waterfowl and wildlife. Any information pertaining to management regime or practices was gleaned from visual observations or from landowners or site managers when possible.

To determine whether wetland connectivity (i.e., proportion of surrounding 100-m buffer composed of wetlands and connectivity to rivers/streams) could predict the use or abundance of waterbirds, I included the proportion of a 100-m and 5-km buffer surrounding plots that was composed of wetlands (i.e., both emergent and woody; Tozer et al. 2010, Beatty et al. 2014a) as classified by the 2011 National Land Cover Dataset (NLCD, Fry et al. 2011; Table 2.3). Additionally, observers scored the NWI polygon's connectivity to rivers or streams on an ordinal scale ranging from 0 – 3 (no connectivity to permanent connection; Table 2.3). These scores were based upon observations of hydrological characteristics (i.e., secondary indicators).

To assess whether wetland complexity impacted waterbird use or abundance, I extracted edge density (ED), a quantitative measure of the interspersion of vegetation and water (i.e., open and aquatic bed; Rehm and Baldassarre 2007). I used the Spatial Statistics by Patch Analyst extension (i.e., the Fragstats interface) in ArcGIS (Rempel et al. 2012) to calculate ED of the cover maps by taking the total length of edge in meters in a given NWI class and dividing it by the total area of the NWI class to get a density of meters of edge per hectare. This metric was then comparable to other sites and other NWI classes at a given site due to its quantitative nature (i.e., edge increases in direct proportion to mixing; McGarigal et al 2002).

To measure disturbance to wetland habitat and hydrology and determine whether disturbance impacts use or abundance of waterbirds, I used a modified version of the Ohio Rapid Assessment Method (ORAM) Version 5.0 during 2016–2017 (Mack 2001). With an emphasis on anthropogenic disturbance, the ORAM scores wetland integrity using six metrics including surrounding land use, hydrology, habitat alteration and development, interspersion, and vegetation communities (Table 2.5). I scored wetland polygons using a subset of the ORAM metrics and submetrics: 2b, 3a-e, 4a-c, 6a-d (Mack 2001; Table 2.5) on a gradient similar to ORAM's method of scoring wetlands along a quality gradient for regulatory purposes. I collected data and calculated a modified ORAM score at the NWI wetland class level.

I assessed potential impacts of surrounding landscape in ArcGIS by calculating the Landscape Development Intensity Index (LDI; Brown and Vivas 2005). The LDI represents a measure of human disturbance on a gradient relative to intensity of land use (Brown and Vivas 2005, Mack 2006). Low LDI values (≤ 2) represent land uses that are more natural (e.g., wetlands, grasslands, forests), whereas high LDI values (>10) represent highly altered/disturbed surrounding land uses (e.g., row-crop agriculture, urbanization). I computed the LDI using the 2011 NLCD at two spatial scales – a local 100-m and landscape-level 5-km buffer (e.g., mallard foraging flight distance; Webb et al. 2010; Table 2.1). I calculated the proportion of different land uses surrounding sites and assigned coefficients based on intensity of the land uses (following recommendations by Mack 2006) for two spatial scales: 100-m and 5-km surrounding plots.

Aerial Surveys

I conducted three aerial surveys at 2–3-week intervals in each year spanning spring migrations of most waterfowl species (March 3, 2016 – April 14, 2016; February 17, 2017 – April 13, 2017). I selected sites where I received permission to conduct ground surveys, described above (Figure 2.1). I created three flight paths each year: Upper Mississippi River and Illinois River, Chicago Region, and Southern and Eastern Illinois. To determine the size of the site from the air, I marked the right wing of the fixed-wing single-engine aircraft (Piper Arrow) to mark the approximate outer boundary of each 25-ha plot at the given altitude (e.g., 50 – 100 m). Surveys were flown at a height of 50–100 m above ground level while travelling 160–240 km/hr. (Havera 1999). The pilot navigated to plot boundaries using a GPS receiver, and I sat in the right-front seat and observed the plots from the right window of the plane. We made two passes along both the north and south side of the plot unless I sighted more waterbirds within the plot, requiring an additional pass to identify species. I identified and enumerated all waterbirds in each plot to species or foraging guild with a focus on dabbling ducks (Anatini). To determine which NWI class polygon a bird or group of birds was observed in, I noted the location of the bird or group on a paper map and later assigned locations to NWI classes using polygons in ArcMap. I censored instances where there was considerable uncertainty as to which NWI class waterbirds were observed in. Aerial surveys were limited by weather and not conducted in high winds (≥ 24 km/h) or inclement weather (e.g., precipitation; Pearse et al. 2008). I recorded light and weather conditions, wind speed, and time of survey. All surveys were diurnal, occurring during 07:30 – 17:00.

Aerial surveys were completed during leaf-off periods when visibility through forest canopies was maximized. Experienced observers conducted all flights and practiced count estimation outside of these surveys. Visibility bias during census-style aerial surveys under similar conditions is low (Gilbert 2018) and it should have been minimal during this survey as generally low numbers of waterbirds (i.e., small group sizes) were most often observed. If the aerial observer was uncertain about species identification or counts, multiple passes over the site were completed. Thus, I assumed that visibility bias was minimal and consistent across NWI classes.

Statistical Analyses

I modeled dabbling duck and other waterbird (e.g., geese, swans, diving ducks, herons) density (e.g., birds by NWI polygon area in hectares) by local and landscape variables using zero-inflated regression due to the considerable number of zeros in the response variables. Zero-inflated models (Lambert 1992) combine a standard discrete distribution such as the Poisson or negative binomial with a binomial distribution (Zeileis et al. 2008). The discrete distribution describes count data, and the binomial distribution accounts for zeros present in greater numbers than predicted by the discrete distribution (Ridout et al. 1998). Zero-inflated regression describes the occurrence of zeros using a mix of discrete and binomial distributions that allows zeros to originate from both distributions. Zeros in the response variable may originate from multiple processes including; non-detection of animal being surveyed (e.g., observer error, difficult to observe species), false zeros (e.g., habitat suitable but animal is not present), or structural zeros (e.g., site not used because habitat unsuitable; Potts and Elith 2006, Zuur et al. 2009). The binomial distribution is treated as modeling false zeros while the count distribution models true zeros that arise under certain conditions (e.g., observer error, difficult to observe species; Zuur et al. 2009). Moreover, the zero-inflation model estimates the probability of an extra zero so that a positive contrast indicates a higher chance of absence; this is opposite of the count portion of the model where a positive contrast indicates greater abundance (Brooks et al. 2017).

I modeled dabbling duck and non-dabbling duck waterbird density using the ‘glmmTMB’ package (Brooks et al. 2017) in R (Version 3.4.2; R Core Team 2017). Specifically, I chose to use glmmTMB to implement a mixed effects approach and account for the non-independence between survey rounds at a site and between sites within years. Therefore, I nested plot ID within year and treated this as a random effect. Due to the varying sizes of NWI polygons, I treated the log-transformed area of the NWI class as an offset, thereby modeling waterbird density (birds/ha). Further, I confirmed that there was overdispersion, or more variation than expected given the data, by comparing log-likelihoods of a zero-inflated Poisson and zero-inflated negative binomial regression model (Zeileis et al. 2008). I chose the negative binomial distribution to account for both zero-inflation and overdispersion. I checked for outliers in the response variable, and I checked my predictor variables for collinearity, thereby excluding highly

correlated variables $r > |0.5|$ to avoid unexpected bias in the standard errors of parameter estimates (Zuur et al. 2010).

I combined variables to represent biological hypotheses thought to influence waterbird use or density in Illinois wetlands during spring migration. Candidate models included wetland hydrology/cover, complexity, connectivity, disturbance, time of survey, and management (Table 2.1). I included subsequent additive combinations of the variables if models were composed of multiple variables. Lastly, I included variables that likely impacted zero inflation (e.g., open vs. forested cover impacted observer's ability to detect waterbirds; Zuur et al. 2009). I used Akaike's Information Criterion (AIC) adjusted for small sample sizes (AIC_c) to rank and evaluate models, and I considered models $\leq 2\Delta AIC_c$ competing (Burnham and Anderson 2002); however, I assessed competing models for pretending variables by examining effect sizes and coefficients (Arnold 2010). I assessed model fit by comparing Pearson residuals against fitted values (Zuur et al. 2009). Further, using the 'sjstats' package (Lüdtke 2018), I calculated the marginal R^2 values representing the variance explained by the fixed factors (Nakagawa et al. 2017). Then, I calculated predicted estimates and 85% confidence intervals (Arnold 2010), and I obtained estimates for variables in the best-supported model ($\leq 2\Delta AIC_c$). I specified a range of values for the covariate of interest while keeping all other covariates in the model at the mean value.

3.4 RESULTS

In 2016 – 2017, I conducted aerial surveys for waterfowl and other waterbirds at 188 plots (i.e., 86 in 2016, and 102 plots in 2017). I completed 9 flights in 2016 and 8 flights during 2017; one flight was missed in 2017 due to weather. In total, I observed 16,081 waterbirds, of which dabbling ducks were the most numerous (Table 3.2). Mallards (*Anas platyrhynchos*) were the most abundant species observed with 5,254 individuals and were present in 42% of plots during at least one survey round. The next most numerous species was American green-winged teal (*Anas crecca*) with 1,987 individuals and were present in 18% of plots during at least one survey round (Table 3.2). Besides dabbling ducks, I observed 2,908 diving ducks, 2,031 geese (Anserini), and 2,019 non-waterfowl waterbirds (e.g., herons and egrets [*Ardea* spp.], American coots, etc.; Table 3.2). In total, I detected 27 species (Table 3.2), and I observed waterbirds in 75% of plots in at least one survey.

I modeled dabbling duck and other waterbird (e.g., geese, swans, diving ducks, herons) density by local and landscape variables (Table 3.1). For dabbling ducks, the best-supported model included total inundation cover (+), cover of shallow inundation (i.e., <45 cm) (+), and cover of inundated woody (both forested and scrub-shrub) vegetation (-) (Table 3.3; $R_m^2=11.1\%$, $R_c^2=31.3\%$). There were two models $\leq 2\Delta AIC_c$: one including total inundated cover and shallow inundation only, and the second included total inundated cover, shallow inundation, inundated herbaceous cover, and inundated woody vegetation cover (Table 3.3). Coefficients of competing models had very similar values, and the variable herbaceous cover had a small effect size, so I used estimates from the top model (Arnold 2010). For every 10% increase in total inundation, there was a 34.8% (22.7 – 48.2%) increase in the dabbling duck density (Figure 3.2). Similarly, with every 10% increase in shallow inundation cover, there was a predicted 26.5% (13.7 – 40.8%) increase in dabbling duck density (Figure 3.3). For every 10% increase in inundated woody vegetation, there was a 14.6% (2.8 – 24.9%) decrease in the dabbling duck density (Figure 3.4).

Regarding the zero-inflation portion where a positive contrast indicates a higher chance of absence, the proportion of forested cover was negatively related to waterbird detections and the proportion of open water was positively related to waterbird detections. There was a 7.8% (1.3 – 13.8%) increase in the odds of detecting at least one waterbird for every 10% increase in open water, whereas there was a 1.8% (-4.5 – 8.5%) decrease in the odds of detecting at least one waterbird with every 10% increase in the total forested cover.

The best supported model for non-dabbling duck waterbird (e.g., diving ducks, geese, herons, American coots) included density inundated cover (+), shallowly inundated cover (+), inundated woody vegetation (-), and inundated herbaceous vegetation (-) (Table 3.4; $R_m^2=13.2\%$, $R_c^2=28.3\%$). There were no competing models (i.e., $2\leq\Delta AIC_c$; Table 3.4). For every 10% increase in total inundation cover, there was a 44.6% (32.5 – 57.8%) increase in other waterbird density (Figure 3.5), whereas with every 10% increase in shallow inundation cover, there was a 19.1% (8.7 – 30.4%) increase in other waterbird density (Figure 3.6). On the other hand, for every 10% increase in inundated woody vegetation, there was a 24.3% (13.2 – 33.9%) decrease in other waterbird density (Figure 3.7). Similarly, for every 10% increase in inundated

herbaceous cover, there was a 37.9% (18.9 – 52.5%) decrease in other waterbird density (Figure 3.8).

The zero-inflation portion of the model indicated that for every 10% increase in inundated open water cover, there was an 8.3% (-1.1 – 16.9%) increase in the odds of detecting at least one waterbird; however, with a 10% increase in the total forested cover, there was 17.1% (7.2 – 28.0%) decrease in the odds of detecting at least one waterbird.

3.5 DISCUSSION

Wetland hydrology and vegetation cover were best predictors of dabbling duck and other waterbird density during spring migration in Illinois. Both dabbling duck density other waterbird density was positively related to total inundation and shallow inundation (<45cm) cover within NWI polygons but was negatively related to inundated woody (both scrub-shrub and forested) vegetation. Other waterbird density was also negatively associated with the cover of inundated herbaceous vegetation. Open water increased the odds that I would detect at least one waterbird, whereas total forested cover decreased the odds of detecting at least one waterbird. It is critical to recognize both direct and indirect factors that influence waterbird use during spring migration, as recent studies have demonstrated associations between reproductive performance and habitat conditions during spring migration (Anteau and Afton 2004, Morrison 2006, Devries et al. 2008). Local hydrology and vegetation cover, for instance, may indirectly influence waterbird habitat selection due to perceived risks and rewards of different habitat resources (Behney et al. 2018). Furthermore, ducks encounter the lowest abundance of food resources during spring compared to other seasons (Brasher et al. 2007) and food may be limiting in some locations and years in the Midwest (Straub et al. 2012, Hagy et al. 2017a).

Water availability and depth influence waterbird distribution, habitat use, and key behaviors such as foraging (Conway et al. 2005, Hagy et al. 2014). Water depth influences the accessibility of available foraging habitat directly because waterbird morphology and size limit where waterbirds can forage (Isola et al. 2000, Davis and Smith 2001). Thus, many waterbird species forage in shallowly flooded areas in wetlands (Isola et al. 2000, Taft et al. 2002, Boertmann and Riget 2006, Webb et al. 2010). The proportion of an NWI polygon that was inundated and shallowly inundated (<45 cm) were both predictors of increased dabbling duck

and non-dabbling duck waterbird density. Many have found that hydrological characteristics and water depth were important predictors of waterbird use of wetlands (e.g., Isola et al. 2000, Taft et al. 2002, O'Neal et al. 2008, Webb et al. 2010). For instance, Kaminski and Prince (1984) studied dabbling duck habitat use in spring and found that species richness and pair densities were positively correlated with the cover of shallow marsh, and American coot habitat use was correlated with total flooded area within marshes (Murkin et al. 1997). Furthermore, Murkin et al. (1997) compared seasonal differences in habitat use and found that mallards used areas in spring that had more water and less vegetation than areas used in summer. Increased inundation may also influence waterbird habitat selection indirectly as smaller regions of inundation may be considered riskier by certain waterbirds, especially those that require inundated areas of a sufficient size to take flight. Generally, waterbirds that forage in open and deep-water habitats are considered area-dependent and are restricted to larger regions of inundation (Paracuellos 2006). Furthermore, permanent wetlands in this study had high levels of inundation during spring migration (i.e., >80%), whereas average forested and emergent wetland inundation was much lower, ranging from 30% to 50% respectively (Chapter 2). Lower inundation potentially influenced waterbird use due to perceived risk of vegetated wetlands (Behney et al. 2018) or due to limited available habitat.

In addition to inundation and water depth, density of dabbling ducks and other waterbirds were negatively related to inundated herbaceous vegetation cover; however, herbaceous cover was only included in a subsequent model $\leq 2\Delta AIC_c$ predicting dabbling duck density and may be considered a pretending variable (Arnold 2010). While an initially surprising result, others have reported similar findings. For instance, Laubhan and Gammonly (2000), demonstrated that mallards, cinnamon teal (*Anas cyanoptera*), and gadwall (*Anas strepera*) preferred seasonally flooded basins with no emergent vegetation and wetlands with short emergent vegetation compared to other available habitat resources prior to the breeding season. Further, Webb et al. (2010) reported that goose abundance was negatively related to emergent vegetation in the Rainwater Basin Region of Nebraska during spring migration. Here, average cover of inundated herbaceous cover was only 2.6% (1.7 – 3.6%), and herbaceous cover was low, even for emergent NWI polygons (NPE <11% [8 – 14%] and PEM <8 [7 – 9%]; Chapter 2). Perhaps this negative

relationship with emergent vegetation cover was partially due to the infrequent cover or the importance of hydrology despite cover.

Dabbling duck and other waterbird density were both negatively associated with inundated woody vegetation cover. Some have found similar relationships with spring migrating waterbird use and woody vegetation cover. For instance, Kaminski and Prince (1984) reported that dabbling duck pair densities and species richness were negatively associated with the cover of forest. However, in the same study, Northern shoveler (*Anas clypeata*) pair densities were positively related to forested cover (Kaminski and Prince 1984). In southeastern Illinois, Williams (2018) found that almost 90% of radio-marked mallard and green-wing teal use locations were either in emergent or woody wetlands, and ducks used emergent and woody wetlands greater than their availability on the landscape during spring migration. Habitat associations often vary during different stages in the annual cycle, especially in migratory species. For instance, during winter in South Carolina, mallard abundance and dabbling duck species richness were positively correlated to forested wetland area (Gordon et al. 1998), and Davis and Afton (2010) reported that radio-marked hen mallards selected forested wetlands while wintering in Louisiana. Beatty et al. (2014a) also studied radio-marked hen mallards over multiple seasons and observed an increase in woody wetland selection during winter and spring migration compared to autumn migration, and total mallard use days were positively associated with proportion of forested wetland area in the Illinois River Valley during spring migration (Stafford et al. 2007). However, I found that average inundation in forested wetlands during spring in Illinois was only 32.6% (29.6 – 35.6%; Chapter 2) during the time of this study, perhaps necessitating the use of other, more open water wetlands.

My results suggest that waterbird density was negatively related to woody and herbaceous vegetation cover during spring in Illinois. Similarly, Beatty et al. (2014a) also found that proximity to open water was an important predictor of mallard habitat selection during spring and autumn, and Yetter et al. (2018) found that mallards in the Illinois River Valley used open water most frequently during autumn. This relationship likely varies interspecifically. For example, Williams (2018) observed that the probability of wetland use by mallards was negatively related to open water cover, whereas green-winged teal had higher probabilities of use corresponding to open water cover in the surrounding landscape. Open water may be used for

foraging when shallow wetlands dry or freeze and may provide areas for birds to roost, preen, and court, potentially minimizing predation from terrestrial predators (Heitmeyer 2006). While vegetated wetlands provide important foraging and resting areas for waterbirds during migration due to vegetation cover and food abundance (Stafford et al. 2007), vegetation cover may also limit visibility and obstruct waterbirds' field of view. For instance, Behney et al. (2018) reported that vegetation above the water's surface was perceived as risky and resulted in ducks altering foraging behavior. Furthermore, foraging intensity was lower in both emergent and forested wetlands compared to open wetlands (Behney et al. 2018).

Most studies rely on existing spatial wetland data such as the NWI (Cowardin et al. 1979) or National Land Cover Dataset (Fry et al. 2011) or other aerial imagery to provide information on waterbird habitat use or selection (e.g., Stafford et al. 2007, Webb et al. 2010, Beatty et al. 2014a). However, existing spatial data often do not accurately reflect actual on the ground conditions. For instance, I found that only 30% of forested NWI polygon area contained inundated forest during spring migration in Illinois, and emergent polygons contained approximately 20% inundated persistent and non-persistent emergent vegetation during summer (Chapter 2). Additionally, existing spatial data may not reflect wetland conditions during the time period that inference is taken and lack insight on temporal changes in wetland inundation and vegetation cover since many wetland systems are seasonal and dynamic. Thus, my study highlights the potential disconnect between local-scale hydrological and vegetation cover and wetland or land use databases and may partially explain why my results differ. I recommend continued caution or the use of ground-truthing when relying on these sources to describe waterbird habitat use or selection.

Spring migration in an energy-intensive period for waterbirds where resting and refueling at stopover wetlands are critical (Arzel et al. 2006, Morrison 2006). However, since I conducted diurnal surveys between 7:30 and 17:00, I may not have identified foraging habitats since many waterbirds forage primarily during crepuscular periods or even nocturnally (McNeil et al. 1992). Anderson and Smith (1999), for example, reported a ten-fold increase in nocturnal waterfowl counts in moist-soil managed playa wetlands compared to diurnal counts. Perhaps, instead, I observed waterbirds roosting or resting. Many waterbird species roost and loaf in large regions of open water and may subsequently forage in more vegetated wetlands. However, Behney et al.

(2018) observed higher feeding intensity in open water wetlands, and Hagy et al. (2017b) reported that dabbling ducks and diving ducks spent a significant portion of their time, 61.8% and 27.1% respectively, foraging diurnally, suggesting that some waterbirds I observed were likely foraging. Additionally, Hagy et al. (2017a) and Straub et al. (2012) found that food abundance is low in wetlands within Illinois and within the Upper Mississippi River and Great Lakes Region. Specifically, Straub et al. (2012) reported that half of forested and lacustrine wetlands had little or no forage value during spring migration in this region. This might suggest that waterbirds were not incentivized to use forested wetlands during spring, especially if they perceive them as riskier in comparison to open wetlands.

The best supported model predicting dabbling duck and other waterbird density included local hydrological and vegetation cover; however, the marginal (11.2 – 13.2%) R^2 values indicated that a large portion of variability was not explained by these variables alone. Other models performed better than the intercept only model, indicating some level of predictability. For instance, models containing management intensity or hydrological management (e.g., O’Neal et al. 2008, Kaminski et al. 2009), wetland connectivity to other wetlands or to rivers and streams (e.g., Webb et al. 2010, Beatty et al. 2014a, Xia et al. 2016), local and landscape disturbance (e.g., Havera et al. 1992, Stapanian et al. 2004, Beatty et al. 2014b), wetland complexity (e.g., Kaminski and Prince 1984, Riffell et al. 2001, Stafford et al. 2007), or time of day (e.g., Pearse et al. 2012) all performed better than the null and could impact waterbird use of wetlands during spring migration. In addition, other factors that were not measured in this study such as invertebrate or seed abundance (Osborn et al. 2017), predation risk, current and cumulative effects of temperature, snow or ice cover (Schummer et al. 2010), the influence of conspecific behavior, disturbance from barges, boats, or vehicles (Havera et al. 1992, Osborn et al. 2017), or a combination of other factors could also affect waterbird density and use of wetlands.

It is important to note that increased waterbird use or density, may not necessarily relate to highly suitable or high-quality habitats. Van Horne (1983) cautioned that density of study organisms should not imply suitability of a habitat without supporting information on biological outcomes. However, in an answer to Van Horne (1983), Bock and Jones (2004) reviewed ornithological studies and subsequently recommended that in breeding areas, adult density is

often positively related to reproduction and thus suitability. Furthermore, approximately 70% of migratory species included in their study showed higher reproductive output with increased density. Consequently, they found that density and suitability may be disjointed in regions of high disturbance (Bock and Jones 2004) in addition to regions of temporal habitat unpredictability, patchiness, and in seasonal habitat which may relate to waterbirds in this study (Van Horne 1983). I assumed that wetlands used by waterbirds are supplying some resources needed during spring migration; however, without the support of demographic data this is partially unsubstantiated.

Lastly, although I did not correct for detection probability or visibility from the aerial surveys (Pearse et al. 2008), I assumed that I successfully detected waterbirds, when present, at a constant rate across NWI types. I believe this assumption is reasonable since an experienced aerial observer accompanied me during initial flights and provided training in the off season, surveys were conducted during leaf-off, and survey plots were small (25-ha). Additionally, this assumption is supported by Gilbert (2018) that reported high detection rates (e.g., 99%) and low count bias while conducting cruise-style aerial surveys for waterbirds in the Illinois and Upper Mississippi River Valley. Here, I implemented zero-inflated regression to include covariates that could impact the number and type of zeros recorded (e.g., false zeros, structural zeros, difficult to survey species, or observer error; Zuur et al. 2009). The zero portion of the regression indicated that both total forested and inundated open water cover were predictors of zero waterbirds; increased cover of open water decreased the chance of detecting zero waterbirds. Even though I conducted surveys during leaf-off periods, total forested cover was negatively related to the occurrence of at least one waterbird. Similarly, Pearse et al (2008) found wetland cover type (open vs. forested) influenced whether the aerial observer correctly detected duck decoy groups. They found that detection in forested wetlands was relatively constant since the forested wetlands in their study were smaller, allowing the observer to scan each wetland completely (Pearse et al. 2008). Furthermore, since I was surveying relatively small, 25-ha wetland plots and not entire wetlands, I also had ample time to completely scan plots, and I assumed that I successfully detected waterbirds, when present, even in forested NWI polygons.

Waterbirds likely experience cross-seasonal effects (Devries et al. 2008, Sedinger and Alisauskas 2014), and since migratory species cross a plethora of jurisdictional boundaries,

management is often difficult to coordinate (Myers et al. 1987). Illinois has undergone extensive wetland loss, and due to the importance of habitat throughout the annual cycle, it is imperative to conserve and manage remaining wetland habitat to meet the needs of wetland-dependent wildlife during spring migration, a critical pre-breeding period. Here, I provided information on factors related to waterbird use of wetlands during spring migration in Illinois, furthering our understanding of habitat use throughout the annual cycle. Specifically, local hydrological and vegetation cover were the most important predictors of waterbird use; however, my research also suggests that consideration of other local and landscape-level characteristics may improve our understanding of wetland use by waterbirds and highlights the disconnect between local-scale hydrological and vegetation cover and wetland or land use databases that may not accurately describe spatiotemporal wetland conditions. While many factors influence waterbird habitat use and selection, at a bare minimum, I recommend supplying surface inundation and regions of shallow water during spring migration in Illinois and suggest that restoration activities may be more impactful in instances where there is hydrological control.

3.6 TABLES AND FIGURES

Table 3.1. Model group, variables, variable code/abbreviation, and description dabbling duck abundance and non-dabbling duck waterbird density models. The variables for zero-inflation were included across all model sets.

Model	Variables	Variable Code	Variable Description
Connectivity	Proportion wetland 100m	WET_100m	Proportion of wetlands (woody and emergent) in surrounding 100-m buffer
	Proportion wetland 5km	WET_5km	Proportion of wetlands (woody and emergent) in surrounding 5-k buffer
	River connectivity	RC	An ordinal factor variable scoring a wetland's connectivity to rivers ranging from 0 (no connection apparent) to 3 (permanent connection)
Complexity	Edge density	ED	Open water vegetation edge divided area to represent interspersed extracted from thematic cover maps
Cover	Inundated herbaceous	HERB	Proportion of NWI class mapped as inundated herbaceous vegetation
	Inundated woody vegetation	WOODY	Proportion of NWI class mapped as inundated woody (both scrub-shrub and forested) vegetation
	Total inundation cover	INUND	Proportion of NWI class inundated (all cover types)
	Shallow inundation cover	SHALLOW	Proportion of NWI class shallowly inundated (<45 cm)
Disturbance	LDI score 5km	LDI_5km	Landscape development intensity score for surrounding 5-k buffer. Higher values indicate more intense land uses
	ORAM score	ORAM	Score representing anthropogenic disturbance and alterations to natural wetland hydrologic regime or habitat, ranging from 0 (highly disturbed/impacted) to 100 (pristine unaltered wetland conditions)
Latitude Management	Latitude	LAT	Latitude of site location measured in degrees.
	Management intensity	MANAGEMENT	An ordinal factor variable associated with wetland management intensity for wildlife (none, low, medium, high)
Time	Time since sunrise	TIME_POST_SR	The difference between time and sunrise in the geographic center of Illinois where minutes are adjusted to a decimal (e.g., 10:15 AM=10.25).
Zero-inflation	Total forest cover	Total_FO	Proportion of NWI class mapped as dry or inundated forested (>6 m).
	Open water	Inund_open	Proportion of NWI class mapped as inundated open water.

Table 3.2. Common names, scientific names, and abundance of waterbird taxa and guilds identified during spring aerial surveys in Illinois, U.S.A. during 2016 – 2017.

Common Name	Scientific Name	Abundance
American green-winged teal	<i>Anas crecca</i>	1,987
American wigeon	<i>Mareca americana</i>	7
Blue-winged teal	<i>Spatula discors</i>	739
Gadwall	<i>Mareca strepera</i>	274
Mallard	<i>Anas platyrhynchos</i>	5,254
Northern pintail	<i>Anas acuta</i>	125
Northern shoveler	<i>Spatula clypeata</i>	563
Wood duck	<i>Aix sponsa</i>	174
	Total dabbling ducks	9,123
Bufflehead	<i>Bucephala albeola</i>	162
Canvasback	<i>Aythya valisineria</i>	61
Common goldeneye	<i>Bucephala clangula</i>	151
Common merganser	<i>Mergus merganser</i>	23
Hooded merganser	<i>Lophodytes cucullatus</i>	5
Lesser scaup	<i>Aythya affinis</i>	1,603
Ring-necked duck	<i>Aythya collaris</i>	652
Ruddy duck	<i>Oxyura jamaicensis</i>	251
	Total diving ducks	2,908
Canada goose	<i>Branta canadensis</i>	1,024
Snow goose	<i>Chen caerulescens</i>	1,007
	Total geese	2,031
Swan	<i>Cygnus</i> spp.	7
American coot	<i>Fulica americana</i>	745
American white pelican	<i>Pelecanus erythrorhynchos</i>	255
Double-crested cormorant	<i>Phalacrocorax auritus</i>	214
Great blue heron	<i>Ardea herodias</i>	76
Great egret	<i>Ardea alba</i>	338
Gull	Family: Laridae	382
Sandhill crane	<i>Grus canadensis</i>	2
	Total non-waterfowl	2,012
	Total other waterbird	6,958
	Total waterbirds	16,081

Table 3.3. Rankings of models according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K), and model weights (w_i) for predicting dabbling duck density by local, landscape, and survey covariates.

Model	K	AIC_c	ΔAIC_c	w_i
INUND + SHALLOW + WOODY	10	2609.00	0.00	0.40
INUND + SHALLOW	9	2609.79	0.79	0.27
HERB + INUND + SHALLOW + WOODY	11	2610.56	1.57	0.18
HERB + INUND + SHALLOW	10	2611.65	2.66	0.11
INUND	8	2616.97	7.97	0.01
INUND + WOODY	9	2617.29	8.29	0.01
RC + WET_5km	11	2618.44	9.44	0.00
HERB + INUND	9	2618.83	9.84	0.00
HERB + INUND + WOODY	10	2619.27	10.28	0.00
RC + WET_100m + WET_5km	12	2619.63	10.64	0.00
ED (complexity)	8	2619.66	10.67	0.00
WET_100m + WET_5km	9	2620.54	11.54	0.00
WET_5km	8	2621.01	12.01	0.00
MANAGEMENT	11	2621.03	12.03	0.00
SHALLOW	8	2622.63	13.63	0.00
SHALLOW + WOODY	9	2624.66	15.66	0.00
HERB + SHALLOW	9	2624.66	15.66	0.00
HERB + SHALLOW + WOODY	10	2626.69	17.69	0.00
LDI_5km	8	2630.13	21.13	0.00
LDI_5km + ORAM	9	2630.65	21.65	0.00
ORAM	8	2630.65	21.65	0.00
NULL	7	2631.66	22.67	0.00
RC	10	2632.55	23.55	0.00
TIME_POST_SR	8	2632.71	23.72	0.00
HERB	8	2632.84	23.85	0.00
WET_100m	8	2632.92	23.93	0.00
LATITUDE	8	2632.95	23.95	0.00
WOODY	8	2633.25	24.25	0.00
RC + WET_100m	11	2634.27	25.27	0.00
HERB + WOODY	9	2634.42	25.43	0.00

Covariates include: ED – edge density; RC – river connectivity; WET_100m – Proportion of wetlands (woody and herbaceous) in 100-m buffer; WET_5km – Proportion of wetlands (woody and herbaceous) in 5-km buffer; HERB – proportion of NWI polygon within plot that was inundated herbaceous; WOODY – proportion of NWI polygon within plot that was inundated scrub-shrub <6m and inundated forest >6m; LDI_5km – Landscape Development Index score for surrounding 5-km buffer; ORAM – modified Ohio Rapid Assessment Method score; INUND – proportion of NWI polygon within the plot that was inundated.; SHALLOW – proportion of NWI polygon within plot that is inundated <45 cm; MANAGEMENT – management intensity (non, low, medium, high); LATITUDE – Latitude of site location measured in degrees; TIME_POST_SR – difference between time of survey and relative sunrise.

Table 3.4. Rankings of models according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K), and model weights (w_i) for predicting other waterbird density (e.g., geese, swans, diving ducks) by local, landscape, and survey covariates.

Model	df	AICc	$\Delta AICc$	w_i
HERB + INUND + SHALLOW + WOODY	11	2636.42	0.00	0.76
INUND + SHALLOW + WOODY	10	2640.18	3.76	0.12
HERB + INUND + SHALLOW	10	2641.78	5.36	0.05
HERB + INUND + WOODY	10	2642.36	5.94	0.04
INUND + WOODY	9	2643.94	7.52	0.02
HERB + INUND	9	2645.41	8.99	0.01
INUND + SHALLOW	9	2645.62	9.20	0.01
INUND	8	2647.34	10.92	0.00
RC + WET_5km	11	2668.19	31.77	0.00
HERB + SHALLOW + WOODY	10	2668.27	31.85	0.00
SHALLOW + WOODY	9	2668.86	32.44	0.00
HERB + SHALLOW	9	2669.12	32.70	0.00
RC + WET_100m + WET_5km	12	2669.94	33.52	0.00
SHALLOW	8	2670.02	33.60	0.00
TIME_POST_SR	8	2670.57	34.15	0.00
MANAGEMENT	11	2671.01	34.59	0.00
WET_5km	8	2671.66	35.24	0.00
NULL	7	2671.77	35.35	0.00
WOODY	8	2671.79	35.37	0.00
HERB	8	2672.19	35.76	0.00
RC	10	2672.43	36.01	0.00
HERB + WOODY	9	2672.51	36.09	0.00
LATITUDE	8	2672.92	36.50	0.00
LDI_5km	8	2673.35	36.92	0.00
ED (complexity)	8	2673.57	37.15	0.00
ORAM	8	2673.66	37.24	0.00
RC + WET_100m	11	2673.68	37.26	0.00
WET_100m + WET_5km	9	2673.68	37.26	0.00
WET_100m	8	2673.75	37.33	0.00
LDI_5km + ORAM	9	2675.35	38.93	0.00

Covariates include: ED – edge density; RC – river connectivity; WET_100m – Proportion of wetlands (woody and herbaceous) in 100-m buffer; WET_5km – Proportion of wetlands (woody and herbaceous) in 5-km buffer; HERB – proportion of NWI polygon within plot that was inundated herbaceous; WOODY – proportion of NWI polygon within plot that was inundated scrub-shrub <6m and inundated forest >6m; LDI_5km – Landscape Development Index score for surrounding 5-km buffer; ORAM – modified Ohio Rapid Assessment Method score; INUND – proportion of NWI polygon within the plot that was inundated.; SHALLOW – proportion of NWI polygon within plot that is inundated <45 cm; MANAGEMENT – management intensity (non, low, medium, high); LATITUDE – Latitude of site location measured in degrees; TIME_POST_SR – difference between time of survey and relative sunrise.

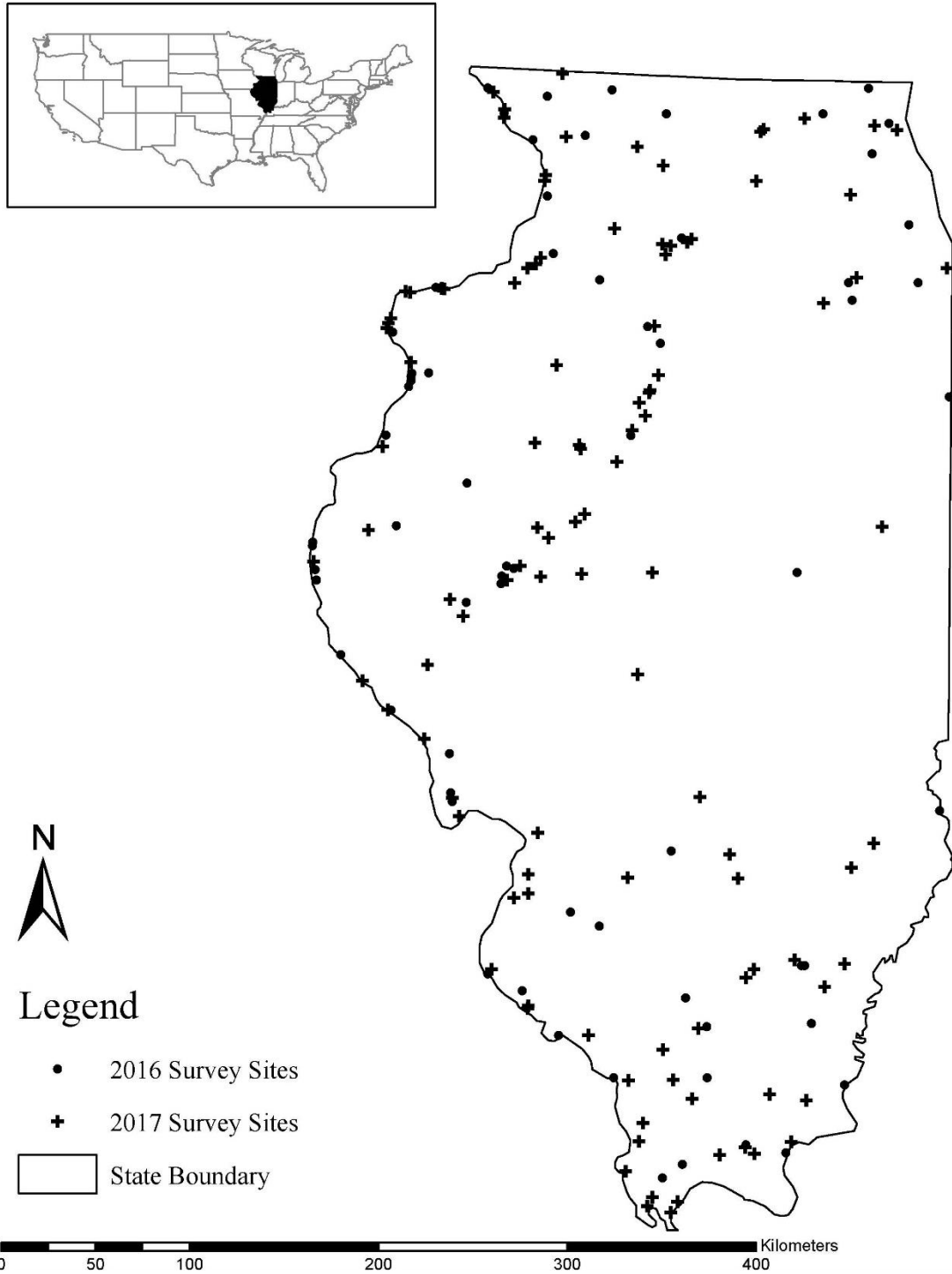


Figure 3.1. Locations of aerial surveys for waterbirds conducted by INHS during spring migration 2016 – 2017 in Illinois, USA.

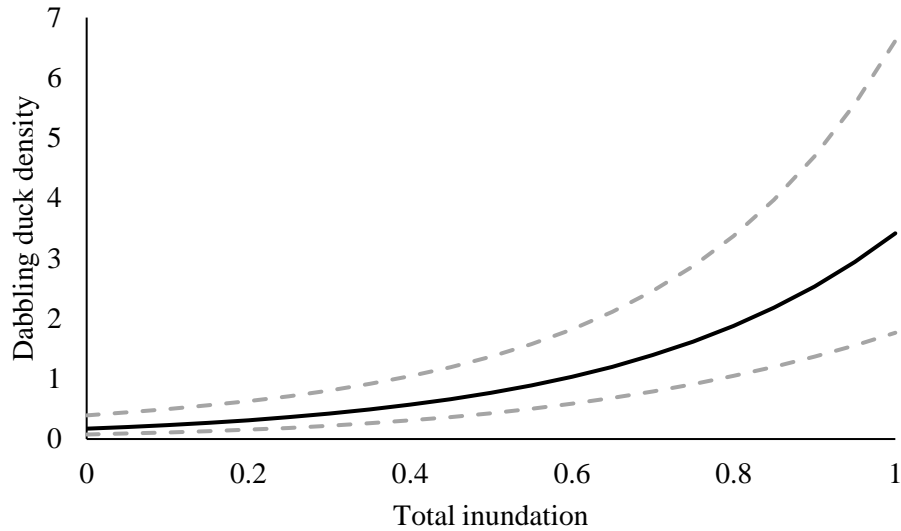


Figure 3.2. Model-predicted dabbling duck density (birds/ha) estimates (\pm 85% CI) by the proportion of total inundation cover (i.e., all vegetation types and open water) in NWI polygons at aerial survey locations in Illinois, U.S.A. during 2016 – 2017. Other variables were held constant at their mean (HERB= 0.026, WOODY=0.14, and SHALLOW=0.28). Estimates were corrected for zero inflation by multiplying by the probability of a non-zero (i.e., $1 - p$), and zero inflation variables were held at their mean (Total_FO=0.44, Inund_open=0.31).

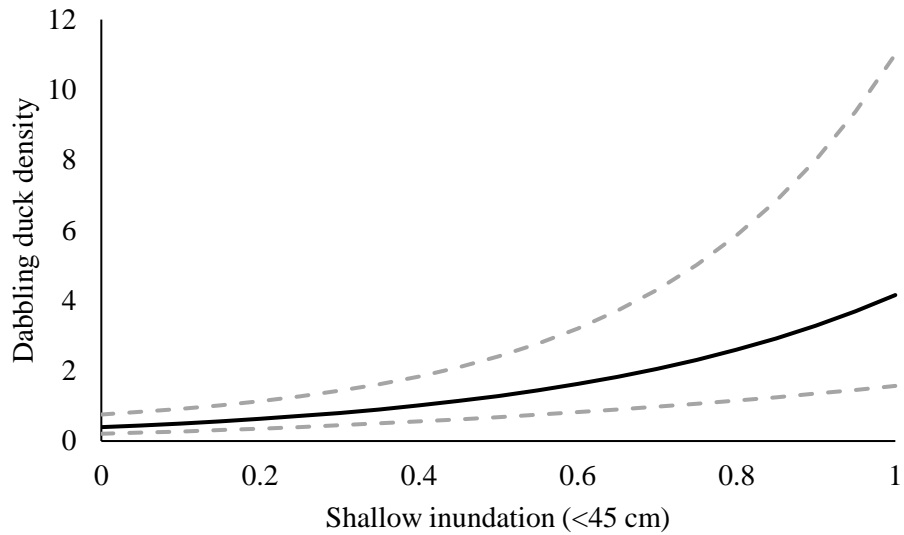


Figure 3.3. Model-predicted dabbling duck density (birds/ha) estimates (\pm 85% CI) by the proportion of shallow inundation (<45 cm) cover in NWI polygons at aerial survey locations in Illinois, U.S.A. during 2016 – 2017. Other variables were held constant at their mean (HERB= 0.026, WOODY=0.14, and INUND= 0.50). Estimates were corrected for zero inflation by multiplying by the probability of a non-zero (i.e., $1 - p$), and zero inflation variables were held at their mean (Total_FO=0.44, Inund_open=0.31).

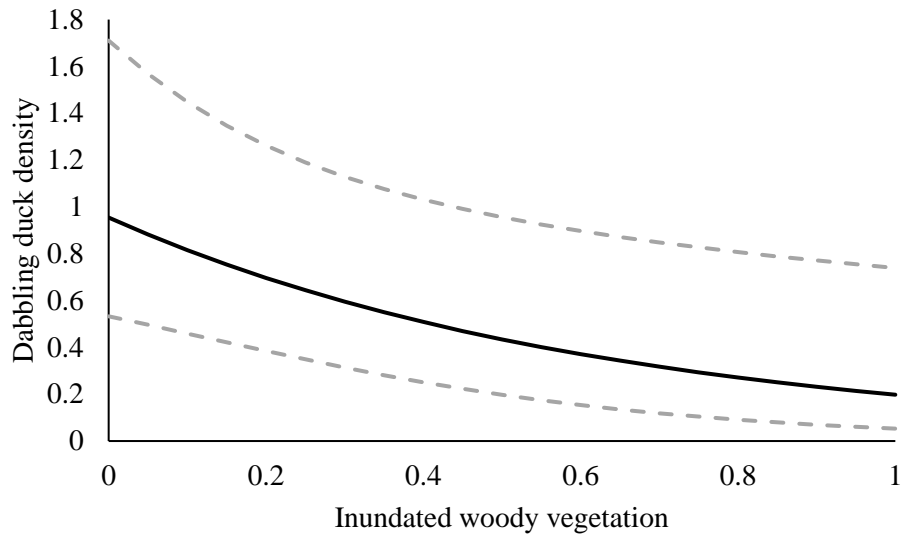


Figure 3.4. Model-predicted dabbling duck density (birds/ha) estimates (\pm 85% CI) by the proportion of inundated woody vegetation (i.e., scrub-shrub and forest) cover in NWI polygons at aerial survey locations in Illinois, U.S.A. during 2016 – 2017. Other variables were held constant at their means (SHALLOW=0.28, HERB= 0.026, and INUND= 0.50). Estimates were corrected for zero inflation by multiplying by the probability of a non-zero (i.e., $1 - p$), and zero inflation variables were held at their mean (Total_FO=0.44, Inund_open=0.31).

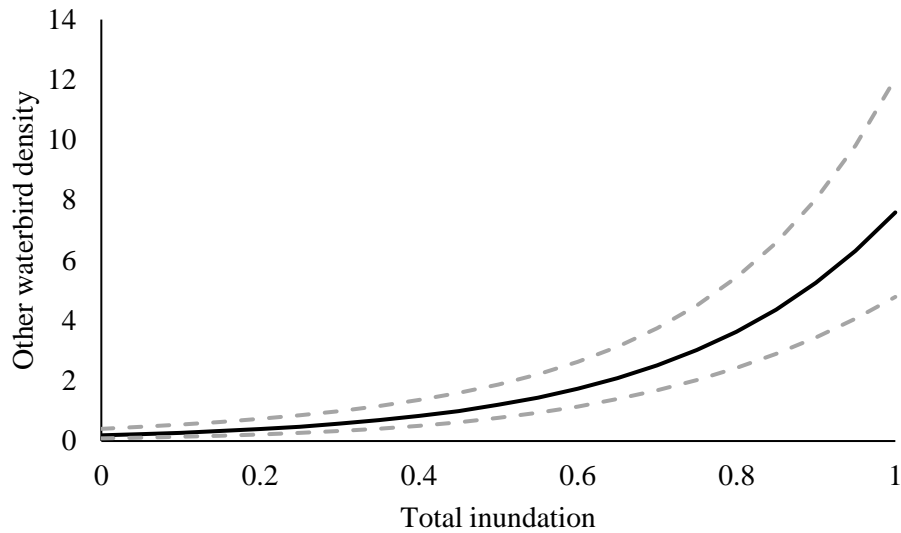


Figure 3.5. Model-predicted other waterbird (e.g., geese, swans, diving ducks, herons) density (birds/ha) estimates (\pm 85% CI) by the proportion of total inundation cover (i.e., all vegetation types and open water) in NWI polygons at aerial survey locations in Illinois, U.S.A. during 2016 – 2017. Other variables were held constant at their mean (HERB= 0.026, WOODY=0.14, and SHALLOW=0.28). Estimates were corrected for zero inflation by multiplying by the probability of a non-zero (i.e., $1 - p$), and zero inflation variables were held at their mean (Total_FO=0.44, Inund_open=0.31).

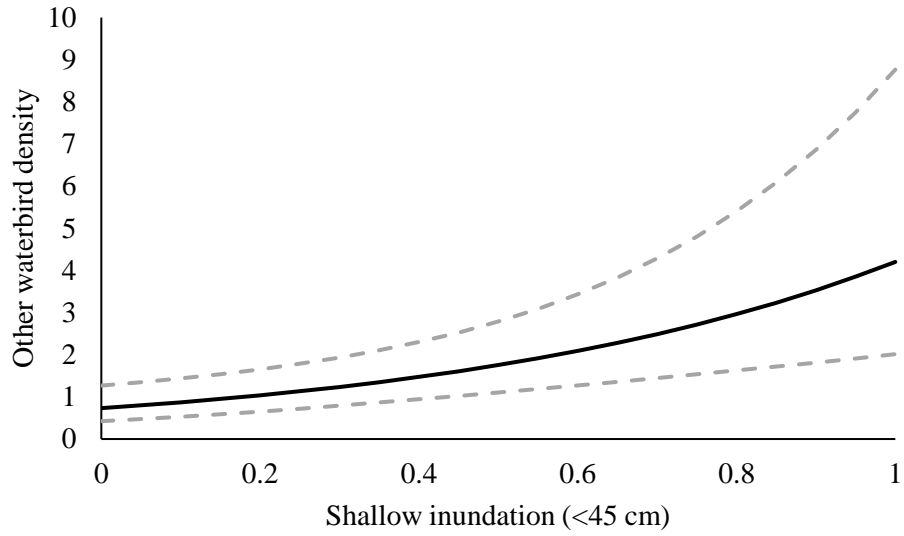


Figure 3.6. Model-predicted other waterbird (e.g., geese, swans, diving ducks, herons) density (birds/ha) estimates (\pm 85% CI) by the proportion shallow inundation (<45 cm) cover in NWI polygons at aerial survey locations in Illinois, U.S.A. during 2016 – 2017. Other variables were held constant at their mean (HERB= 0.026, P_inund_WOOD=0.14, and INUND= 0.50). Estimates were corrected for zero inflation by multiplying by the probability of a non-zero (i.e., 1 – p), and zero inflation variables were held at their mean (Total_FO=0.44, Inund_open=0.31).

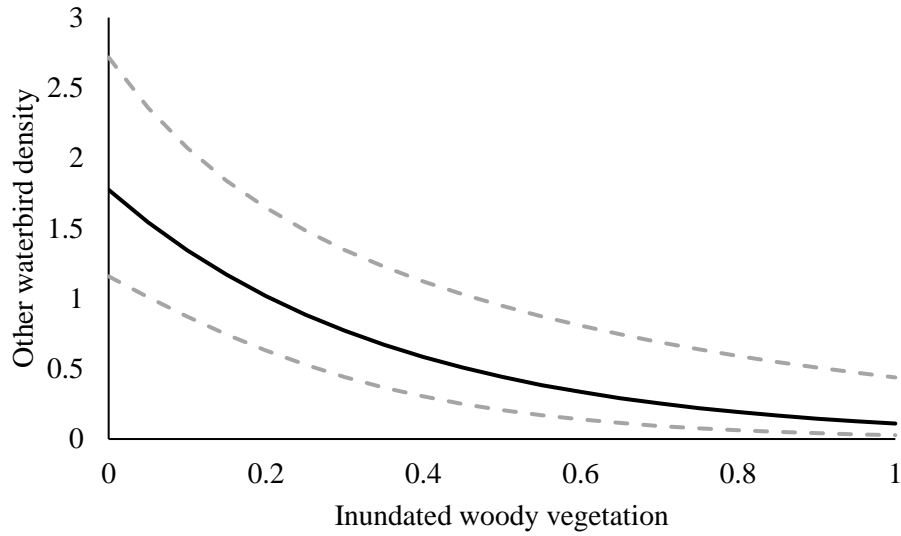


Figure 3.7. Model-predicted other waterbird (e.g., geese, swans, diving ducks, herons) density (birds/ha) estimates (\pm 85% CI) by the proportion of inundated woody vegetation (i.e., scrub-shrub and forest) cover in NWI polygons at aerial survey locations in Illinois, U.S.A. during 2016 – 2017. Other variables were held constant at their mean (SHALLOW=0.28, HERB=0.026, and INUND= 0.50). Estimates were corrected for zero inflation by multiplying by the probability of a non-zero (i.e., $1 - p$), and zero inflation variables were held at their mean (Total_FO=0.44, Inund_open=0.31).

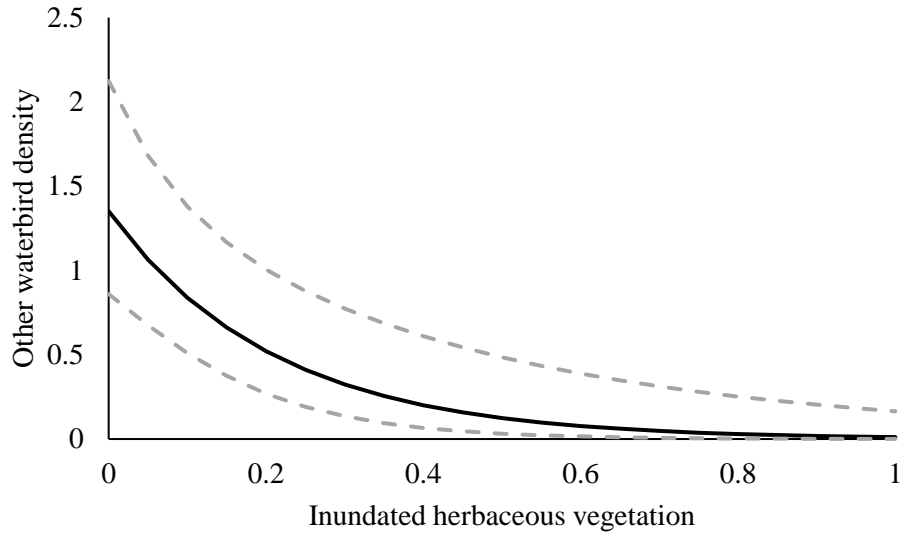


Figure 3.8. Model-predicted other waterbird (e.g., geese, swans, diving ducks, herons) density (birds/ha) estimates (\pm 85% CI) by the proportion of inundated herbaceous vegetation (i.e., short, tall and mixed) cover in NWI polygons at aerial survey locations in Illinois, U.S.A. during 2016 – 2017. Other variables were held constant at their mean (SHALLOW=0.28, WOODY=0.14, and INUND= 0.50). Estimates were corrected for zero inflation by multiplying by the probability of a non-zero (i.e., $1 - p$), and zero inflation variables were held at their mean (Total_FO=0.44, Inund_open=0.31).

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CHAPTER 4: HABITAT CHARACTERISTICS ASSOCIATED WITH MIGRATING AND BREEDING SORA IN ILLINOIS

4.1 ABSTRACT

I surveyed secretive marsh birds and other waterbirds associated with dense-emergent vegetation (hereafter, marsh birds) using call-broadcast surveys during mid-April – mid-June, corresponding to migration and breeding chronology in Illinois. I performed single-season occupancy modeling to index species occurrence while accounting for imperfect detection. I analyzed occupancy as a function of local wetland characteristics, landscape context, and management practices. I detected seven species of marsh birds during call-broadcast surveys, but I restricted analyses to sora (*Porzana carolina*) because they were highly correlated with detections of all marsh birds, and no other species had sufficient detections for individual occupancy models. Detection probability decreased with survey period. Local wetland characteristics including inundated persistent emergent vegetation, forest, and non-persistent emergent vegetation were predictors of sora site occupancy. For every 10% increase in inundated persistent emergent vegetation cover, the odds of sora occupancy increased 5.0 times (85% CI 1.5 – 16.8), whereas a 10% increase in inundated forested cover resulted in a 5.1% (0.4 – 57%) decrease in sora occupancy. Occupancy rates for sora in this study were substantially greater than other marsh bird species previously reported, and I suspect that detections included migrating individuals. Wetland managers should provide increased cover of inundated dense persistent emergent vegetation or a mix of emergent vegetation types rather than focus on inundated non-persistent emergent vegetation.

4.2 INTRODUCTION

Wetlands are declining globally, and some regions, such as the midwestern United States, have experienced extensive wetland losses since the 1900s (Tiner 1984, Mitsch and Gosselink 2000). For instance, Illinois lost >85% of its original 3.2 million ha of wetlands between 1780 and 1980 (Dahl 1990). Freshwater wetlands dominated by herbaceous hydrophytes (i.e., emergent; Cowardin et al. 1979) provide critical habitat for many species of wetland-dependent birds of conservation concern and have suffered the greatest rates of loss among all freshwater wetland types (Dahl 2011). Marsh birds, including rails (Rallidae), bitterns (Ardeidae), and grebes (Podicipedidae), are reliant on emergent wetlands for migrating and/or breeding habitat,

and most species have experienced widespread population declines since the 1970's (Eddleman et al. 1988, Ribic et al. 1999, Conway and Sulzman 2007, Ward et al. 2010). Population declines are presumably linked to losses in quantity and suitability of wetland habitat (Conway et al. 1994, Soulliere et al. 2007, Ward et al. 2010, Bolenbaugh et al. 2011, Conway 2011). The combination of emergent wetland loss and marsh bird population declines necessitates appropriate wetland management and studies regarding habitat use for these difficult to study species.

There is a paucity of information regarding marsh bird population trends and habitat associations across the United States. In particular, little is known about their population status and distribution in the Midwest, including Illinois (e.g., Darrah and Kremetz 2010, Bolenbaugh et al. 2011, Harms and Dinsmore 2013). Marsh birds migrate through and breed in Illinois (Kleen et al. 2004), and most require emergent vegetation, both non-persistent emergent (NPE; moist soil, *Polygonum* spp.) and dense persistent emergent vegetation (PEM; e.g., *Typha* spp.), for breeding and cover (Gibbs et al. 1992, Lor and Malecki 2006, Darrah and Kremetz 2010). Some marsh birds also avoid forested wetlands (Bolenbaugh et al. 2011). However, vegetation cover preference varies among species (Weller and Spatcher 1965, Lor and Malecki 2006). For instance, soras (*Porzana carolina*), least bitterns (*Ixobrychus exilis*), and Virginia rails (*Rallus limicola*) nest above water in dense emergent vegetation (Weller and Spatcher 1965). King rails (*Rallus elegans*), on the other hand, are associated with a variety of vegetation and wetland types ranging from freshwater and coastal salt marshes to scrub-shrub swamps and upland fields near marshes (Meanley 1969). Common gallinules (*Gallinula galeata*) occur in high density in semi-permanently flooded wetlands that have narrow-leaved persistent emergent vegetation and submerged vegetation for food (Brackney and Bookhout 1982).

In addition to vegetation cover, the structure and interspersion of vegetation and hydrological conditions are key components of marsh bird habitat. Marsh birds are often associated with areas of interspersed water and emergent vegetation. For instance, Rehm and Baldassarre (2007) and Darrah and Kremetz (2010) found that least bitterns had greater site occupancy with increasing interspersion of open water to vegetation. Further, Bolenbaugh et al. (2011) found that least bitterns and pied-billed grebes (*Podilymbus podiceps*) were positively associated with 50:50 interspersion of vegetation and water, whereas American bitterns

(*Botaurus lentiginosus*), Virginia rails, and soras were found in more highly vegetated wetlands (70–100% vegetation 30-0% water ratio). On the other hand, marsh birds are often associated with more permanent wetlands with stable water levels as this promotes development of dense emergent vegetation such as cattail (e.g., *Typha* spp.; Anteau 2012). Soras and Virginia rails breed in freshwater emergent wetlands with shallow to intermediate water depths (Griese et al. 1980, Johnson and Dinsmore 1986), whereas common gallinules use deeper, more open wetlands. Average water depth for common gallinule nests was 40 cm according to Miller (1946), whereas king rails nested in marshes with shallow (0–25 cm) inundation (Eddleman 1988).

While vegetation and hydrological characteristics influence the suitability of habitat for marsh birds, due to the highly fragmented landscape and extensive wetland loss in Illinois, remaining wetlands are often managed to provide wetland cover to benefit wildlife (Rundle and Fredrickson 1981, Gray et al. 2013). Wetland management (e.g., hydrological and vegetation control and/or manipulation) may impact marsh bird use of wetlands. For example, Monfils et al. (2014) studied breeding bird use of diked and undiked coastal marshes in Michigan and found greater abundances of American bittern, least bittern, and common gallinule in diked wetlands. And while agencies often strive for multispecies management, wetland management for waterfowl and waterfowl production remains a primary objective. However, Johnson and Dinsmore (1986) posit that waterfowl management could provide compatible breeding rail habitat if typical drawdowns and dewatering efforts could be delayed until after marsh bird breeding (e.g., see Rundle and Fredrickson 1981). This sentiment was echoed by Fournier et al. (2018) that found increased sora abundance associated with moist-soil vegetation cover.

In addition to local impacts from wetland management, landscape-level factors (Haig et al. 1998) such as wetland connectivity to other wetlands and to rivers or surrounding land use may impact marsh bird use of wetlands. Landscapes with high wetland density, especially emergent wetlands, are often considered more suitable for marsh birds (Tozer et al. 2010). Kahler (2013), for example, found an increase in common gallinule and Virginia rail abundance in areas with greater percent emergent marsh within 100-m and 10-km radii. Similarly, Smith and Chow-Fraser (2010) observed that more isolated marshes had lower species richness of obligate marsh-nesting birds than less isolated marshes. Wetland connectivity to rivers, on the

other hand, may have variable impacts on marsh bird use. For instance, Xia et al. (2016) found that wetland connectivity to large river systems influenced species richness and abundances of different foraging guilds; however, the guild which included swimming Rallids seemingly was not affected. Furthermore, impacts from disturbance or surrounding land use may influence marsh bird use of wetlands. Peterson and Niemi (2007) found that wetland birds responded to the degree of anthropogenic disturbance within and surrounding the wetlands, specifically with intensity of surrounding land use. Similarly, Ward et al. (2010) studied wetland-dependent birds in northeast Illinois and observed that increased development near wetlands appeared to alter hydrology and resulted in habitat degradation and population declines of several marsh bird species.

To improve management and conservation planning for marsh birds in Illinois, it is important to identify factors associated with marsh bird occupancy and habitat selection. However, due to the secretive nature of marsh birds and their selection of wetlands with dense emergent vegetation, accurate population estimates and information on habitat associations and use are difficult to determine (Eddleman et al. 1988, Conway and Gibbs 2005). For example, the Breeding Bird Survey does not adequately sample birds in densely vegetated wetlands (Bystrak 1981, Gibbs and Melvin 1993, Ward et al. 2010). My goal was to identify factors affecting marsh bird occupancy of wetlands during spring migration and the breeding season. I modeled occupancy relative to wetland vegetation and hydrological cover, vegetation interspersion, wetland management, wetland connectivity, and local and landscape anthropogenic disturbance for focal marsh bird species (e.g., IDNR 2005, Soulliere et al. 2007, Conway 2011). I hypothesized that marsh bird occupancy would be influenced by both local and landscape variables, and I predicted marsh bird occupancy would be positively related with cover of non-persistent vegetation, persistent emergent vegetation, and shallow surface water. In addition, I predicted that interspersed wetlands and highly connected wetlands with low levels of local and landscape disturbance would have greater probability of marsh bird occupancy, and, furthermore, marsh bird occupancy would be greater within wetlands with intermediate levels of hydrological wetland management that encouraged semi-permanent water regimes.

4.3 MATERIALS AND METHODS

Study Area

The study area for this project included wetlands outlined by National Wetlands Inventory (NWI; Cowardin et al. 1979) in the state of Illinois on both public and private land. My study spanned Illinois' 14 Natural Divisions, which are distinct ecoregions differentiated by topography, soils, glacial history, flora, and fauna (Schwegman 1973), with the Lake Michigan Natural Division excepted (Figure 2.1). Ten focal species of marsh birds migrate through or breed in Illinois (Kleen et al. 2004, Conway 2011). Three of the ten marsh birds included are Illinois state endangered (common gallinule, black rail [*Laterallus jamaicensis*], and king rail), whereas one species is state threatened (least bittern).

Procedures and Analyses

I surveyed habitat conditions in 2016 – 2017 for secretive marsh birds during mid-April – mid-June, corresponding to migration and breeding chronology in Illinois (Kleen et al. 2004, Conway 2011). Each year, I selected 100 – 120 plots (25 ha each) located within NWI-mapped wetlands and visited each plot once during the survey period. Plots were generated randomly using a spatially balanced stratification (Theobald et al. 2007) based on NWI wetland classes and Natural Divisions. Using the Reversed Randomized Quadrant-Recursive Raster tool (RRQRR) in ArcGIS 10.4 (ESRI 2011), I assigned spatially balanced 25-ha plots overlapping NWI-mapped wetland polygons (Theobald et al. 2007). From an earlier survey period (mid-February – mid-April) corresponding to spring migrating waterfowl, we randomly substituted 50.0% of plots dominated by forested polygons with emergent wetland polygons to better encompass migrating and breeding marsh bird habitat, as marsh birds are known to avoid forested wetlands (Bolenbaugh et al. 2011). I did this by increasing the inclusion probability of emergent wetland polygons, as classified by NWI, in the ArcGIS RRQRR tool. I then used Neyman allocation (Neyman 1934) to allocate survey effort proportionally based on NWI wetland area within Natural Divisions with a minimum of three plots per Natural Division. This sampling design with unequal effort among strata allowed for unbiased estimation (Johnson et al. 2009) and was flexible to uncertainty in removing and adding sites (e.g., denied access by landowner; Theobald

et al. 2007). Furthermore, this process has been shown to generate results representative of the larger population of unsampled wetlands (Miller 2016, Tozer et al. 2018).

From the spring habitat assessment, and with the addition of new plots with emergent NWI classes, I selected 30 plots in 2016 and 25 plots in 2017 (i.e., 55 plots total) with dense emergent vegetation (Conway 2011) where I then surveyed for secretive marsh birds (Figure 4.1). This selection process was not random, as each plot with suitable habitat (i.e., flooded emergent vegetation) was surveyed if the area of emergent vegetation was >0.5 ha (Conway 2011). Surveys were conducted following the North American Marsh Bird Monitoring Protocol (Conway 2011). I used call-response broadcasting to increase detection rates (Gibbs and Melvin 1993, Conway and Gibbs 2005, Lor and Malecki 2002). At each wetland plot, I conducted surveys at one or more points (i.e., survey route) if adequate spacing and vegetation (i.e., ≥ 400 m) allowed to avoid double-counting birds. I placed survey points on the edge of or within flooded emergent vegetation (Conway 2011). I visited each survey point within a plot three times at approximate two-week intervals during mid-April to mid-June during peak marsh bird breeding (Lor and Malecki 2002, 2006) and conducted surveys from 30 minutes prior to sunrise and within two hours post-sunrise when marsh birds are active (Conway 2011). I used a game caller (Western Rivers Pursuit, Maestro Game Calls, LLC., Dallas, Texas, U.S.A.; Primos Turbo Dogg, Primos Hunting, Flora, Mississippi, U.S.A.) to broadcast a recording beginning with a 5-minute passive period followed by 10 minutes of 1-minute alternating series of 30 seconds of calls from one species and 30 seconds of silence until all marsh bird species' calls were broadcasted. The species' call order began with the least intrusive species and proceeded to the more gregarious following recommendations by Ribic et al. (1999). Call order was the following: black rail, least bittern, yellow rail (*Coturnicops noveboracensis*), sora, Virginia rail, king rail, American bittern, common gallinule, American coot (*Fulica americana*), and pied-billed grebe. Broadcasts were projected at 80-90 decibels (Conway and Gibbs 2005). During call-response surveys, I pointed the game caller toward emergent vegetation and repeated subsequent surveys using the same cardinal direction. I did not conduct surveys during high wind (≥ 25 km/hr) or inclement weather that would decrease the observer's ability to hear waterbird responses.

Prior to conducting surveys, participants received standardized training on field protocols and bird identification through in-person training sessions, detailed guidebooks, and audiovisual

media (Conway 2011, Tozer et al. 2018). To account for ambient conditions and other variables that may impact detection probability, I recorded windspeed, ambient temperature (°C), weather/cloud cover, background noise, and observer initials for each survey (Conway 2011; Table 3.1). These variables were standardized (Conway 2011, North American Breeding Bird Survey 2011). Specifically, I recorded wind conditions using the Beaufort scale, weather/cloud cover using the National Weather Service sky code 0 – 9 (National Weather Service 2011), and background noise as an ordinal value from 0 indicating no background noise to 4 where intense noise likely limited an observer’s ability to hear marsh birds beyond 25 m (Conway 2011).

Following each call-response survey, I evaluated habitat conditions within 50-m radius of each survey point (Conway 2011). I first estimated the proportion of the survey area that was inundated. Next, I assessed the depth profile throughout the survey area and estimated the area that was shallowly (<45 cm) inundated. Additionally, I estimated the total area of open water and area of submerged and floating-leaved vegetation (i.e., aquatic bed; e.g., *Ceratophyllum* spp., *Potamogeton* spp.). I then estimated percent cover of inundated vegetation including woody vegetation (both scrub-shrub [<6 m] and forested [>6 m]) and emergent vegetation within the survey area. I split emergent vegetation into dense persistent emergent vegetation (PEM; e.g., *Typha* spp., *Scirpus* spp., *Phragmites australis*) and non-persistent emergent vegetation (NPE; e.g., *Sagittaria* spp., *Pontederia* spp.; Cowardin et al. 1979).

In a separate visit during mid-April – early-June, I assessed habitat conditions within the larger 25-ha plots which contained wetlands where call-response surveys were conducted (i.e., Chapter 1). During these visits, I mapped structural wetland composition (tall and short herbaceous, woody, mudflat, etc.; Table 2.3) and separated inundation types using a field PC (Juniper Systems, Archer) with a GPS on foot or by boat. I then digitized wetland inundation and cover to create thematic cover maps in ArcGIS for each wetland plot.

Using these thematic maps, I assessed complexity and interspersion of vegetation by extracting edge density (ED), a quantitative measure of the interspersion of vegetation and water (units of m/ha) (Rehm and Baldassarre 2007). I used the Spatial Statistics by Patch Analyst extension (i.e., the Fragstats interface) in ArcGIS (Rempel et al. 2012) to calculate ED of the

cover maps by taking the total length of edge in meters in a given NWI class and dividing it by the total area of the NWI class.

While mapping wetland inundation and vegetation cover, I noted evidence of wetland management, including evidence of drawdowns, presence of impoundments, mowing or disking, moist-soil plant management, and water-control structures (e.g., culverts, stop-log structures). Based on evidence of management while searching the 25-ha plots, I determined the intensity of management on an ordinal scale (e.g., none, low, medium, and medium/high). This scale ranged from, for example, unmanaged bottomland forests to highly managed duck club areas with water control, impoundments, and at times, planting of food plots specifically for waterfowl and wildlife.

To assess disturbance to wetland habitat and hydrology, I measured wetland integrity within the 25-ha plots using a modified Ohio Rapid Assessment (ORAM) Version 5.0 (Mack 2001). With an emphasis on anthropogenic disturbance, the ORAM scores wetland integrity using six metrics including surrounding land use, hydrology, habitat alteration and development, interspersions, and vegetation communities. I scored wetland polygons using a subset of the ORAM metrics and submetrics: 2b, 3a – e, 4a – c, 6a – d (Mack 2001; Table 2.5) on a gradient similar to ORAM's method of scoring wetlands along a quality gradient for regulatory purposes. I included this score for the wetland where a call-response survey point was located in my analysis to determine whether wetland disturbance influences marsh bird use or occupancy.

I further determined potential impacts of surrounding landscape on marsh bird occupancy in ArcGIS by calculating the Landscape Development Intensity Index (LDI; Brown and Vivas 2005). The LDI represents a measure of human disturbance on a gradient relative to intensity of land use (Brown and Vivas 2005, Mack 2006). For example, low LDI values (≤ 2) represent land uses that are more natural (e.g., wetlands, grasslands, forests), whereas high LDI values (>10) represent highly altered/disturbed surrounding land uses (e.g., row-crop agriculture, urbanization). I computed the LDI using the 2011 National Land Cover Dataset (NLCD; Fry et al. 2011) at a landscape-level 5-km scale (Harms and Dinsmore 2013). I calculated the proportion of different land uses surrounding sites in ArcGIS and assigned coefficients based on intensity of the land uses (following recommendations by Mack 2006).

To determine whether wetland connectivity could predict marsh bird occupancy, I included the proportion of a 100-m and 5-km buffer surrounding wetlands that was comprised of wetlands (i.e., both emergent and woody) as classified by NLCD in my analysis (e.g., Tozer et al. 2010). Additionally, observers scored NWI classes' connectivity to rivers or streams on an ordinal scale ranging from 0 – 3 (no connectivity to permanent connection; Table 2.4). These scores were based upon observations of hydrological characteristics (i.e., secondary indicators).

Statistical Analyses

I assessed marsh bird populations and habitat associations using single-season occupancy modeling to index species occurrence while accounting for imperfect detection probability (MacKenzie et al. 2002, MacKenzie and Royle 2005). I conducted single-season occupancy modeling using the unmarked package (Fiske and Chandler 2011) in R (Version 3.4.2; R Core Team 2017). The results of repeated call-response surveys were used to create detection histories for each survey point within a wetland. I used a two-step modeling process where models with covariates on detection were tested first while keeping occupancy constant (i.e., intercept only), and I then incorporated the top detection model in the occupancy models (Olson et al. 2005, Darrah and Krementz 2010, Harms and Dinsmore 2013). I assumed wetland plots were independent and closed to changes in occupancy throughout the survey season according to survey periods recommended by Conway (2011). I modeled sora occupancy, but low numbers of detections precluded analyses of other species.

I checked predictor variables for collinearity and excluded combinations of correlated variables ($r \geq |0.5|$), to avoid bias in parameter estimates and inflated standard errors (Zuur et al. 2010). Then, I developed candidate models based on a thorough review of the literature of factors that may influence detection and the occupancy of secretive marsh birds. Variables for the detection model included Julian date and adjusted date (date adjusted for North/South region timing; i.e., South region surveys began prior to North), temperature, background noise, agency (i.e., Illinois Natural History Survey or Southern Illinois University), weather/sky, wind, and start time of survey (Table 3.1). The candidate models for occupancy included wetland management, vegetation cover, hydrology, wetland connectivity, wetland complexity, and disturbance (Table 3.2). I included all additive combinations from the candidate model sets in

the model selection. I used Akaike's Information Criterion (AIC) adjusted for small sample sizes (AIC_c) to rank and evaluate models, and I examined models $\leq 2 \Delta AIC_c$ to determine whether they were competitive (e.g., uninformative parameters, Burnham and Anderson 2002, Arnold 2010). I obtained coefficient of determination (i.e., R^2) values according to Nagelkerke (1991) representing the proportion of variance in the dependent variable associated with the predictor variables. I calculated predicted estimates and 85% confidence intervals of site occupancy by marsh birds and obtained estimates for variables included in models $\leq 2 \Delta AIC_c$. I present effect sizes, 95% confidence intervals, and predicted estimates across the measured range of values for the covariate of interest while holding all other variables in the model at their mean.

4.4 RESULTS

I conducted marsh bird call-back surveys at 52 points within 30 wetland plots in 2016 and at 39 points within 25 wetland plots in 2017 in Illinois that contained inundated emergent vegetation. The proportion of sites where sora were detected (naïve occupancy, not accounting for detection probability) was 0.72 (0.63 – 1.00 [85% CI]).

Of the 10 focal species, I did not detect yellow rail, black rail, or king rail. The majority of detections were soras (66%), and I failed to detect marsh birds during any of three surveys at 34.5% of points. I recorded detections of 226 sora, 69 American coot, 20 Virginia rail, 12 pied-billed grebe, 9 least bittern, 5 American bittern, and 1 common gallinule (Table 4.3).

Sites where soras were detected contained on average 23% (19 – 27%) cover by inundated non-persistent emergent vegetation (NPE), 39% (35 – 43%) inundated dense persistent emergent vegetation, and 5% (4 – 5%) inundated forest, whereas sites without marsh bird detections contained 32% (27 – 36%) inundated non-persistent emergent, 33% (27 – 39%) inundated dense persistent emergent vegetation, and 14% (11 – 17%) inundated forest.

The best-supported detection model included adjusted date. The odds of detecting sora decreased 10.2% (7.9 – 12.5 [$R^2 = 0.48$]) each day of the survey period (Figure 4.2). Due to low model uncertainty ($w_i = 90\%$), I only included adjusted date detection in the occupancy modelling process.

The best-supported occupancy model for sora included the proportion of inundated persistent emergent vegetation and the proportion of inundated forest (Table 4.5). For every 10% increase in inundated dense persistent emergent vegetation cover within the survey area (i.e., 50-m radius surrounding survey point), the odds of sora occupancy increased 5.0 (1.5 – 16.8; Figure 4.3) times, whereas every 10% increase of inundated forest resulted in a 5.1% (0.4 – 57%) decrease in sora occupancy (Figure 4.4). One additional model ($<2 \Delta AIC_c$; $w_i = 0.18$) matched the best supported model and included a positive effect of non-persistent emergent vegetation, but 85% confidence intervals overlapped 0 (Table 4.5; Figure 4.5).

4.5 DISCUSSION

Local vegetation cover was the most important predictor of sora occupancy between mid-April to mid-June in Illinois, whereas adjusted date was the most important predictor of detection. Inundated persistent emergent vegetation and forest cover, and to a lesser extent, non-persistent emergent vegetation were important components predicting sora wetland use in Illinois, as others have found in Midwestern states. Generally, emergent vegetation was positively related to sora occupancy, but effects were variable and contingent on whether it was persistent or non-persistent. There was a markedly stronger relationship with increased inundated dense persistent emergent vegetation cover over non-persistent emergent cover. Other factors including surrounding land use, management intensity, wetland connectivity, and complexity had little effect on sora occupancy, and are likely overshadowed by the importance of dense persistent emergent vegetation, as it is limited in cover during spring migration and early summer breeding in Illinois (Chapter 2). Even in emergent NWI wetlands, average cover of inundated dense persistent emergent (20% [19.6 – 20.4%]) and non-persistent emergent vegetation (19% [18.4 – 19.1%]) was not substantial and composed only 5% and 9% of total NWI polygon area in Illinois respectively (Chapter 2). However, despite potentially limited habitat, this study suggests that soras are utilizing Illinois wetlands with dense persistent emergent vegetation, and managers should incorporate regions of dense persistent emergent vegetation to provide soras and other marsh bird habitat during migration and breeding.

Sora are sometimes considered a generalist rail species, and exhibit variable habitat use during migration and breeding, preferring more robust vegetation during breeding and non-persistent vegetation during migration. However, generally, soras select habitats with robust,

standing vegetation (Sayre and Rundle 1984). Likewise, Griese et al. (1980) found soras preferred cattail marshes for breeding in Colorado. Similarly, I found that sora occupancy greatly increased with inundated dense persistent emergent vegetation mid-April to mid-June in Illinois. Furthermore, sora occupancy was also positively related with non-persistent emergent cover, which supports findings by Wilson et al. (2018) that spring migrating soras were positively related to smartweed (*Polygonum* spp.) cover, a commonly encountered group of herbaceous species considered non-persistent emergent in our study. Further, Baschuck et al. (2012) found that increasing proportion of shallow-marsh vegetation increased sora densities, and in studying autumn migration habitat use, Fournier et al. (2018) noted that sora presence was positively related to annual moist-soil vegetation cover, both of which were considered non-persistent in our study. However, the variable non-persistent emergent cover was included in a competing model, had highly variable confidence intervals, and may instead be a pretending variable.

Since I classified emergent vegetation on persistence, this makes some comparisons to other studies that separated emergent vegetation based on height (i.e., short <1 m and tall >1 m) challenging. In general, persistent emergent species are considered tall (i.e., >1 m) and robust and included cattail (*Typha* spp.), bulrush (*Scirpus*, *Bulboschoenus* spp.), and common reed (*Phragmites* spp.), whereas non-persistent emergent are considered short and included sedges (*Carex*, *Cyperus* spp.), rushes (*Juncus* spp.), arrowheads (*Sagittaria* spp.), and reed canary grass (*Phalaris arundinacea*). I chose to differentiate emergent species this way as persistent emergent species provided erect structure during migration and early nesting for marsh birds in Illinois and provide cover through remaining vegetation from the previous growing season (Cowardin et al. 1979), whereas non-persistent species are often still growing and may not be suitable until they reach a specified height or density. For instance, areas of new sedge (e.g., *Carex* spp., *Cyperus* spp.) were not used by soras until they reached 20 – 30 cm above the water surface (Johnson and Dinsmore 1986).

I found a decrease in sora occupancy in sites with greater inundated forested vegetation cover, common to other marsh bird studies (e.g., Darrah and Krementz 2009, 2010; Harms and Dinsmore 2013). For instance, Fournier et al. (2018) also found a negative relationship between sora use and woody vegetation cover in Missouri wetlands. At the landscape-scale (5-km

radius), Bolenbaugh et al. (2011) found the amount of wooded wetland was negatively associated with occupancy by least bittern, rails (sora and Virginia), and an open-water group (pied-billed grebe, common moorhen, and American coot). Avoidance of wooded wetlands by marsh birds could be due in part to predator impacts as woody vegetation may serve as perches or corridors for avian or mammalian predators (Chamberlain et al. 2007) or perhaps the cover of or proximity to suitable emergent vegetation (Chapter 2). Efforts to reforest wetlands by programs such as the Wetland Reserve Program are substantial and likely meet habitat requirements of many wetland species (King et al. 2006), but, based on these results, these reforestation efforts may not benefit marsh birds.

Compared to other studies assessing marsh bird occupancy (Budd 2007 [0.18 – 0.27], Darrah and Krementz 2010 [0.21 – 0.31]), my estimate of site occupancy by soras (i.e., 0.80) was high. For instance, my estimates of sora occupancy were 2.6 – 3.8 times greater than least bittern and pied-billed grebe occupancy in the lower Illinois and Upper Mississippi River Valleys (Darrah and Krementz 2010). This likely reflects differences in species that were modeled. For instance, both Budd (2007) and Darrah and Krementz 2010 modeled site occupancy by least bitterns and pied-billed grebes, which may not represent sites used by soras, as pied-billed grebes select deeper, and potentially more open wetlands compared to other marsh birds (Lor and Malecki 2006, Harms and Dinsmore 2013). Furthermore, my estimate of detection (0.44) was greater than Darrah and Krementz (2010) where detection probability ranged 0.18 to 0.29. However, Darrah and Krementz (2009), while studying King rail occupancy in the Illinois and Upper Mississippi River Valleys, had similar detection probability (0.35 – 0.43). There are many variables that may influence detection probability including wetland size, as larger wetlands presumably support more nesting pairs than smaller wetlands (Johnson and Dinsmore 1986); thus, perhaps differences in detection probabilities observed here could be due to wetland size, marsh bird density, breeding stage (Robertson 2012) or other variables not measured directly in this study.

Most a priori models performed poorly, indicating no measurable effect on sora occupancy. Hydrological characteristics such as percent inundated or percent shallowly inundated were not important predictors of sora occupancy in this study. My results were similar to Baschuck et al. (2012) and Fournier et al. (2017) that found sora densities and Virginia rail

habitat selection were not affected by water depth, respectively. I also did not detect effects of interspersions (i.e., edge density), wetland management, wetland connectivity to rivers or other wetlands, and local or landscape disturbance on sora occupancy; however, the models including wetland management intensity and survey year performed better than the null indicating a general increase in sora occupancy with increased levels of wetland management and variable occupancy between years. Bradshaw (2018) reported that wetlands actively managed for waterfowl had lower overall marsh bird occupancy when compared to wetlands with more passive emergent marsh management in Illinois. However, Bradshaw (2018) focused on surveying public wetlands with a broader suite of management techniques than the random wetlands with emergent vegetation included in this study. Contrary to some studies (e.g., Conway et al. 1994, Darrah and Krementz 2010, Bolenbaugh et al. 2011), interspersions did not influence nest density of 5 marsh bird species in New York (Lor and Malecki 2006) or sora occupancy in this study. Also, regarding wetland connectivity, Harms and Dinsmore (2013) found no influence of wetland connectivity by including wetland area within 1, 3, and 5 km of study wetlands. Additionally, the coefficient of determination values for the best supported models were relatively low ranging from 16 to 18% (Nagelkerke 1991), indicating most variation in occupancy was not explained by local vegetation cover. Other variables such as distinct vegetation species (Conway and Sulzman 2007), food abundance (Baschuk et al. 2012), or other forms of disturbance that were not considered in this study could influence site occupancy by soras.

Single-season occupancy allows inclusion of time-varying covariates associated with detection; however, it does not allow time-varying covariates to be included in modeling occupancy (MacKenzie et al. 2002). Thus, this approach may not account for the phenology of non-persistent emergent vegetation, as some non-persistent species grow rapidly during the sampling period, whereas others reach maximum height and density much sooner, potentially confounding or obfuscating the slightly positive relationship observed in this study (Shaver and Melillo 1984, Squires and van der Valk 1992). For instance, Harms and Dinsmore (2013) found that an increase in the fast-growing reed canary grass (*Phalaris arundinacea*; considered non-persistent emergent in this study) increased marsh bird occupancy due to the dense cover it provided in their study. Thus, future studies should consider other approaches where time-

varying covariates may be included to determine the influence of spatial changes in vegetation cover on marsh bird occupancy or use.

I found that detection of soras varied over time. Specifically, detection probability decreased as the survey periods progressed. There are multiple factors that may influence detection (Conway and Gibbs 2011), and others have found variable detection probabilities while studying marsh birds (Gibbs and Melvin 1993, Lor and Malecki 2002). Conway (2011) recommended that initial surveys be conducted after migration and before the initiation of breeding due to the assumption that marsh birds rarely vocalize during migration and that observers are detecting breeding birds. This may be a reasonable assumption in some cases as Glahn (1974) did not detect marsh birds until territory formation and courtship began and noted that after the last of June in Colorado, very few rails that were known to be present responded to played-back calls. Variation in detection probability has been seen in other studies (Rehm and Baldassarre 2007) and may be due to marsh birds vocalizing primarily during a short period, largely during nest initiation and early egg-laying (Tozer et al. 2016). Thus, lack of responses and lower detection probability as the survey period progressed may not be indicative of the absence of marsh birds. However, contrarily, Kaufmann (1983) observed that marsh birds gave similar vocalization during migration and breeding, and Robertson (2012) found that Virginia rail and sora responded similarly to call-broadcast surveys during egg laying, incubation, and hatching. Since Illinois is on the edge of some marsh bird breeding ranges, and some studies at similar latitudes consider soras to be migrants (e.g., Harms and Dinsmore 2013), further information is needed to determine whether the sampling periods suggested by Conway (2011) are suitable for sampling breeding marsh birds in Illinois or whether migrants are included.

If migrant birds were included in my occupancy analysis, the closed population assumption of single-season occupancy modeling was likely breached. However, according to Bailey and Adams (2005), if this is the case, occupancy estimates may be biased; however, researchers may still gain valuable insight into the proportion of sites used by the target species. Additionally, the direction of the bias is influenced by the direction of species' movements (Bailey and Adams 2005). Furthermore, Kendall (1999) discusses robustness to violations in the closure assumption and potential alternatives, of which, the emigration only scenario may best fit the potential migratory movement observed in my study. The emigration only scenario may

arise when all animals are present in a study area when sampling begins, but, during the course of sampling, some individuals begin moving out of the study area (e.g., to more northern breeding areas). Kendall (1999) also noted that if detected and non-detected individuals leave with the same probability, the population size estimate at the time of the first “capture” or detection may be unbiased. Here, I believe some migrants were included in the analysis, thus results represent habitat use during both the migration and breeding period in Illinois. However, further information regarding the timing of marsh bird vocalizations and conducting call-back surveys more frequently during short (i.e., several days) time periods would help determine whether these adjustments in analysis or interpretation are necessary.

Here, I demonstrated that local vegetation cover was the most important predictor of sora occupancy between mid-April to mid-June in Illinois wetlands, whereas adjusted date was the most important predictor of detection. Both the Illinois DNR and the waterbird strategy committee of the Upper Mississippi River Great Lakes Region Joint Venture (Soulliere et al. 2007) posit that breeding habitat is likely limited in this region, and my research supports this notion. Even in emergent NWI wetlands, average cover of inundated dense persistent emergent (20% [19.6 – 20.4%]) and non-persistent emergent vegetation (19% [18.4 – 19.1%]) was not substantial and composed only 5% and 9% of total NWI polygon area in Illinois respectively (Chapter 2). While other local and landscape factors such as wetland connectivity or surrounding land use likely influence marsh bird use of wetlands, such limited suitable habitat may reduce influence of these effects. My research also suggests soras are more common in Illinois during this survey period than other marsh bird species, so future emphasis into the habitat preferences and wetland use of other marsh bird species should be a priority in Illinois. Wetland managers should focus on providing increased cover of inundated dense persistent emergent vegetation or a mix of emergent vegetation types rather than focus on inundated non-persistent emergent vegetation if the goal of management includes site occupancy by soras or other marsh birds.

4.6 TABLES AND FIGURES

Table 4.1. Variables and variable descriptions for the detection (p) model selection.

Variable	Variable description
Adjusted date	Date adjusted for North and South region start date
Agency	Illinois Natural History Survey (INHS) or Southern Illinois University (SIU)
Background noise	Ordinal variable associated with ambient noise at a site (0, no background noise, to 4, where intense noise likely limited an observer's ability to hear marsh birds [Conway 2011])
Latitude	Latitude of site location measured in degrees.
Julian date	Date represented by integer following Julian calendar
Null	Intercept-only model
Sky/weather	National Weather Service sky code 0 – 9 (National Weather Service 2011)
Start time	Start time of call-response survey
Temperature	Temperature in Celsius.
Wind	Beaufort scale; ordinal variable associated with wind speed

Table 4.2. A priori model sets, variables, variable code, and descriptions of variables included in the occupancy (Ψ) model selection.

Model	Variable(s)	Variable Code	Variable description
Complexity	Edge density	ED	Open water vegetation edge divided area to represent interspersion
Connectivity	River connectivity	RC	A factor variable scoring a wetland's connectivity to rivers ranging from 0 (no connected apparent) to 7 (permanent connection)
Cover	Prop wetlands (100m)	WET_100m	Proportion of wetlands (woody and emergent) in surrounding 100-m buffer
	Prop wetlands (5km)	WET_5km	Proportion of wetlands (woody and emergent) in surrounding 5-km buffer
	Forest	FO	Cover of inundated forested within 50-m radius of call-response survey point
	Non-persistent emergent	NPE	Cover of inundated non-persistent emergent vegetation (e.g., moist-soil, <i>Sagittaria</i> spp.) within 50-m radius of call-response survey point
	Persistent emergent	PEM	Cover of inundated dense persistent emergent vegetation (e.g., <i>Typha</i> spp.) within 50-m radius of call-response survey point
Disturbance	Open water	OW	Cover of inundated open water within 50-m radius of call-response survey point
	LDI 5km	LDI_5km	Landscape development intensity score for surrounding 5-k buffer
	ORAM score	ORAM	Score representing anthropogenic disturbance and alterations to natural wetland hydrologic regime or habitat, ranging from 0 (highly disturbed/impacted) to 100 (pristine unaltered wetland conditions)
Hydrology	Percent inundated	INUND	Proportion of 50-m radius survey area surrounding call-response point that is inundated
	Percent shallow	SHALLOW	Proportion of 50-m radius survey area surrounding call-response point that is inundated <45cm
Latitude	Latitude	LATITUDE	Latitude of site location measured in degrees.
Management	Management intensity	M_INT	None, low, medium, high
Year	Year	YEAR	Year of survey, either 2016 or 2017

Table 4.3. The number of marsh bird detections in 2016 and 2017 by species. Call-response surveys were used to elicit responses and marsh birds were recorded if they were seen or heard. Note the number of detections may not equal the number of individuals due to auditory detections and the movement of individuals (i.e., could have double-counted individuals).

Species	2016	2017	Total
American Bittern	2	3	5
American Coot	63	6	69
Black Rail	0	0	0
Common Gallinule	0	1	1
King Rail	0	0	0
Least Bittern	0	9	9
Pied-billed Grebe	7	5	12
Sora	98	128	226
Virginia Rail	13	7	20
Yellow Rail	0	0	0
Total	183	159	342

Table 4.4. Rankings of detection (p) models according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K), and model weights (w_i). Occupancy was held constant while testing for detection.

Model	K	AIC_c	ΔAIC_c	w_i
Adjusted date	3	240.20	0.00	0.90
Julian date	3	244.53	4.33	0.10
Temperature	3	269.89	26.69	0.00
Start time	3	287.82	47.93	0.00
Background noise	6	288.83	48.58	0.00
Null	2	294.60	48.69	0.00
Agency	4	294.19	50.70	0.00
Latitude	3	291.03	50.83	0.00
Sky/weather	8	292.35	52.15	0.00
Wind	7	298.42	53.60	0.00

Table 4.5. Rankings of models according to Akaike's Information Criterion for small sample sizes (AIC_c) with the number of parameters (K), and model weights (w_i) for predicting total marsh bird occupancy (Ψ) with top detection (p) variable adjusted date. Covariates listed below. See Table 4.2 for further descriptions.

Model	K	AIC_c	ΔAIC_c	w_i
FO + PEM	5	225.47	0.00	0.43
FO + NPE + PEM	6	227.28	1.81	0.18
FO + OW	5	227.64	2.17	0.15
FO	4	228.84	3.37	0.08
FO + NPE + OW	6	229.51	4.04	0.06
FO + OW + PEM	6	229.92	4.45	0.05
FO + NPE	5	230.79	5.32	0.03
Cover [FO + NPE + OW + PEM]	7	231.90	6.43	0.02
PEM	4	233.76	8.29	0.01
NPE + PEM	5	235.82	10.35	0.00
Year	4	237.82	12.35	0.00
M_INT	7	238.00	12.53	0.00
OW	4	238.67	13.20	0.00
OW + PEM	5	238.74	13.27	0.00
Null	3	240.20	14.73	0.00
NPE + OW	5	240.61	15.14	0.00
NPE	4	241.03	15.56	0.00
NPE + OW + PEM	6	241.04	15.57	0.00
Disturbance [LDI_5k + ORAM]	4	241.04	15.57	0.00
LATITUDE	4	241.48	16.01	0.00
LDI_5 km	4	241.60	16.13	0.00

Table 4.5. Continued.

Model	K	<i>AIC_c</i>	ΔAIC_c	w_i
RC	4	241.64	16.17	0.00
SHALLOW	6	241.67	16.20	0.00
INUND	6	241.92	16.45	0.00
WET_5km	6	241.98	16.51	0.00
WET_100m	5	241.99	16.52	0.00
RC + WET_100m	5	242.05	16.58	0.00
ORAM	7	242.42	16.95	0.00
Connectivity [RC + WET_100m + WET_5km]	4	243.49	18.02	0.00
RC + WET_5km	4	243.60	18.13	0.00
Hydrology [INUND + SHALLOW]	4	243.79	18.32	0.00
WET_100m + WET_5km	5	244.12	18.65	0.00
Complexity [ED]	4	245.64	20.17	0.00

Covariates for occupancy include: ED – edge density; RC – river connectivity; WET_100m – Proportion of wetlands (woody and herbaceous) in 100-m buffer; WET_5km – Proportion of wetlands (woody and herbaceous) in 5-km buffer; FO – proportion of inundated forest within 50-m radius of call-response survey point; NPE – proportion within 50-m radius of call-response survey point that consists of inundated non-persistent emergent vegetation; PEM – proportion within 50-m radius of call-response survey point that consists of inundated dense persistent emergent vegetation; OW – proportion of open water within 50-m radius of call-response survey point; LDI_5km – Landscape Development Index score for surrounding 5-km buffer; ORAM – modified Ohio Rapid Assessment Method score; INUND – proportion of 50-m radius from call-response survey point that is inundated; SHALLOW – proportion of 50-m radius from survey point that is inundated <45 cm; M_INT – management intensity (non, low, medium, high); Year – year of survey, either 2016 or 2017.

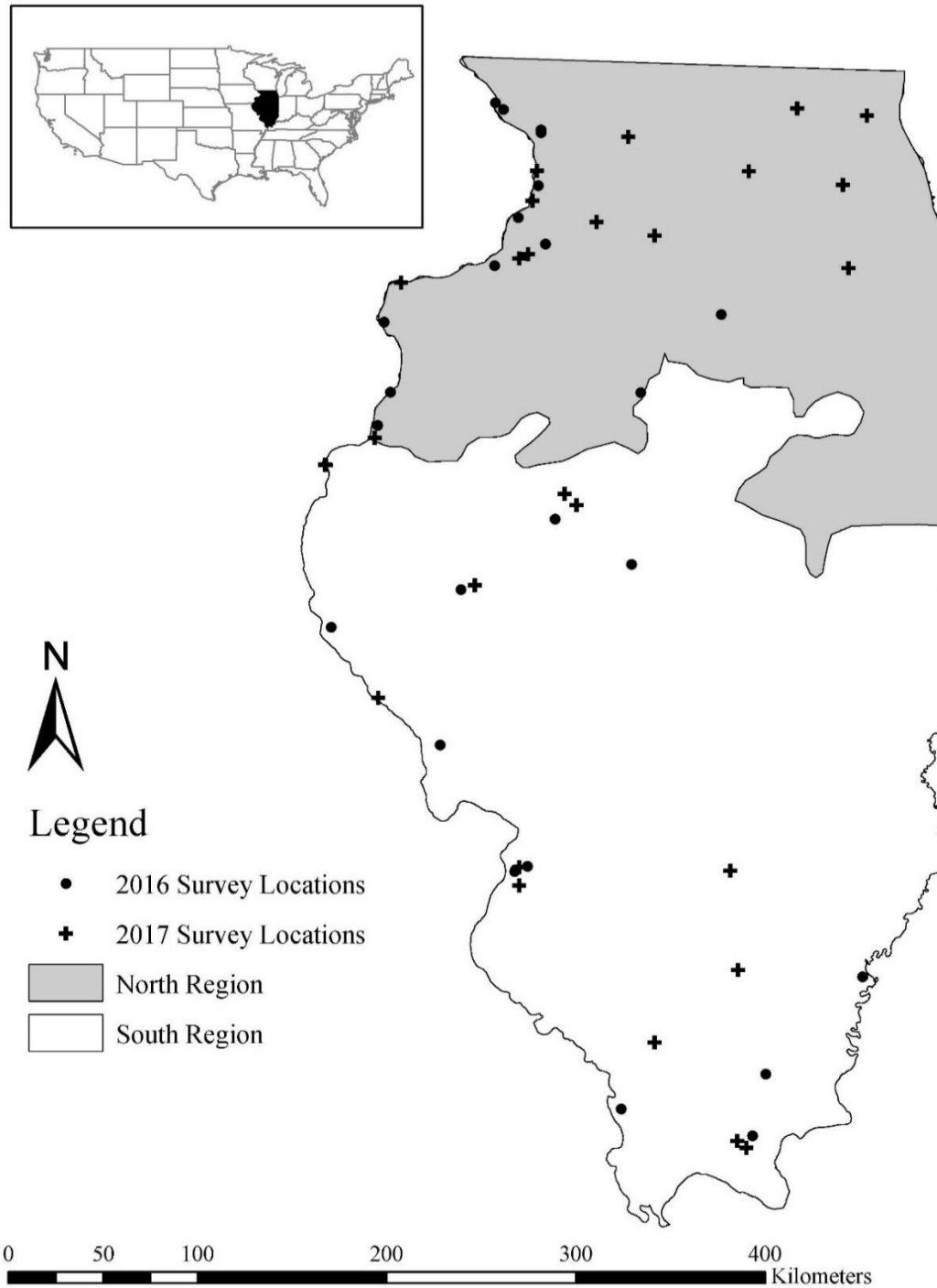


Figure 4.1. Locations of call-response surveys for marsh birds for 2016 – 2017. Surveys were conducted by Illinois Natural History Survey and Southern Illinois University personnel during the summer survey period (Mid-April to mid-June) corresponding to marsh bird breeding in the region. The North and South regions are based on the average maximum temperatures in May, from PRISM at Oregon State University (Conway 2011).

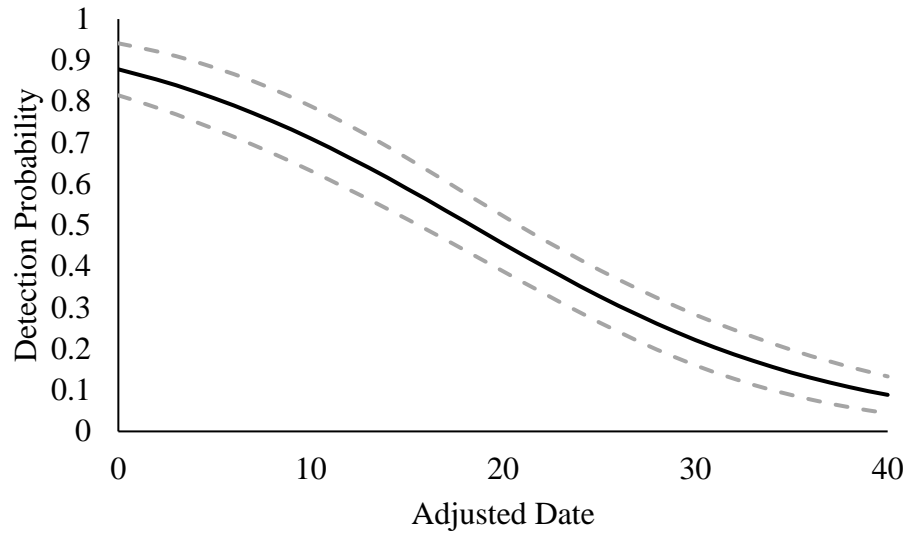


Figure 4.2. Detection probability (85% CI in grey) of sora across adjusted dates for marsh bird surveys conducted throughout Illinois during spring/summer (mid-April to mid-June) 2016 – 2017.

Adjusted date is date adjusted for North/South region survey timing (i.e., South region surveys began prior to North).

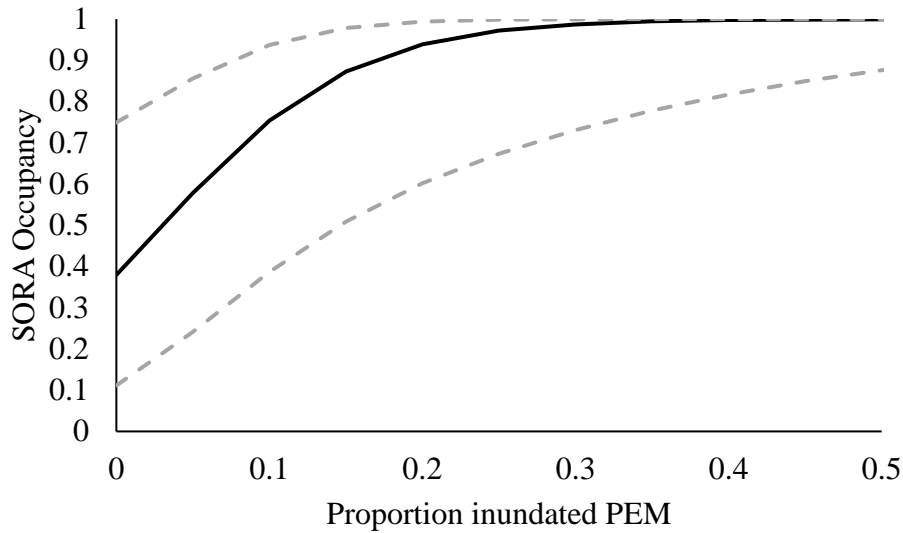


Figure 4.3. Model predicted relationship (85% CI in grey) between proportion of inundated dense persistent emergent vegetation (PEM) and probability of site occupancy for soras during surveys conducted throughout Illinois during spring/summer (mid-April to mid-June) 2016 – 2017. Predicted values were calculated holding the proportion of inundated forest at its mean (0.077).

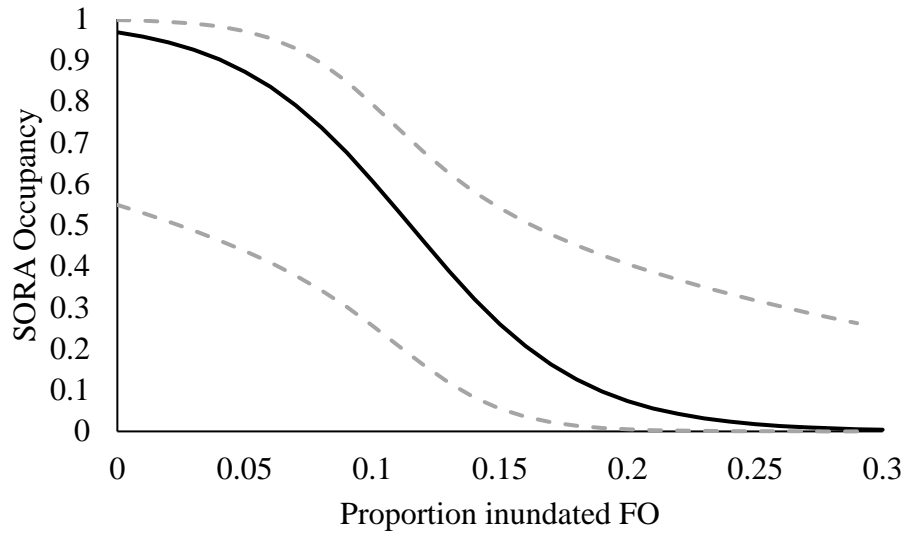


Figure 4.4. Model predicted relationship (85% CI in grey) between proportion of forest (FO) and probability of site occupancy for soras during surveys conducted throughout Illinois during spring/summer (mid-April to mid-June) 2016 – 2017. Predicted values were calculated holding the proportion persistent emergent vegetation at 0.10.

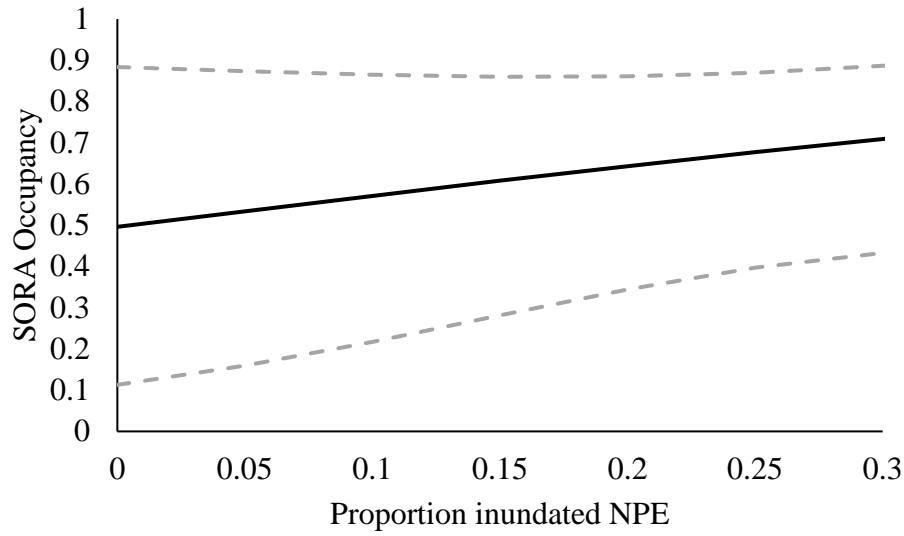


Figure 4.5. Model predicted relationship (85% CI in grey) between proportion of inundated non-persistent emergent vegetation (NPE) and probability of site occupancy for soras during surveys conducted throughout Illinois during spring/summer (mid-April to mid-June) 2016 – 2017. Results are from a competing model.

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CHAPTER 5: SUMMARY

Wetland-dependent birds such as waterfowl, marsh birds, and shorebirds, rely on wetlands to meet dynamic physiological needs. However, Illinois has undergone extensive wetland loss, and due to the importance of habitat throughout the annual cycle, it is imperative to conserve and manage remaining wetland habitat to meet the needs of wetland-dependent wildlife. However, extant wetlands are often degraded, and current wetland databases, such as the National Wetlands Inventory (NWI), do not accurately describe the suitability of available wetland habitat for waterbirds.

In Chapter 2, I assessed habitat conditions in spring, summer, and autumn, corresponding to critical time periods that could have population-level demographic effects for the focal waterbird guilds. I demonstrated that suitable vegetation and water cover did not compose a substantial portion of NWI wetland and deepwater habitats during any of the survey periods. Furthermore, important cover varied between NWI classes and Illinois Natural Divisions. Shallow inundation (<45 cm) considered suitable foraging habitat for dabbling ducks was less limited with an estimated 29% cover of NWI polygon area in Illinois during spring migration; however, this included extremely dense vegetation that likely was inaccessible to some waterbirds. Furthermore, hemi-marsh vegetation cover, often considered suitable for many waterbirds including dabbling ducks and marsh birds, was infrequently encountered, and suitable habitat for migrating and breeding marsh birds and migrating shorebirds was particularly limited during critical periods. For secretive marsh birds, cover of important emergent vegetation was low with an estimated 5% cover of inundated dense persistent emergent vegetation (PEM; e.g., *Typha* spp.) and only 9% inundated non-persistent emergent vegetation (NPE; e.g., moist-soil) across NWI polygon area. Estimated mudflat cover for migrating shorebirds was also low (e.g., $\leq 6\%$), but was greater in late spring / early summer than autumn (e.g., 6% vs. 4%) during 2016 – 2017, suggesting that mudflats were more limiting during autumn migration in Illinois. My study revealed that wetlands with greater complexity and connectivity to other wetland types offer the greatest amount of suitable inundation and cover and highlights the disconnect between local-scale hydrological and vegetation cover and wetland or land use databases which may not accurately describe spatiotemporal wetland conditions.

In Chapter 3, I modeled dabbling duck and other waterbird (diving ducks, geese, herons, coots, etc.) density by local and landscape characteristics during spring migration in Illinois. I found that local hydrological and vegetation cover were the most important predictors of waterbird density, followed by wetland connectivity to rivers and wetland density within 5 km. For every 10% increase in the proportion of total surface inundation cover, dabbling duck density increased by 34.8%, whereas other waterbird density increased by 44.6%. Furthermore, dabbling duck density increased by 26.5% for every 10% increase in the proportion of shallow (<45 cm) inundation cover. Waterbird density was negatively related to woody and herbaceous vegetation cover potentially due to perceived risk associated with more vegetated wetlands or limited food availability. This study highlights the majority of NWI area does not contain suitable foraging habitat during spring migration, and, when tied with limited food abundance, the importance of spring migration in subsequent breeding success should be further examined.

In Chapter 4, I surveyed secretive marsh birds and other waterbirds associated with emergent vegetation communities using call-broadcast surveys during mid-April – mid-June, corresponding to spring migration and breeding periods of marsh birds in the Midwest. I analyzed occupancy as a function of local wetland characteristics, landscape context, and management practices. I detected 7 species of marsh birds during call-broadcast surveys, but I restricted analyses to soras (*Porzana carolina*) because they were highly correlated with detections of all marsh birds and no other species had sufficient detections for individual models. Detection probability decreased with date, and thus declined across survey periods. Local wetland characteristics including inundated persistent emergent vegetation, forest, and non-persistent emergent vegetation were predictors of sora site occupancy. For every 10% increase in inundated persistent emergent vegetation cover, the odds of sora occupancy increased 5.0 times (85% CI 1.5 – 16.8), whereas a 10% increase in inundated forested cover resulted in a 5.1% (0.4 – 57%) decrease in sora occupancy. The occupancy rates for soras in this study were substantially greater than for other marsh bird species previously reported. Wetland managers should provide increased cover of inundated dense persistent emergent vegetation or a mix of emergent vegetation types rather than focus on inundated non-persistent emergent vegetation to provide migrating and breeding habitat during spring and early summer for soras and potentially

other marsh bird species. Managers need to retain water in places to promote the growth of persistent emergent vegetation.

As wetland and waterbird research in Illinois continues, research objectives and emphasis should be placed on identifying variables appropriate for predicting wetland suitability for waterbirds by combining National Wetlands Inventory data with other available spatial data and addressing demographic responses (e.g., survival, nest success, breeding propensity) of waterbirds to habitat suitability in Illinois. Furthermore, the decoupling of actual, on-the-ground vegetation and hydrological cover and various spatial databases should be further investigated. Results of this study may be used by the Upper Mississippi River and Great Lakes Region Joint Ventures and Illinois DNR, and this research sets the stage for future waterbird habitat research at varying spatial scales in Illinois (i.e., Natural Divisions) and a platform to tie suitable wetland conditions to waterbird demographics.