

MULTI-SCALE ASSESSMENT OF FACTORS AFFECTING MARSHBIRD DENSITY IN
THE NORTHERN PLAINS

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MASTER OF SCIENCE

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ABSTRACT

Over the last two centuries >50% of North American wetlands have been drained for agricultural production. Consequently, wildlife that depend on wetlands are declining and are of high conservation concern. The Prairie Pothole Region (PPR) of North America is considered one of the most important wetland regions in the world. It has been estimated that approximately only 35% of the wetland area remains in the PPR, yet it is still a stronghold for many wetland organisms and provides habitat for >100 wetland-dependent birds. We investigated habitat selection and abundance of marshbirds across multiple scales during the 2016-2017 breeding seasons. We found multiple scales to affect marshbird species abundance, and densities estimates that were amongst the highest reported in the PPR. Overall, this information provides a baseline for species with previously unknown densities in this region that will improve our ability to conserve marshbirds in the Northern Great Plains.

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CHAPTER 1. DENSITY AND ABUNDANCE OF SECRETIVE MARSHBIRDS IN NORTH DAKOTA

Introduction

Human land use change has led to losses in biodiversity worldwide (Foley et al. 2005, Butchart et al. 2010, Cardinale et al. 2012, Hooper et al. 2012). Many factors have contributed to these losses including direct habitat destruction, urban expansion (Kalnay and Cai 2003), overexploitation (Pimm and Raven 2000), and the degradation of soil and water from poor farming practices and pesticide use (Pimentel et al. 1992). Additionally, changes to the hydrologic cycle from agricultural intensification and irrigation (Donald 2004) have also contributed to the declines in biodiversity (WWF 2016). These factors in addition to the needs of a growing human population have led to drastic changes in freshwater ecosystems. For example, agriculture now accounts for ~85% of global water use (Gleick 2003). Moreover, according to a recent global review, 87% of wetland area has been lost during the last 300 years (Davidson 2014), and in the United States, >50% of wetlands have been drained specifically for intensive agriculture (Tiner 1984). This loss of wetlands coupled with excessive nutrient inputs, increased sediment loads, and agriculture chemicals have all contributed to degrading and declining wetland ecosystems (Foley et al. 2005). As a result, preserving remaining wetland systems should be a high priority for the conservation of biodiversity and ecosystem services that these systems provide.

Wetland ecosystems provide numerous ecological services to society. At the global level, they aid in the stability of available levels of nitrogen, atmospheric sulfur, and carbon dioxide (Mitsch and Gosselink 2007). At more localized levels, wetlands provide water filtration and purification, raw material, food, recreation and aesthetic value, nutrient cycling, and flood water

retention, while also harboring multiple threatened and endangered species (Costanza et al. 1997, Hansson et al. 2005). Over the past two centuries, the United States has lost approximately 47.0 million hectares of wetland ecosystems, with the majority comprised of inland freshwater wetlands that vary in degree of emergent vegetation and open water (Tiner 1984). These losses have disproportionate effects on biodiversity because freshwater wetlands are one of the most diverse and productive ecosystems relative to their size (Hansson et al. 2005). For instance, according to the Living Planet Index, between 1970 and 2012, freshwater wetland-dependent species experienced a 39% decrease in abundance (WWF 2016). These losses underscore the importance of monitoring wetland wildlife to create and execute proactive conservation actions.

Monitoring wetland flora and fauna is essential to effective conservation management by detecting plant invasions and identifying population declines prior to the risk of extinction (Hagan 1992). Wetlands are vulnerable systems and especially susceptible to invasions. For example, 24% of the world's most invasive plants are wetland species (Zedler and Kercher 2004). Several of these invaders (i.e., *Typha x glauca* exotic hybrid cattail) form monotypic stands, which lower wetland biodiversity, change hydrology, and alter vegetative structure (Zedler and Kercher 2004). Additionally, 9.5% (125,000) of all known animal species, and one-third of all vertebrates, are freshwater species despite the fact inland freshwater covers <1% of the planet's surface (Balian et al. 2008). Therefore, wetlands are biodiversity hotspots and with all of the anthropogenic pressures on remaining wetlands, they are highly susceptible to species losses (Dudgeon 2010).

The widespread loss and degradation of wetland systems have resulted in a high conservation priority status for many wetland wildlife species. Of particular concern, are wetland dependent birds (NABCI 2016). These species include all individuals that rely on wetland

systems for most or all of their reproductive activities. Within the broad group of wetland birds, a subset of these species that include rails (*Rallidae*), bitterns (*Ardeidae*), and grebes (*Podicipedidae*) are referred to as “secretive marshbirds” and are of great scientific interest because of our limited knowledge base of these species and their conservation concern status (Conway 2009). Limited information on these species is compounded by their cryptic coloring, infrequent vocalization, and generally inconspicuous behavior which is why they are referred to as secretive (Conway 2009). Recent research has tried to address many information gaps that could improve conservation for this suite of species. However, despite this increased focus on secretive marshbird research in recent years, there are still many distributional and ecological facets related to their conservation that are not well understood (Fournier et al. 2017). To date, no published studies have investigated the influence of vegetation components and landscape characteristics on secretive marshbirds in the Prairie Pothole Region (PPR) of North Dakota. Additionally, few studies have utilized distance sampling as a metric for estimating how these parameters influence secretive marshbird abundances while accounting for imperfect detection.

Of the states within the PPR, wetland surface area was highest in North Dakota (Dahl 2014). As a result, this area should act as a stronghold for secretive marshbird populations, making it an ideal location for gaining information on their ecological and conservation needs. Therefore, we conducted a two year study at the Chase Lake Wetland Management District which is managed by the United States Fish and Wildlife Service and consists of approximately 17,400 ha and is centrally located within the PPR of North Dakota. Our main objective was to examine local and landscape factors influencing secretive marshbird abundances. In order to accomplish this, we utilized distance sampling along with call-broadcast methodologies following the North American Marshbird Monitoring Protocol (Conway 2009). Based on limited

previous research, we hypothesized that at local scales, wetland emergent vegetation would be important in determining species' abundances while at larger scales, the amount of wetlands or developed areas (e.g., agricultural fields) would be primary factors influencing abundance.

Methods

Site Description

Our study was conducted on public lands managed by the Chase Lake Wetland Management District of North Dakota, USA (Figure 1.1). The Chase Lake Management District is in the Prairie Pothole Region of North Dakota and is characterized by rolling hills and high wetland densities of up to 50 potholes^{km²} (Dahl 2014). The region has a temperate climate with cold winters and warm, dry summers. The historical average temperatures for May, June, and July are 12.8, 18.1, and 21.2° C and rainfall totals average 6.7, 8.7, and 8.3 cm for May, June, and July, respectively (NDAWN 2017). The Chase Lake Wetland Management District consists of approximately 17,400 hectares (ha), with roughly 5,260 ha in native prairie, and over 4,600 ha of temporary, seasonal, semi-permanent and permanent wetlands. The predominant upland vegetation is a mixture of short and mid-grasses comprised of little bluestem (*Schizachyrium scoparium*), blue grama (*Bouteloua gracilis*), and western wheatgrass (*Pascopyrum smithii*). However, many of the uplands are heavily invaded by Kentucky blue grass (*Poa pratensis*) and smooth brome grass (*Bromus inermis*). Common forbs include lead plant (*Amorpha canescens*), common milkweed (*Asclepias syriaca*), and Canada anemone (*Anemone canadensis*). Wetland vegetation is almost entirely dominated by cattail (*Typha* spp.), particularly the hybrid cattail (*Typha x glauca*) which can dominate wetlands and form monotypic stands (Mitchell et al. 2011). Chase lake Wetland Management District was established in 1993, and was historically managed primarily for waterfowl production by establishing dense stands of vegetation achieved

by practicing idle management. Currently, portions of the refuge are grazed by cattle to try and reduce plant biomass build-up and promote biodiversity.

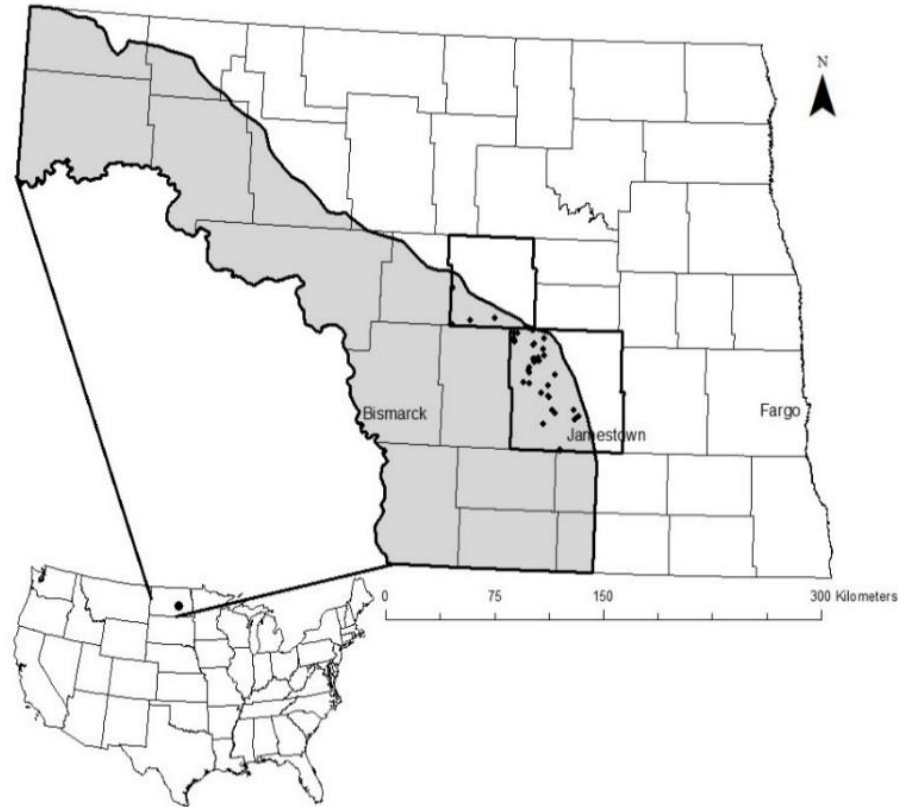


Figure 1.1. Research sites within Chase Lake Wetland Management District. Data was collected in 2016-2017 from 63 different wetlands (points) on public lands managed by the Chase Lake Wetland Management District within the Missouri Coteau (shaded area) region of North Dakota.

Site Selection

We used the National Wetland Inventory (NWI; USFWS 2016), which categorizes wetlands into systems, subsystems, and classes based on hydrology and vegetative characteristics (Cowardin et al. 1979) to select our wetlands. We standardized our wetland selection to sites that are classified as aquatic bed, emergent, and unconsolidated bottom in the Palustrine system within the NWI database. Wetlands within these classes fit one or more of the following habitat criteria required by our focal species: 1) shallow water (less than 1m deep), 2) surrounded by

little to no trees, and 3) the presence of emergent vegetation (Harms and Dinsmore 2012). Additionally, we only included seasonally flooded (C) and semi-permanently flooded (F) wetlands and excluded wetlands that were classified as temporarily flooded (A) according to the Cowardin System of Wetland Classification (Cowardin et al. 1979). Temporarily flooded wetlands were excluded because our focal species require water throughout the breeding season (Conway 2009). We stratified wetlands into five categories based on area (< 1 ha, 1-5 ha, 5-10 ha, 10-20 ha, and > 20 ha) to reduce biasing our dataset to small wetlands that are disproportionately abundant within this landscape (Harms and Dinsmore 2012). We randomly selected 15 wetlands from each size class using Sampling Design Tool for ArcGIS (ver. 10.3; ESRI 2015), but because very few wetlands existed that were > 20 ha and fit our selection criteria, we sampled all wetlands within in this size class. We then randomly generated one survey point for all wetlands less than 10 ha in size, two survey points for wetlands in the 10 - 20 ha size class, and three points for each wetland in the > 20 ha size class (Harms and Dinsmore 2012).

Bird Surveys

We sampled from May 15 - July 15 in 2016 and 2017. We completed three sampling rounds each year during the expected peak of the marshbird breeding season and all sampling rounds were approximately 20 days long (Round 1: May 15-June 4, Round 2: June 5-June 24, Round 3: June 25-July 15). This framework increased the likelihood of detecting inconspicuous and infrequently vocalizing individuals, and those species that coexist in the same area but have different reproductive timing (Conway 2011). During each survey, we conducted 200 m point counts using call-broadcasts to elicit a response and increase bird detection. We followed the standardized North American Marshbird Monitoring Protocol (Conway 2009) and were provided

calls for nine focal species that included American bittern (*Botaurus lentiginosus*), least bittern (*Ixobrychus exilis*), king rail (*Rallus elegans*), sora (*Porzana carolina*), Virginia rail (*Rallus limicola*), yellow rail (*Coturnicops noveboracensis*), pied-billed grebe (*Podilymbus podiceps*), red-necked grebe (*Podiceps grisegena*) and American coot (*Fulica americana*). We projected calls with an MP3 player (JLab Audio Eclipse Fit Clip 4GB, Jlab Audio, Oceanside, CA, USA) attached to an amplified speaker (Braven Model BRV-1s, BRAVEN LC, Oak Canyon Irvine, CA, USA), which was placed 0.5 m above the substrate and broadcasted calls at 90 dB toward the center of the wetland (Conway 2009). The North American Marshbird Monitoring Program dictated that we have a five minute passive listening period, during which all avian species present were recorded, followed by nine minutes of call-broadcast vocalizations (Conway 2009). Each minute of the sequence corresponded to one species, and consisted of 30 seconds of vocalizations and 30 seconds of silence with calls ordered by species dominance to increase callbacks of subdominant species early in the repertoire (Conway 2009). We limited the maximum detection distance to 200 m to reduce the potential of double counting and birds were recorded by both visual and aural cues and each individual detection was given a radial distance estimate. Because distance sampling assumes that distances are unbiased (Buckland et al 2001), we only recorded the distance upon first detection of an individual, regardless of any successive detections. Prior to conducting the surveys, wind speed, temperature, and cloud cover were recorded. Surveys were not completed if wind speed exceeded 20 km/hr or in heavy rain or fog (Conway 2009).

Vegetation Measurements

We conducted vegetation surveys at each point count location and four points 25 m away in each cardinal direction during the middle of each breeding season (~3rd week of June). At

each sampling point, we measured water depth (cm) and vegetation height (cm) using a tape measure and summarized the vegetation community using a 1.0 m² quadrats which have been shown to best quantify wetland vegetation in the PPR of North Dakota (DeKeyser et al. 2003) and categorized them based on modified Daubenmire cover classes, 0-5, 6-25, 26-50, 51-75, 76-95, and 96-100 (Daubenmire 1959). We visually estimated percent coverage of the major types of emergent vegetation which included cattail (*Typha* spp.), sedge (*Carex* spp.), rushes (*Schoenoplectus* spp.), grasses, and other structural vegetation components such as open water, bare ground, and litter. Survey points that extended into open water allowed for no vegetation measurements and were thus recorded as open water.

To gather wetland level characteristics, we placed four points evenly distributed along the perimeter of each wetland and recorded the width of the emergent zone. We also measured water depth at five equally spaced points along a transect placed through the emergent zone to quantify the slope of the wetland basin morphology. Additionally, we categorized wetlands based on percent vegetation cover and water (Stewart and Kantrud 1971). We then used the Landscape Development Intensity index (LDI), in combination with the 2011 National Land Cover Dataset (NLCD 2011) in ArcGIS v10.3 (Environmental Systems Research Institute 2014) and ground-truthing during site visits, as a quantitative human disturbance gradient (Brown and Visas 2005). The LDI quantifies and weighs anthropogenic disturbance by assigning coefficients to land uses (Brown and Visas 2005). We included five land uses (Natural system, Natural open water, Row crops, Single family residential, and Highway) based on their relevance to land uses consistent of the PPR of North Dakota. Finally, similar to Gilbert et al. (2006) and Mita et al. (2007), in an attempt to assess the surround landscape further, we measured the total wetland area within 1

km, total perennial cover within 1 km, the distance to the nearest wetland, and recorded the land use of the adjacent uplands.

Data Analysis

We calculated density estimates for species with > 60 detections (Buckland et al 2001) using package “Unmarked” within the R statistical environment (R Development Core Team 2016). Species that met this criterion were Virginia rail, sora, pied-billed grebe, American coot, and American bittern. Within “Unmarked” we used the “gdistsamp” function which fits models of animal abundance to data collected following Distance sampling protocols and accounts for imperfect detection probabilities (Fiske and Chandler 2011). This package expands on the sampling model of Royle et al. (2004), which effectively relaxes the assumption that individuals at a distance of 0 are assumed to be detected with certainty (R Development Core team 2016). The package “Unmarked” requires the user to set distance bins, therefore, we assigned raw distance bins to all species we included in our analysis. Virginia Rail, sora, and American coot received distance bins of 0 – 50 m, 50 – 100 m, 100 – 150 m, 150 – 200 m, American bittern received distance bins of 0 – 50 m, 50 – 100 m, 100 – 150 m, and we assigned distance bins of 0 – 70 m, 70 – 140 m, and 140 – 190 m to pied-billed grebe. Because we followed strict weather protocols set forth by the North American Marshbird Monitoring Program, we did not include any weather covariates on detection probability in models. Additionally, observer was not included as a covariate because there was only one observer (JTO), and that individual was trained in bird identification and distance estimation prior to each field season. We assumed that detection of birds did not vary by year because we surveyed the same wetland type both years, and because the length our survey season accounted for seasonal variation on detectability, we pooled data from each year for analysis (Harms and Dismore 2012). The density estimates given

come from an integrated likelihood model and are assessed using parametric bootstrapping procedures (Royle et al. 2004).

We used a hierarchical modeling scheme starting at the broadest, landscape scale and ending at the sampling point. We chose this method of organization because it likely reflects selection patterns used by migrating birds when returning to breeding grounds after migration (Brown and Dinsmore 1986, Lor and Malecki 2006, Hovick et al. 2012). At each step, we ranked the models using Akaike Information Criterion adjusted for small sample sizes (AIC_c) and used the model with the lowest AIC_c score from each step as a base model in the following step (Hovick et al. 2012). We considered models with a $\Delta AIC_c \leq 2$ to have strong support (Burnham and Anderson 2002). We tested for correlations amongst covariates by constructing a correlation matrix. If two variables were highly correlated ($r \geq 0.80$), the variable that made the least biological sense in the model was removed (Lor and Malecki 2006). Only two variables, amount of surrounding cropland and LDI, were highly correlated, therefore, we removed cropland from our models. For each species, we first compared the available key functions half-normal, hazard rate, exponential, and uniform, which describes the shape of detection function based on observed distances. Next, we developed univariate models examining broad landscape effects, followed by incorporating measurements collected at the wetland scale, and finishing by including fine-scale local vegetative characteristics (Table 1.1.).

Results

We detected 734 individuals representing seven species of secretive marshbirds from 2016 and 2017. Due to limited detections of two species (least bittern and red-necked grebe), we focused our analyses on five species with > 60 detections each. We detected slightly more secretive marshbirds in 2017 ($n = 417$) than in 2016 ($n = 327$). Virginia rail was the most

abundant species and accounted for over a quarter of our total detections ($n = 190$), while least bittern was detected the fewest times ($n = 3$) (Table 1.2.). Relative abundances of secretive marshbirds generally increased with wetland size class and ranged from 4.4 birds^{-point} for wetlands <1 ha to 11.2 birds^{-point} for wetlands >20 ha (Table 1.2.).

Individual species density estimates

Individual species densities estimates ranged from 0.67 individuals^{-point} for American bittern (SE = 0.24) to 2.54 individuals^{-point} for Virginia rail (SE = 0.53) (Table 1.3.). Modeling results indicated that factors across multiple scales influenced secretive marshbird densities. Three of the five best models explaining species' densities included all three scales we examined, while four of the five best models included at least two scales (Table 1.4.). The only species' density that could not be explained by multiple scales was Virginia rail, which had the null model as the best model for each of three scales examined. Of the variables we included in models, distance to the nearest wetland was influential at the landscape scale for two species, basin morphology was influential for two species at the wetland scale, and there were multiple vegetation characteristics that were important in explaining species' densities at the point scale (Table 1.4.).

Table 1.1. Descriptions and summary statistics for variables used in models. Statistics and descriptions of landscape, wetland, and point level variables used in models to examine environmental factors on secretive marshbirds at Chase Lake Management District, ND, USA 2016-2017.

Level	Variable	Mean (SE)	Range	Variable description
<i>Landscape-level</i>				
	TREATMENT			Management practice utilized in the surrounded uplands
	OPENWATER%	8.37 (6.23)	0-37.52	Percentage of open water in a 1 km buffer
	PERENNIAL%	35.82 (27.81)	1.0-82.46	Percentage of perennial cover in a 1 km buffer
	DISTANCETOWET	78.27 (44.67)	5.62-219.56	Distance to the nearest wetland (m)
	LDI	2.92 (0.66)	1.90-4.22	Landscape Development Index score based on surrounding land use
	TOTWET	53.23 (18.63)	14.24-150.58	Total area of all wetland habitat in a 1 km buffer
<i>Wetland-level</i>				
	CC	2.71 (0.54)	1-3	Wetland classification based on percent emergent vegetation and water
	SIZE	3.04 (1.29)	1-5	Wetland classification based on size
	TRANSITION	18.42 (6.23)	3.68-35.64	The slope of the wetland basin
<i>Point-level</i>				
	CATTAIL%	36.17 (16.52)	0-86.0	Cattail cover averaged across 3 1.0-m ² quadrats
	SEDGE%	0.01 (0.07)	0-1.0	Sedge cover averaged across 3 1.0-m ² quadrats
	RUSH%	2.89 (6.27)	0-47.33	Rush cover averaged across 3 1.0-m ² quadrats
	WATER%	13.99 (19.59)	0-70.67	Water cover averaged across 3 1.0-m ² quadrats
	LITTER%	36.75 (20.93)	0-78.33	Litter cover averaged across 3 1.0-m ² quadrats
	BG%	4.49 (9.27)	0-63.0	Bare ground cover averaged across 3 1.0-m ² quadrats
	GRASS%	2.91 (5.63)	0-35.00	Grass cover averaged across 3 1.0-m ² quadrats
	WD	4.40 (6.29)	0-34.33	Water depth averaged across 3 1.0-m ² quadrats
	VEGHEIGHT	171.42 (21.83)	89-203.67	Tallest piece of vegetation averaged across 3 1.0-m ² quadrats

Table 1.2. Total detections and relative abundance by survey point for seven secretive marshbird species. Total detections of secretive marshbirds, and relative abundance per survey point by size class in North Dakota, 2016 and 2017.

Species	Wetland Size (ha)					Totals
	<1	>1-5	>5-10	>10-20	>20	
	n=15	n=15	n=15	n=30	n=9	84
American bittern	2	12	21	33	8	76
American coot	10	20	28	51	16	125
Least bittern	1	1	0	1	0	3
Pied-billed grebe	8	26	38	54	20	146
Red-necked grebe	0	4	1	7	14	26
Sora	21	29	33	72	16	171
Virginia rail	22	41	32	70	27	192
Individuals per point	4.4	8.9	10.2	9.6	11.2	

Table 1.3. Secretive marshbird density estimates. Density estimates (standard error) from point count distance sampling surveys conducted from 2016-2017 within public lands managed by the Chase Lake Wetland Management District. Density estimates are presented for five species with met the 60 detection threshold. Scaled to individuals per ha and km² for comparison.

Species	birds ^{-point}	birds ^{-ha}	birds ^{-km2}
Sora	1.60 (0.35)	0.13	12.74
Virginia rail	2.54 (0.53)	0.20	20.22
American bittern	0.67 (0.45)	0.05	5.33
American coot	1.03 (0.74)	0.08	8.20
Pied-billed grebe	0.70 (0.84)	0.06	5.57

Table 1.4. Density model outputs for secretive marshbirds. Model outputs for five species of secretive marshbirds meeting the minimum detection threshold for density estimation. Models explaining the effects of variables on secretive marshbirds (see table 1.1 for variable description).

Species	Model	ΔAIC_c^a	K^b	w_i^c
Sora	Step 1			
	(OPENWATER%)	0	2	0.91
	(LDI)	5.88	2	0.00
	Step 2			
	(OPENWATER% + TRANSITION)	0	3	0.36
	(OPENWATER% + SIZE)	0.12	3	0.33
	Step 3			
(OPENWATER% + TRANSITION + WATER%) ^d	0	4	0.99	
(OPENWATER% + TRANSITION + WD)	9.73	4	0.00	
Virginia rail	Step1			
	(Null)	0	1	1.00
	(OPENWATER%)	26.3	2	0.00
	Step 2			
	(Null)	0	1	1.00
	(TRANSITION)	31.79	2	0.00
	Step 3			
(Null) ^e	0	1	1.00	
(WD)	15.71	2	0.00	
American bittern	Step 1			
	(Null)	0.00	1	0.34
	(LDI)	0.67	2	0.24
	Step 2			
	(SIZE)	0	2	0.34
	(Null)	0.67	1	0.24
	Step 3			
(SIZE + WD) ^f	0	3	0.29	
(SIZE + LITTER%)	0.62	3	0.21	
Pied-billed grebe	Step 1			
	(DISTANCETOWET)	0	2	0.82
	(LDI)	5.3	2	0.06
	Step 2			
	(DISTANCETOWET + CC)	0	3	0.94
	(DISTANCETOWET + TRANSITION)	5.69	3	0.05
	Step 3			
(DISTANCETOWET + CC + VEGHEIGHT) ^g	0	4	0.45	
(DISTANCETOWET + CC + BG%)	1.58	4	0.20	

Table 1.4. Density model outputs for secretive marshbirds (continued). Model outputs for five species of secretive marshbirds meeting the minimum detection threshold for density estimation. Models explaining the effects of variables on secretive marshbirds (see table 1.1 for variable description).

Species	Model	ΔAIC_c^a	K^b	w_i^c
American coot	Step 1			
	(DISTANCETOWET)	0	2	0.53
	(PERENNIAL%)	0.53	2	0.41
	Step 2			
	(DISTANCETOWET + TRANSITION)	0	3	0.72
	(DISTANCETOWET + CC)	1.92	3	0.27
	Step 3			
	(DISTANCETOWET + TRANSITION + CATTAIL%) ^h	0	4	0.57
(DISTANCETOWET + TRANSITION + VEGHEIGHT)	2.79	4	0.14	

^a Akaike's information criterion adjusted for small sample sizes, based on differences within each stage.

^b Number of parameters used in each model.

^c Model weight.

^d Best model has an AIC_c score of 1075.98.

^e Best model has an AIC_c score of 1195.81.

^f Best model has an AIC_c score of 490.37.

^g Best model has an AIC_c score of 818.63.

^h Best model has an AIC_c score of 815.01.

Discussion

Alteration and loss of natural ecosystems resulting from anthropogenic forces has resulted in declines in wetland biodiversity (Butchart et al. 2010). As a result, many wildlife species that depend on wetland ecosystems have lost the majority of their habitat and undergone precipitous population declines, resulting in conservation concern statutes for many wetland wildlife species (Conway 2009). To fill considerable knowledge gaps in secretive marshbird conservation, we examined the influence of variables affecting secretive marshbird abundance in the PPR of North Dakota and found that multiple scales were important when determining species abundances. Additionally, our data suggests that the PPR may be a significant stronghold

for secretive marshbird populations as our density estimates were as much as 14 times greater than published density estimates outside the PPR (Harms and Dinsmore 2012). Overall, our work highlights the importance of this region to secretive marshbird conservation.

Density estimates for four of the five species we examined were best explained by multiple scales. Similar to our findings, other studies have reported that multiple scales influence secretive marshbird abundance (Naugle et al. 1999, Mora et al. 2011, Glisson et al. 2015). We found that soras were significantly associated with variables at landscape, wetland, and point scales, while other studies have reported no influence of scale or that fine scale parameters alone are the most influential in explaining their abundance (Tozer et al. 2010, Glisson et al. 2015). Additionally, we found that pied-billed grebe abundances were associated with variables at all scales, a result well documented. Research in Missouri and Illinois found that pied-billed grebes were influenced by landscape variables and wetland variables (Darrah and Krementz 2010), while a study in South Dakota and another in New York found wetland variables and local vegetation components influenced pied-billed grebe occupancy (Naugle et al. 1999, Lor and Malecki 2006). Furthermore, an additional study in South Dakota found pied-billed grebes were influenced by local vegetation components (Naugle et al. 2001). Similar to our results, studies in Iowa and Ohio found American coots were influenced by multiple scales (Fairbairn and Dinsmore 2001, Mora et al. 2011) Finally, American bitterns were associated with variables at the wetland and point scale. However, because of their affinity to nest in surrounding upland grasses, we expected our results to show what other studies have suggested for American bittern association with surrounding landscape variables (Naugle et al. 2001).

Our results indicated that the distance to the nearest wetland was the most influential landscape variable in predicting secretive marshbird abundance. Several studies have found that

marshbirds prefer wetland complexes as opposed to isolated wetlands (Brown and Dinsmore 1986, Fairbairn and Dinsmore 2001, Smith and Chow-Fraser 2010). The PPR consists of wetland complexes, and has been labeled as one of the most important wetland regions in the world due to their productivity based on the number of wetlands and their surrounding vegetation (Mitsch and Gosselink 2007). While distance to the nearest wetland positively influence abundance of pied-billed grebes and American coots, sora density was negatively associated with the amount of surrounding open water within 1 km. Soras typically construct nests comprised of cattail, and inhabit wetlands dominated by emergent vegetation, with shallow, unbalanced water levels that produced a mosaic of emergent vegetation (Johnson and Dinsmore 1986, Gibbs and Melvin 1990). Therefore, we speculate that vegetation structure is more important than open water in wetland selection for soras as we observed soras most commonly at wetlands with dense vegetation and limited open water.

Many factors at the wetland scale were influential in explaining secretive marshbird densities. For instance, wetlands with a high ratio of open water to emergent vegetation had the greatest densities of pied-billed grebes. Previous research from New York suggest that pied-billed grebes build nests over deeper water with more open water than emergent vegetation (Lor and Malecki 2006). Additionally, because pied-billed grebes forage by diving for prey, they tend to be associated with larger wetlands with more open water (Murkin et al. 1997). Transition of the wetland basin was another wetland scale covariate that influenced both sora and American coot densities. Sora density peaked in wetlands that were characterized by a basin with a low degree of slope, which allows for interspersions of water along with tall, robust stands of emergent vegetation. This likely creates maneuverable habitat that allows this species to remain unseen while still providing adequate nesting and brood rearing structure (Gibbs and Melvin

1990). American coot density peaked in wetlands with a high degree of slope, associating American coots with deeper wetlands in our study, which is a result that has also been reported in Saskatchewan, Iowa, and South Dakota (Sugden 1979, Brown and Dinsmore 1986, Naugle et al. 2001). Deep wetlands tend to be larger (Cowardin et al. 1979), providing deepwater zones for American coots to dive and escape predators (Alisauskas and Arnold 1994). Additionally, American coots require large open water runways to get airborne (Alisauskas and Arnold 1994), making them more dependent on larger bodies of water. Finally, wetland size was a wetland scale variable that helped explain American bittern density. Similar to what Brown and Dinsmore (1986) found in Iowa and Naugle et al. (2001) found in South Dakota, our results suggest American bitterns to be area-dependent, and associated with larger wetlands.

We found that point scale variables influenced density estimates of four of our focal species. Similar to a study in New York, we found American bitterns were associated with shallow water depths (Lor and Malecki 2006). Similarly to what we found, research in New York found that pied-billed grebes are positively associated with vegetation height, opposed to sparse cover at the local point scale (Lor and Malecki 2006). At the point scale, American coots were negatively associated with percent of cattail coverage, this result further expands on American coots dependency of open water. Sora were associated with the percent water cover at the point scale, this supports what Johnson and Dinsmore (1986) found, in that sora breeding densities reached their peak in shallow sites where vigorous emergent vegetation occurs. Additionally, Walkinshaw (1940) and Glahn (1974) found sora nests often occur at changes in vegetation types and open water, supporting our finding of soras association with higher percent open water at the point scale.

Based on our density estimates, the PPR is a stronghold for secretive marshbird populations. Few studies have reported density estimates using distance sampling methodologies, but for those that have, estimates from our study are much greater. For example, our estimate for Virginia rail of 0.2 individuals^{-ha} was 14 times larger than estimates from portions of Iowa (Harms and Dinsmore 2012). Additionally, our estimate of 0.13 individuals^{-ha} for sora was 2.4 times larger, while our pied-billed grebe estimate of 0.06 individuals^{-ha} was 1.4 times larger than estimates published from Iowa (Harms and Dinsmore 2012). Iowa has lost nearly 90% of their wetland habitat and 99% of their grasslands since European Settlement (Dahl 1990, Smith 1998), creating a fragmented landscape which may explain some of these large discrepancies. Conversely, wetland densities can reach over 50 basins^{-km²} in the PPR of North Dakota (Dahl 2014). Within the PPR, wetlands comprised of small, shallow depressions and larger wetland basins surrounded by uplands can create a mosaic that is highly suitable for high densities of secretive marshbirds.

The number of spatial scales that influenced secretive marshbird abundance varied by species. This may mean that species-specific management is necessary in some cases. However, there are some broad generalization that can be drawn from our results. Overall, we suggest conserving wetland complexes, or managing areas with wetlands that include varying water depths and vegetation heights. By doing so, large, deep wetlands that American bittern, American coot, and the grebe species depend on are protected (Brown and Dinsmore 1986, Muller and Storer 1999, Darrah and Krementz 2010), while also preserving the small depressional wetlands that Virginia rail and soras use for breeding (Brown and Dinsmore 1986, Tozer et al. 2010, Glisson et al. 2015). Additionally, by concentrating our efforts on wetlands

complexes, we not only provide a broad range of suitable breeding habitat for secretive marshbirds, but also for a wide variety of wetland and grassland-dependent species.

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CHAPTER 2. DENSITY ESTIMATES OF MARSHBIRDS IN THE PRAIRIE POTHOLE REGION OF NORTH DAKOTA

Introduction

The continued loss of biodiversity has negatively affected ecosystem function and the goods and services they provide to society (Cardinale et al. 2012), and these losses continue throughout the world primarily as a result of anthropogenic forces (Butchart et al. 2010, Cardinale et al 2012). Few systems have experienced as much alteration and degradation as freshwater wetland ecosystems (Dudgeon 2010). As a result, biodiversity declines have been greater in wetland ecosystems than in most terrestrial ecosystems (Sala et al. 2000). Specifically, the anthropogenic drivers of wetland biodiversity loss include factors such as overexploitation, water pollution, flow modification, invasion of exotic species, and destruction or degradation of native communities (Dudgeon et al. 2006). In part, wetland losses have occurred because they are arguably the most vulnerable ecosystem to human induced land use change due to the disproportionate richness of freshwater ecosystems provide as habitat for native flora and fauna (Dudgeon et al. 2006). Therefore, because of these continued losses, policy and management require research that can inform the conservation and management of wetland dependent organisms and wetland systems.

Wetlands provide a wide diversity of goods and services to society, including water supply, raw material, food, recreation opportunities and cultural inspiration (Williams and Dodd 1978, Tiner 1984, Covich et al. 2004, Hansson et al. 2005). Land use and cover change has affected wetlands across the continent, reducing their ability to provide ecological services, while also limiting their capacity to sustain native biodiversity (Dudgeon et al. 2006). Due to an increasing human population, and demand for natural resources, aggressive farming practices

have led to draining and farming of many wetlands throughout North America (Wright and Wimberly 2012). While this trend has been common across all of North America, the northern Great Plains has experienced some of the most intensive drainage of wetlands (Dahl 1990, Dahl 2014).

The Prairie Pothole Region (PPR) of North America is a large expanse of wetland potholes, which were carved out by glacial movements during the Pleistocene Epoch. This area encompasses approximately 780,000 km² (Kantrud et al. 1989) and covers portions of Iowa, Minnesota, North and South Dakota, Manitoba, Alberta and Saskatchewan. The core of the PPR occurs in North Dakota, and this area is considered one of the most important wetland regions in the world (Dahl 2014). Historically, the PPR was comprised of an upland/wetland mosaic, however, it is currently estimated that 60 - 65% of the original wetlands within the PPR have been lost (Dahl 2014). Portions of the PPR have endured an extensive change in their landscape, with places like Iowa losing 90% of their wetland area, while approximately 500 km² of wetlands were lost in North Dakota, South Dakota and Minnesota between 1964 and 1968 almost entirely to agriculture development (Mitsch and Gosselink 2007). The PPR is known as the duck factory, producing approximately 50-75% of North American waterfowl annually (Klett et al. 1986). Therefore, while the importance of this area for sportsman and the hunting industry is obvious, the PPR also provides breeding and foraging opportunities for an additional 100 species of non-game birds (Greenwood et al. 1995, Naugle et al. 2001).

Wetland dependent birds, or marshbirds, are wetland-obligate species that generally breed in wetlands, or wetland like plant communities (Naugle et al. 1999). Since the 1970s, marshbird populations have been declining throughout North America, and several species are of conservation concern status at the local and regional levels (Conway 2009). Historically,

marshbirds of the PPR selected wetlands with native emergent vegetation, interspersed with water, but current effects of land cover and land use change on these species is unknown (Lor and Malecki 2006). Additionally, marshbirds are frequently under-sampled by large-scale monitoring programs such as the Breeding Bird Survey leading to a lack of information on population trends (Conway 2011). Therefore, there is a need to adequately survey marshbirds to fill the current knowledge gap. Few studies have utilized distance sampling as a metric for estimating these parameters in marshbird communities, furthermore, true density estimates are needed as a baseline for populations in the regions which will help inform future threatened and endangered species policies and conservation actions.

We investigated marshbird abundance in the PPR of North Dakota following distance sampling methodologies and using the North American Marshbird Sampling Protocol on public lands managed by the Chase Lake Wetland Management District. Our main objectives was to quantify abundance of marshbirds in the PPR of North Dakota, while assessing factors influencing marshbird abundance across multiple scales (i.e., 1 km landscape, wetland, and point). Based on previous research, we predicted that fine-scale, local measurements, such as vegetation height and cattail coverage would be important in determining species' abundances. Furthermore, we expected wetlands with a higher ratio of emergent vegetation to open water, and landscape variables, such as the amount of surrounding perennial cover, to be positively associated with all marshbird species.

Methods

Study Area

The Chase Lake Management District is located in Stutsman and Wells counties which are positioned in the Prairie Pothole Region (PPR) of North Dakota. This area is characterized by

rolling hills and high wetland densities of up to 50 potholes^{km²} (Dahl 2014). The Chase Lake Wetland Management District consists of approximately 17,400 ha, with roughly 5,260 ha in native prairie and over 4,600 ha of temporary, seasonal, semi-permanent and permanent wetlands. The predominant upland vegetation is a mixture of short and mid-grasses with little bluestem (*Schizachyrium scoparium*), blue grama (*Bouteloua gracilis*), and western wheatgrass (*Pascopyrum smithii*) comprising the majority of native grasses. Additionally, the uplands are heavily invaded by Kentucky blue grass (*Poa pratensis*) and smooth brome (*Bromus inermis*). Common forbs include lead plant (*Amorpha canescens*), common milkweed (*Asclepias syriaca*), and Canada anemone (*Anemone canadensis*). Wetland vegetation is dominated by cattail (*Typha* spp.), particularly the hybrid cattail (*Typha x glauca*) which often forms monotypic stands after establishment (Mitchell et al. 2011). Chase lake Wetland Management District was established in 1993, and was historically managed for waterfowl production which focused on the creation dense nesting cover for waterfowl. Contemporary management has tried to re-introduce cattle herbivory and fire to promote biodiversity and reduce invasive plant species that took over during the decades of idle management. The region has a temperate climate with cold winters and warm, dry summers. Historical average temperatures for May, June, and July are 12.8, 18.1, and 21.2 ° C and rainfall totals average 6.7, 8.7, and 8.3 cm for May, June and July, respectively (NDAWN 2017).

Site Selection

We used the National Wetland Inventory (NWI; USFWS 2016), which categorizes wetlands into systems, subsystems, and classes based on hydrology and vegetative characteristics (Cowardin et al. 1979), to select our wetlands. We standardized our wetland selection to sites that are classified as Aquatic Bed, Emergent, and Unconsolidated Bottom in the Palustrine

system within the NWI database. Wetlands within these classes fit one or more of the following habitat criteria required by our focal species: 1) shallow water (less than 1m deep), 2) surrounded by little to no trees, and 3) the presence of emergent vegetation (Harms and Dinsmore 2012). Additionally, we only included seasonally flooded (C) and semi-permanently flooded (F) wetlands, and excluded wetlands that were classified as temporarily flooded (A) according to the Cowardin System of Wetland Classification (Cowardin 1979). We stratified wetlands into five categories based on area (< 1 ha, 1-5 ha, 5-10 ha, 10-20 ha, and > 20 ha) to reduce biasing our dataset to small wetlands that are disproportionately abundant within this landscape (Harms and Dinsmore 2012). We randomly selected 15 wetlands from each size class using Sampling Design Tool for ArcGIS (ver. 10.3; ESRI 2015), but because very few wetlands existed that were > 20 ha and fit our selection criteria, we sampled all wetlands within in this size class. After wetland selection from each size class, we randomly generated one survey point for all wetlands < 10 ha, two survey points for wetlands in the 10 - 20 ha size class, and three points for each wetland > 20 ha (Harms and Dinsmore 2012).

Bird Surveys

We conducted 5-minute point counts for all marshbirds from May 15—July 15 in 2016 and 2017 using a 200 m detection radius. We followed standardized protocols developed by the North American Marshbird Monitoring Program designed to survey secretive marshbirds (Conway 2009). To ensure we covered the entire breeding season, we surveyed all wetlands three times each year. Survey rounds were approximately 20 days apart (Round 1: May 15-June 4, Round 2: June 5-June 24, Round 3: June 25-July 15), which should include the large range of reproductive periods of marshbirds. Each survey was performed either during the early morning (one-half hour before sunrise to three hours after), or late evening (two hours before sunset until

one-half hour after) survey period. We did not sample if wind speed exceeded 20 km/hr or in heavy rain or fog (Conway 2009). Because half of our passerine detections (i.e., 49.4%) occurred during our evening sampling period, we assumed there was no difference in detection probability between survey periods and used both in our analysis. Birds were recorded based on visual and aural cues, and to avoid double counting birds, we limited the maximum detection distance to 200 m. For each individual detected we determined a radial distance estimate, and because distance sampling assumes that distances are unbiased (Buckland et al 2001), we only recorded the distance upon first detection of an individual, regardless of any successive detections. Prior to conducting the surveys, wind speed, temperature, and cloud cover were recorded.

Vegetation Measurements

We conducted vegetation surveys during middle of the each breeding season (~3rd week of June). At each avian sampling point, and four points 25 m away in each cardinal direction, we recorded water depth (cm) and vegetation height (cm) using a tape measure. Additionally, we used a 1.0 m² quadrat based on the findings of DeKeyser et al. (2003) in the PPR of North Dakota, and modified Daubenmire cover classes, 0-5, 6-25, 26-50, 51-75, 76-95, and 96-100 (Daubenmire 1959) to summarize the vegetation community. We visually assessed percent coverage of the dominate emergent vegetation which included cattail (*Typha* spp.), sedge (*Carus* spp.), rushes (*Schoenoplectus* spp.), grasses, and other structural vegetation components such as open water, bare ground, and litter. Survey points that extended into open water allowed for no vegetation measurements and were thus recorded as open water.

In order to obtain wetland level characteristics, four points were evenly distributed along the perimeter of each wetland, where we recorded the width of the emergent zone. Additionally, we placed a transect through the emergent zone and measured water depth at five equally spaced

points along that line transect to quantify the basin morphology and slope. Furthermore, wetlands were categorized based on percent cover of water and vegetation (Stewart and Kantrud 1971). To assess the surrounding landscape within 1 km of each wetland, we used five land uses that were characteristic of land uses within the PPR of North Dakota within the Landscape Development Intensity Index (LDI) as a quantitative human disturbance gradient (Brown and Visas 2005), along with the 2011 National Land Cover Dataset (NLCD 2011) in ArcGIS v10.3 (Environmental Systems Research Institute 2014) and were also ground-truthed during site visits. The LDI quantifies and weighs anthropogenic disturbance by assigning scores to land uses, the land uses we included were Natural systems, Natural open water, Row crops, Single family residential, and Highway (Brown and Vivas 2005). In order to further assess the surrounding landscape, similar to Gilbert et al. (2006) and Mita et al. (2007), we measured the distance to the nearest wetland, total perennial cover within one km, the total wetland area within a one km buffer, and documented the land use of the adjacent uplands.

Data Analysis

We calculated density estimates for species with > 60 detections, and were of conservation interest (Buckland et al 2001) using package “Unmarked” within the R statistical environment (R Development Core Team 2016). Species that met this criterion were marsh wren (*Cistothorus palustris*), common yellowthroat (*Geothlypis trichas*), sedge wren (*Cistothorus platensis*), song sparrow (*Melospiza melodia*), yellow-headed blackbird (*Xanthocephalus xanthocephalus*), and Wilson’s snipe (*Gallinago delicata*). This methods provides true, rather than relative density by incorporating detection probabilities that account for imperfect detection (Fiske and Chandler 2011). These density estimates are assessed using parametric bootstrapping procedures, and are generated from an integrated likelihood model. Expanding on the sampling

model of Royle et al. (2004), “Unmarked” loosens the assumption that individuals at a distance of 0 are assumed to be detected with certainty (R Development Core Team 2016). Implementing distance bins is a requirement of “Unmarked”, therefore marsh wren, sedge wren, song sparrow, and yellow-headed blackbird all received distance bins of 0 – 50 m, 50 – 100 m, 100 – 150 m, 150 – 200 m, common yellowthroat received distance bins of 0 – 80 m, 80 – 120 m, 120 – 200 m, and to Wilson’s snipe, we assigned distance bins of 0 – 70 m, 70 – 120 m, 120 – 170 m, 170 – 200 m. The protocols we followed and set forth by the North American Marshbird Monitoring Program included strict weather protocols, therefore, we did not included any weather covariates on detection probability in models. Observer was also not included as a covariate because there was only one observer (JTO), and that individual was trained in bird identification and distance estimation prior to each field season. We assumed that detection of birds did not vary by year because the length of our survey season accounted for seasonal variation on detectability, and we surveyed the same habitat type both years. Therefore, we pooled data from each year for analysis (Harms and Dinsmore 2012).

When building our models, we first incorporated variables at the broad, landscape scale, and ended with variables at the fine, point scale. We chose to use this hierarchical modeling scheme based on scale, because it likely mirrors selection of wetlands by migrating birds after migration (Brown and Dinsmore 1986, Lor and Malecki 2006, Hovick et al. 2012). After ranking the models using Akaike Information Criterion adjusted for small sample sizes (AIC_c), we selected the model with the lowest AIC_c score as a base model for the following scales (Hovick et al. 2012). We considered models with a $\Delta AIC_c \leq 2$ to have strong support (Burnham and Anderson 2002). We tested for correlation amongst covariates by constructing a correlation matrix. If two variables were highly correlated ($r \geq 0.80$), the variable that made the least

biological sense in the model was removed (Lor and Malecki 2006). One pair of variables were highly correlated, LDI and the amount of surrounding croplands, therefore we removed cropland cover within 1 km of wetlands from our models. For each individual species, we compared the available key functions half-normal, hazard rate, exponential, and uniform for modeling detection curves. Broad landscape scale effects were used to develop univariate models, we then incorporated measurements collected at the wetland scale, and finished by adding fine-scale local variables, see Table 2.1. for variables by scale.

Results

We detected 5,012 individuals consisting of 44 species of wetland birds. We conducted analysis on six species that made up nearly 60% of our detections, and were of conservation interest. Of the species examined, Wilson' snipe had the fewest detections (n=60), while marsh wren was the most abundant (n =979) (Table 2). Marshbird relative abundances were the greatest in the 5-10 ha wetland size class and ranged from 25.9 birds^{-point} for wetlands <1 ha to 40.1 birds^{-point} for wetlands 5-10 ha (Table 2.2.).

Individual species density estimates

Species density estimates ranged from 0.53 individuals^{-point} for Wilson's snipe (SE = 0.12) to 14.85 individuals^{-point} for marsh wren (SE = 1.06) (Table 2.3.). Modeling results indicated that factors across multiple scales influenced one of the six species' densities, while the null model was best for the remaining five. The best model explaining Wilson's snipe density increased and were positively associated when larger amounts of perennial cover were within 1 km, when the transition of the wetland basin was high, and when large amounts of bare ground was present at the point scale (Table 2.4.).

Table 2.1. Descriptions and summary statistics for variables used in model selection. Statistics and descriptions of landscape, wetland, and point level variables used in models to examine environmental factors on marshbirds at Chase Lake Management District, ND, USA 2016-2017.

Level	Variable	Mean (SE)	Range	Variable description
<i>Landscape-level</i>				
	TREATMENT			Management practice utilized in the surrounded uplands
	OPENWATER%	8.37 (6.23)	0-37.52	Percentage of open water in a 1 km buffer
	PERENNIAL%	35.82 (27.81)	1.0-82.46	Percentage of perennial cover in a 1 km buffer
	DISTANCETOWET	78.27 (44.67)	5.62-219.56	Distance to the nearest wetland (m)
	LDI	2.92 (0.66)	1.90-4.22	Landscape Development Index score based on surrounding land use
	TOTWET	53.23 (18.63)	14.24-150.58	Total area of all wetland habitat in a 1 km buffer
<i>Wetland-level</i>				
	CC	2.71 (0.54)	1-3	Wetland classification based on percent emergent vegetation and water
	SIZE	3.04 (1.29)	1-5	Wetland classification based on size
	TRANSITION	18.42 (6.23)	3.68-35.64	The slope of the wetland basin
<i>Point-level</i>				
	CATTAIL%	36.17 (16.52)	0-86.0	Cattail cover averaged across 3 1.0-m ² quadrats
	SEDGE%	0.01 (0.07)	0-1.0	Sedge cover averaged across 3 1.0-m ² quadrats
	RUSH%	2.89 (6.27)	0-47.33	Rush cover averaged across 3 1.0-m ² quadrats
	WATER%	13.99 (19.59)	0-70.67	Water cover averaged across 3 1.0-m ² quadrats
	LITTER%	36.75 (20.93)	0-78.33	Litter cover averaged across 3 1.0-m ² quadrats
	BG%	4.49 (9.27)	0-63.0	Bare ground cover averaged across 3 1.0-m ² quadrats
	GRASS%	2.91 (5.63)	0-35.00	Grass cover averaged across 3 1.0-m ² quadrats
	WD	4.40 (6.29)	0-34.33	Water depth averaged across 3 1.0-m ² quadrats
	VEGHEIGHT	171.42 (21.83)	89-203.67	Tallest piece of vegetation averaged across 3 1.0-m ² quadrats

Table 2.2. Total detections and relative abundance by survey point for six species of marshbird. Total detections of marshbirds, and relative abundance per survey point by size class in North Dakota, 2016 and 2017.

Species	Wetland Size (ha)					Totals
	<1 n=15	>1-5 n=15	>5-10 n=15	>10-20 n=30	>20-30 n=9	
Common Yellowthroat	83	114	125	237	73	632
Marsh wren	125	163	188	365	138	979
Sedge wren	14	19	23	33	15	104
Song sparrow	70	116	142	220	56	604
Wilson's snipe	9	23	8	16	4	60
Yellow-headed blackbird	87	89	115	214	49	554
Individuals per point	25.9	34.9	40.1	36.2	37.2	

Table 2.3. Marshbird density estimates. Density estimates (SE) obtained from point count distance sampling surveys conducted within public lands managed by the Chase Lake Wetland Management District in 2016-2017. Density estimates are presented for five species with met the 60 detection threshold. Scaled to individuals per ha and km² for comparison.

Species	birds/point	birds/ha	birds/km ²
Common yellowthroat	9.39 (0.82)	0.75	74.76
Marsh wren	14.85 (1.06)	1.18	118.23
Sedge wren	2.22 (0.42)	0.18	17.68
Song sparrow	14.37 (1.72)	1.14	114.41
Wilson's snipe	0.53 (0.12)	0.04	4.22
Yellow-headed blackbird	7.41 (0.76)	0.59	59.00

Table 2.4. Density model outputs for six species of marshbirds. Model outputs for six species of marshbirds meeting the minimum detection threshold for density estimation. Models explaining the effects of variables on marshbirds (see table 2.1 for variable description).

Species	Model	ΔAIC_c^a	K^b	w_i^c
Marsh Wren	Step 1			
	(Null)	0	1	1.00
	(TREATMENT)	68.96	5	0.00
	Step 2			
	(Null)	0	1	1.00
	(TRANSITION)	53.0	2	0.00
	Step 3			
	(Null) ^d	0	1	1.00
	(WATER%)	64.34	2	0.00
Common Yellowthroat	Step 1			
	(Null)	0	1	1.00
	(OPENWATER%)	53.41	2	0.00
	Step 2			
	(Null)	0	1	1.00
	(SIZE)	51.32	2	0.00
	Step 3			
	(Null) ^e	0	1	1.00
	(GRASS%)	51.23	2	0.00
Sedge Wren	Step 1			
	(Null)	0.00	1	0.88
	(TREATMENT)	3.41	2	0.15
	Step 2			
	(Null)	0	1	0.99
	(SIZE)	11.92	2	0.00
	Step 3			
	(Null) ^f	0	1	0.87
	(WATER%)	5.47	2	0.06
Song Sparrow	Step 1			
	(Null)	0	1	1.00
	(TREATMENT)	123.92	5	0.00
	Step 2			
	(Null)	0	1	1.00
	(SIZE)	124.6	2	0.00
	Step 3			
	(Null) ^g	0	1	1.00
	(BG%)	118.3	2	0.00

Table 2.4. Density model outputs for six species of marshbirds (continued). Model outputs for six species of marshbirds meeting the minimum detection threshold for density estimation. Models explaining the effects of variables on marshbirds (see table 2.1 for variable description).

Species	Model	ΔAIC_c^a	K^b	w_i^c
Yellow-headed blackbird	Step 1			
	(Null)	0	2	1.00
	(LDI)	21.52	2	0.00
	Step 2			
	(Null)	0	1	1.00
	(TRANSITION)	59.97	2	0.00
	Step 3			
	(Null) ^h	0	1	1.00
	(VEGHEIGHT)	51.91	2	0.00
Wilson's snipe	Step 1			
	(PERENNIAL%)	0	2	0.49
	(LDI)	1.01	2	0.30
	Step 2			
	(PERENNIAL% + TRANSITION)	0	3	0.74
	(PERENNIAL% + SIZE)	3.72	3	0.16
	Step 3			
	(PERENNIAL% + TRANSITION + BG%) ⁱ	0	4	0.25
	(PERENNIAL% + TRANSITION + LITTER%)	1.06	4	0.14

^a Akaike's information criterion adjusted for small sample sizes, based on differences within each stage.

^b Number of parameters used in each model.

^c Model weight.

^d Best model has an AIC_c score of 2806.69.

^e Best model has an AIC_c score of 1970.53.

^f Best model has an AIC_c score of 726.22.

^g Best model has an AIC_c score of 2246.39.

ⁱ Best model has an AIC_c score of 504.03.

Discussion

Wetlands across the United States are one of the most heavily altered and degraded ecosystem types (Dudgeon 2010). These changes have led to declines in marshbird populations putting increasing pressure on existing wetlands to sustain remaining populations (State of the birds 2016). Although wetlands within the PPR have endured losses, it remains one of the most

intact wetland ecosystems in the United States and therefore should be a stronghold for marshbirds and other wetland-dependent wildlife. Our data supports this as our density estimates ranged from 4.22 Wilson' snipe^{-km²} to 118 marsh wrens^{-km²}. Furthermore, compared to studies in Mississippi and the Great Lakes Region, our density estimates were 8.8 times greater for common yellowthroat, 5.7 times greater for marsh wren, and only slightly greater for sedge wren (Twedt and Wilson 2017, Panci et al. 2017). We had hypothesized that surrounding landscape variables (i.e., agriculture fields and perennial cover) and wetland vegetation characteristics (i.e., cattail coverage and water depth) would generate differences in wetland use by marshbirds and influence their abundances. However, we found that the variables we measured generally did not influence marshbird abundance, or the variables were not able to have an effect because of the spatial scale at which we measured them. Conversely, the best model explaining Wilson's snipe density supported our hypothesis. Surrounding land use, wetland, and fine-scale vegetation factors were all predictive of Wilson's snipe density. By quantifying the density and assessing the influence of landscape and local vegetation components of marshbirds, our research provides managers some guidance for future conservation and provides policy makers with baseline estimates that can be important for future planning. Overall, our results emphasize how important this region is for marshbird conservation.

Density estimates for five of the six species we examined were not influenced by any of the covariates we included. Research in Iowa, Chesapeake Bay, and Ontario has shown that marshbirds are associated with landscape variables (Fairbairn and Dinsmore 2001, DeLuca et al. 2004, Smith and Chow-Fraser 2010), and local vegetation variables in Iowa and the Great Lakes region (Harms and Dinsmore 2015, Panci et al. 2017). However, we attribute our findings to several species being abundant at nearly all wetlands we examined. Moreover, wetlands in the

PPR of North Dakota can reach densities over 50 potholes^{km²} (Dahl 2014). As a result, there is limited variability in detections for five of our most abundant species. For example, common yellowthroat and song sparrow were present in 100% of the wetlands we surveyed, marsh wren was found in 92% of our sites, yellow-headed blackbird was present in 86%, and sedge wren was found in 62% of the wetlands we surveyed. However, Wilson's snipe were present at 42% of our sites, and as a result, models were informative at predicting abundance.

Wilson's snipe abundance was influenced by multiple scales. At the landscape level, our results indicate that Wilson's snipe density was positively associated by the amount of perennial cover within 1 km. This result follows the trend of previous studies which shows that Wilson's snipe are dependent on adjacent upland vegetation for breeding habitat, because they typically place nests in short grass or sedge-like grasses on moist but unflooded ground near wetlands (Tuck 1972, Johnson and Ryder 1977, McKibben and Hofmann 1985). At the wetland level, we found Wilson's snipe density was higher in wetlands with a basin that consisted of a steep slope, associating them with deeper wetlands. However, research in Colorado found that Wilson's snipe prefer to breed in shallow wetlands dominated by *Carex* spp., which allow for the presence of short and sparse vegetation (Johnson and Ryder 1977). Wetlands in the PPR of North Dakota, particularly shallow wetlands, are almost all entirely invaded by cattail, leaving little to no bare ground or open water. This likely leads to a lack of foraging ability in shallow wetlands of the PPR for Wilson's snipe. Therefore, are dependent on wetlands deeper than 1.5 feet where cattail cannot establish and create a monoculture that covers the entire basin. At the point scale, Wilson's snipe density was positively associated with the amount of bare ground. Wilson's snipe are often associated with bare ground for forage areas (Arnold 1981). Additionally, Wilson's snipe are in the family *Scolopacidae*, or shorebirds, therefore, they probe for insects with a long

bill, often in moist soil or mud (Arnold 1981). This result was unsurprising because Wilson's snipe are associated with bare ground, as it is essential for foraging.

To guide practical marshbird conservation, we need reliable density estimates and an understanding of how changing land uses affect them. We found no variables we measured to influence five of the six species, therefore further investigation should consider examining the influence of surrounding land use out to larger scales (i.e., 5-km) or by measuring more specific vegetation structure and composition at very fine scales. However, our study does provide some of the first density estimates that account for imperfect detection for marshbirds in the PPR of North Dakota. Furthermore, our density estimates unequivocally show evidence that marshbirds are extremely abundant in the PPR (Klett et al. 1986, Niemuth et al. 2006). Our estimates would suggest that the marshbirds we examined are not currently of conservation concern within this region. Finally, this research highlights the importance of the PPR for wetland-dependent, non-game bird species in addition to its known importance for North America's waterfowl populations.

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