Secretive Marshbird Response to Invasive Wetland Plant Management

in the Prairie Pothole Region of Minnesota

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DEDICATION

I dedicate this work to my parents, Major and Cathy, my brother Joe, and sister Mandy. Thank you, family, for making me who I am; I love you. And to my newer family Becky, Clayton, Logan, Darby, Wiley, Allison, Emmett, Isabelle, Oliver – let's continue to explore the world with curiosity and reverence.

ABSTRACT

Marshbirds are difficult to survey due their secretive nature and association with dense wetland vegetation. Recently developed standardized survey protocols are used to monitor patterns of abundance, primarily at large spatial scales, but also can be used to assess marshbird response to management. We estimated abundances of 5 species of marshbirds (American bittern [*Botaurus lentiginosus*], least bittern [*Ixobrychus exilis*], pied-billed grebe [*Podilymbus podiceps*], sora [*Porzana carolina*], and Virginia rail [*Rallus limicola*]) in relation to vegetation management techniques of Prairie Pothole wetlands. In northwestern Minnesota, management in autumn 2105 included herbicide application to wide-spread cattail (*Typha* spp.) mats with the goal to break up dense vegetation patches and restore wetlands to hemi-marsh conditions. In a before-after, control-impact study design we conducted standardized call-broadcast surveys for marshbirds during breeding seasons 2015 – 2018. We observed that American bittern, pied-billed grebe, sora, and Virginia rail abundances initially decreased, and then increased at 2nd and 3rd seasons post-treatment at sites where herbicides had been applied. In west-central Minnesota, long-term vegetation management included varying frequencies of multiple control methods. Using a habitat-informed detection probability we transformed bird counts to densities to compare abundances of marshbirds across survey locations surrounded by variable amounts of suitable habitat. We compared abundances of marshbirds among categories of wetlands with management histories of low frequency of prescribed fire, high frequency of prescribed fire, and high frequency of prescribed fire and grazing. Fire and grazing as applied in the system we studied did not appear to influence Prairie Pothole Region wetland characteristics enough to result in changes in marshbird abundance, but abundance of marshbirds was related to characteristics of individual wetlands that did not appear to respond to fire and grazing. Pied-billed grebe abundance was positively associated with higher areas of open water, soras were more abundant in wetlands with high ratios of open water to emergent vegetation, and Virginia rails were more abundant in wetlands with scrub-shrub wetland cover types.

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Overview

Large-scale standardized bird surveys in North America (e.g., Waterfowl Breeding Population and Habitat Survey, North American Breeding Bird Survey) provide limited information to assess abundance and population trends of marshbirds. Furthermore, the current standardized survey protocol for marshbirds is based on passive listening and call broadcasts at survey locations, often along the edges of wetlands where a portion of the area surrounding survey locations includes cover not used by marshbirds (i.e., marshbird habitat is not distributed uniformly around survey locations). Counts of marshbirds from locations surrounded by variable amounts of suitable habitat cannot be compared directly without accounting for those differences. We present an algorithm for deriving estimates of density that accounts for the distribution of suitable habitat around survey locations and for decreased detectability as a function of distance, allowing direct comparison of marshbird abundance across survey locations. We applied this approach to surveys for 4 species of marshbirds in the Prairie Pothole Region of western Minnesota, U.S.A, and for 3 of those 4 species, estimates of density provided different insight on abundance than did raw counts. We suggest this approach is not limited to marshbird surveys, as heterogeneity of suitable habitat surrounding survey locations is a challenge that occurs in other wildlife surveys.

Key words: abundance, avian point count, density, marshbirds, survey methods

INTRODUCTION

Marshbirds are notoriously difficult to survey (Seamans et al. 2011, Soulliere et al. 2012), in large part due to their secretive nature and difficulty of observing them in dense, emergent wetland vegetation. Conway (2011) developed a standardized marshbird survey protocol and Johnson et al. (2009) described a framework for monitoring marshbirds that facilitate assessing marshbird population changes across broad geographical regions (e.g., Bolenbaugh et al. 2011, Seamans et al. 2013, Pickens and King 2014). However, assessing marshbird abundance among survey locations is not straightforward because the distribution of marshbird habitat around survey locations can be highly variable. For example, the same count of individuals at 2 survey locations may not represent the same abundance when amount and distribution of marshbird habitat differs between those survey locations (Fig. 1.1). Comparing counts among survey locations, study sites, or regions is potentially inappropriate if the distribution of marshbird habitat around survey locations is dissimilar. Whereas density, as a ratio of number of birds to area of suitable habitat, can be compared across survey locations and study sites.

Repeated, standardized surveys of animals conducted at the same locations can provide information about changes in abundance (Gibbs and Melvin 1997, Tozer 2016). Such surveys can also produce information useful to assess abundance or occupancy in relation to habitat features (e.g., Glisson et al. 2015). However, how variation in the distribution of habitat around survey locations affects assessment of trends and habitat relations has not been thoroughly evaluated. In some contexts, surveys for marshbirds present an extreme example of the potential influence of unequal distribution of habitat around survey locations because survey locations are often at the edge of wetlands and the area surrounding survey locations may include a large portion of cover types that are not used by marshbirds (Fig. 1.1). Under such circumstances, it is necessary to account for the spatial configuration of habitat surrounding survey locations when drawing inference about factors that affect detection probability (e.g., modeling change in abundance of island scrub-jays [*Aphelocoma insularis*] confined to chaparral and forest cover types of Santa Cruz Island, California, U.S.A. [Sillet et al. 2012]). Auditory and visual surveys of marshbirds are further complicated by the fact that detection probability generally decreases as the distance between the observer and an individual marshbird increases, and such relationships are species-specific (Conway et al. 2008). To address these issues, we developed an approach to transform counts of marshbirds observed on surveys to a standardized measure of density that can be compared among survey locations and study areas.

Herein we describe a process to derive estimates of marshbird density that accounts for the distribution and amount of wetland surrounding survey locations that also incorporates detection probability as a function of distance between observer and a marshbird. Our motivation for developing this process was to be able to compare marshbird abundance in relation to management strategies used to control invasive vegetation (primarily cattails [*Typha* spp.]; see Chapter 2) in Prairie Pothole Region wetlands, and a realization that the same count at 2 survey locations did not necessarily represent the same abundance of marshbirds. We suggest this approach may be useful in other systems where the distribution of available habitat around survey locations is heterogeneous.

STUDY AREA

We conducted surveys for marshbirds in Prairie Pothole Region wetlands in west-central Minnesota, U.S.A. (Fig. 1.2) to assess marshbird response to management activities targeted at controlling invasive vegetation (primarily *Typha* spp.). The landscape consisted of gentle hills, scattered shallow lakes, and many small wetlands that are not hydrologically connected by surface water. We identified wetland survey locations within properties managed by U.S. Fish and Wildlife Service Morris Wetland Management District (Vacek and Friske 2012), stratified by management history, and grouped them together by proximity into survey routes.

MARSHBIRD SURVEYS

During the early portion of breeding seasons (May – June) of 2015 – 2016 observers conducted 2 surveys each at 128 locations distributed across 75 parcels (i.e., U.S. Fish and Wildlife Service Waterfowl Production Areas and easement properties). We followed standardized monitoring protocols (Conway 2011), except we conducted 2 (rather than 3; 1 in the 3.5-hour period around sunrise and 1 in the 3.5-hour period around sunset) visits per year to each survey location for a total of 467 surveys. Surveys targeted American bitterns (*Botaurus lentiginosus*; hereafter bitterns), pied-billed grebes (*Podilymbus podiceps*; hereafter grebes), soras (*Porzana carolina*), and Virginia rails (*Rallus limicola*). During surveys we broadcasted calls of the targeted species plus calls of yellow rails (*Coturnicops noveboracensis*) and least bitterns (*Ixobrychus exilis*), although we did not obtain sufficient detections for analyses for these 2 species. Observers recorded aural and visual detections of individuals of all targeted species, and estimated distance to bird locations using laser range finders and by relating features of the landscape to aerial imagery printed on maps with interval buffer circles around survey locations. The protocol for this study was approved by the University of Minnesota Institutional Animal Care and Use Committee (IACUC protocol #1503-32456A).

WETLAND COVER AT SURVEY LOCATIONS

We quantified the amount and configuration of wetland cover surrounding survey locations in ArcGIS (Desktop 10.3 Environmental Systems Research Institute, Inc., Redlands, CA, USA) using recently updated and publicly published geospatial data from the National Wetland Inventory (NWI; Minnesota Department of Natural Resources 2016; 91 survey locations). We supplemented available data by combining National Vegetation Classification Standard (NVCS; U.S. Fish and Wildlife Service 2008a) wetland cover on U.S. Fish and Wildlife Service properties and manually updated boundaries of older (2009) NWI data by interpreting recent years' aerial imagery and topographic maps (Minnesota Geospatial Office 2015 [2010, 2013, 2015; USGS topographic maps]; 40 survey locations). We measured area of wetland cover as the sum of types most associated with marshbird species: temporary, seasonal, and semi-permanent wetlands (NVCS analogs of 2, 3, and 4, respectively, of the Stewart and Kantrud [1971] classification system), and summed a subset of classes associated with deep marsh and open water, semi- and permanent wetlands (wetland types 4, 5) most associated with grebes.

ANALYTICAL APPROACH

We developed an algorithm to estimate density from marshbird counts, accounting for detection probability as a function of distance and the amount and distribution of wetland cover surrounding survey locations, for the purpose of comparing density across survey locations. Detection probability varied among marshbird species due to different behaviors and vocalizations. Therefore, we applied this process independently for bitterns, grebes, soras, and Virginia rails. Briefly, we calculated habitat-informed species-specific detection probabilities based on the assumption that detection probability decreased as a function of distance from observer and that incorporated the proportion of wetland cover surrounding survey locations by distance. We then applied the habitat-informed detection probabilities to adjust marshbird counts and divided the adjusted counts by area of wetland cover to derive estimates of marshbird density.

Determining Distance from Survey Locations over which to Estimate Density

The probability that an observer detects a marshbird decreases as a function of the distance between the observer and a marshbird and at some distance approaches zero. Marshbirds at long distances have low probability of detection, but as distance from the observer increases, the area surveyed increases exponentially (Fig. 1.3). When a few detections at great distances are used to represent density over a large area, resulting estimates are highly uncertain. It is therefore necessary to estimate a truncation distance, beyond which marshbird detections are ignored to minimize the influence of distant detections on estimates of density (Buckland et al. 2015). Several suggestions for deriving truncation distance have been proposed, including the minimum distance that contained 90% of observations (Buckland et al. 2001:151), twice the mean detection distance (Buckland et al. 2015:69), and the distance at which detection probability (fit to a half-normal probability density function) for the species equals 0.1 (Buckland et al. 2001:151). We derived estimates of marshbird density (see below) using all 3 of these methods of estimating truncation distance, and sets of density estimates were highly similar with one another; therefore, we chose the simplest of these approaches (distance that contained 90% of detections) to truncate data in our analyses.

Calculating Habitat-Informed Detection Probability

To model the relationship between detection probability and distance from observers to marshbirds, we pooled marshbird counts by species across survey locations for all visits over both years and then grouped counts in concentric distance bins from the location of the observer (i.e., survey locations). Observers used survey location maps that included buffers at 25-m intervals, and because observers tend to estimate distances at convenient units (e.g., 50 m, 100 m, 150 m), we grouped data into 23-m distance bins to minimize the effect of observers rounding to convenient distances (Buckland et al. 2001; Fig. 1.3). We summed the area (ha) of wetland cover within the same 23-m distance bins across survey locations. We then divided the summed (across survey locations) marshbird counts within distance bins, c_{raw}, by the summed wetland area within each 23-m distance bin, w, up to the truncation distance to scale the proportion of counts to wetland area, p_{raw}:

$$p_{raw} = \frac{c_{raw}}{w}$$

We then scaled p_{raw} values in all distance bins to p_{raw} of the first bin such that the habitat-informed detection probability in the first bin was 1 (i.e., all birds close to the observer were detected). This transformed the raw proportion of counts in each distance bin to represent a detectability function, where habitat-informed detection probability at distance 0 m equaled 1 and declined as distance increased.

Deriving Estimates of Density from Marshbird Counts

We applied the habitat-informed detection probability to transform marshbird counts into densities. Using the maximum of the 2 counts of marshbirds at each survey location in each year we first adjusted counts in each distance bin to account for detection probability, and then summed adjusted counts across all distance bins out to the truncation distance. We similarly summed the area of wetland across distance bins out to the truncation distance, and divided the sum of adjusted counts by the sum of wetland area for each survey location to derive an estimate of marshbird density for each marshbird species at each survey location for each year. For example, consider survey location 272H where we recorded 4 and 2 soras during visits in 2015. We grouped the maximum count per year for each year (n = 4) by their estimated distances, where 2 birds were at 46 m, 1 was at 120 m, and 1 was at 190 m. We applied the associated habitat-informed detection probability to counts within each distance bin; in the 46-m distance bin we divided 2 soras by 0.42 to result in 4.81 as the adjusted count, 1 sora by 0.07 for an adjusted count of 13.97, and 1 sora by 0.04 for an adjusted count of 27.56. We summed adjusted counts across distance bins (total adjusted count = 46.35 for survey location 272H) and divided by the amount of wetland area from the survey location out to the truncation distance of 184 m for soras (5.42 ha at survey location 272H), for the final estimated density of 8.55 birds per wetland hectare in 2015 at survey location 272H.

Comparing Marshbird Counts and Estimated Densities

We evaluated the effect of converting counts to estimates of density using correlation and simple linear regression. We assumed that unadjusted counts and derived estimates of density would be positively correlated, and if there was not an increase in information, the slope of the least-squares regression line relating counts to density would equal 1. In that case, unadjusted counts would be appropriate to compare marshbird abundances between and among survey locations without accounting for detection probability or the distribution of wetlands around survey locations. Slopes ≠ 1 would suggest that incorporating detection probability and accounting for the amount of wetland around survey locations to derive estimates of density was appropriate to compare marshbird abundances between and among survey locations.

RESULTS

Observers detected 1,019 marshbirds of our targeted species at survey locations across years (2015 and 2016; Appendix A). We detected soras most frequently and at the highest number of sites (35% of all detections, at 73% of sites), followed by American bitterns (18%, at 64% of sites). Habitat-informed detection probability (based on all detections, including those detected outside the 10-minute survey period at a survey location) for soras and Virginia rails were concentrated at distances close to zero and decreased to near zero at 100 m, indicating that we detected higher proportions of these species closer to observers than bitterns and grebes. Truncation distance varied among species (Fig. 1.3). Species-specific truncation distances (distance that captured 90% of detections) were 100 m for Virginia rails (distance bin 92 – 115 m), 200 m for soras (distance bin 184 – 207 m), 240 m for bitterns (distance bin 230 – 253 m), and 320 m for grebes (distance bin 322 – 345 m).

To estimate species-specific density, we applied habitat-informed detection probabilities to observations during the 10-minute survey period using the maximum count at each survey location. The mean estimated densities across survey locations in 2015 (n = 112) were 0.16 birds per wetland ha for bitterns (0.07 to 0.25; 95% confidence interval), 0.81 for grebes (0.08 to 1.52), 1.75 for soras (1.20 to 2.29), and 0.77 for Virginia rails (0.42 to 1.14). In 2016 (n = 127) densities were 0.33 for bitterns (0.20 to 0.48), 0.77 for grebes (0.33 to 1.22), 2.83 for soras (2.00 to 3.81), and 1.20 for Virginia rails (0.73 to 1.71; Table 1.1).

Maximum counts and estimates of density that incorporated detection probability and the distribution of wetlands around survey locations were positively correlated for bitterns (r =0.76, P < 0.001), grebes (r = 0.42, P < 0.001), soras (r = 0.57, P < 0.001), and Virginia rails (r = 0.61, P < 0.001; Fig. 1.4). The slope of the least-squares line relating maximum counts to estimates of density was approximately 1 for bitterns (estimated slope = 0.95 +/- 0.10; 95% CI), and > 1 for grebes (slope = 1.44, +/- 0.39), soras (slope = 2.18 +/- 0.40), and Virginia rails (slope = 2.04 +/- 0.34).

DISCUSSION

Marshbirds are closely tied to habitat comprising specific land-cover types that vary in abundance and distribution around survey locations. Failure to account for distribution of habitat around survey locations under these circumstances can confound comparisons of abundance among survey locations. To address this issue, we developed an algorithm that accounts for the amount and distribution of potential habitat around marshbird survey locations. For 3 of the 4 marshbird species we considered, abundance represented by counts differed from estimates of density that accounted for detection probability and the distribution of potential marshbird habitat surrounding survey locations (Table 1.1; Fig. 1.4). For grebes, soras and Virginia rails, slopes >1 for linear relationships between abundance represented by counts vs. estimates of density indicated that counts underestimated true abundance. Because wetland cover was not distributed similarly around survey locations (Figs. 1.1 and 1.3), raw counts did not appropriately represent the differences in marshbird abundances as they do not consider the amount and distribution of potential habitat surveyed at individual survey locations. Therefore, estimated density was a more appropriate measure of abundance in this context as it would likely be for other species and other settings with heterogeneous potential habitat surrounding survey locations (e.g., Zylstra et al. 2010, Isaac et al. 2011, Sillet et al. 2012).

Additionally, our results affirmed the need to adjust counts based on individual species' biology. Not all species have the same habitat requirements or select habitat at the same spatial scales. In density calculations for bitterns, soras and Virginia rails we used the area of the entire wetland basin, including wet meadow and temporary wetlands types, because these species use those cover types. However, compared to the other marshbird species in our study, grebes primarily use deep water wetland types with high proportion of open water for underwater foraging (Chapter 2). Therefore, we restricted measures of available habitat for grebes to those cover types, and derived estimates of density with habitat-informed detection probability that incorporated only deep- and open-water wetland cover types.

For bitterns, the linear relationships between abundance represented by counts vs. estimates of density had a slope close to 1, indicating that accounting for detection probability and the distribution of wetland cover around survey locations did not improve comparisons of abundance across survey locations; that is, counts represented abundance equally as well as density. Bitterns appeared to be influenced by the presence of an observer, in that detections close to survey locations were less likely than detections farther from survey locations. We accounted for this effect by scaling detection probability for bitterns to the second bin (i.e., detection probability in the second bin = 1.0).

We also note that our estimates of marshbird density could provide insight to land managers about regional population size of individual marshbird species. By applying estimates of density to measures of the species-specific area of available habitat across the entire study area, it would be possible to estimate population size during years when surveys are conducted. Finally, we suggest that ours or a similar approach can provide more appropriate insight into spatial variation in abundance for marshbirds and other species that have strong associations with particular cover types that are not similarly distributed around survey locations.

Table 1.1. Transforming raw counts to habitat-informed densities

Maximum counts of marshbirds, adjusted for the distribution of wetland cover around survey locations, and estimated density incorporating both detection probability and amount and distribution of wetland cover around survey locations derived from surveys in 2015 and 2016 at Prairie Pothole Region wetlands of west-central Minnesota, U.S.A.

		Maximum observed count		**Adjusted count**		**Density of birds per target wet ha**
Species	**Year**	**sum**	**mean (SD)**	**sum**	**mean (SD)**	**mean (SD)**
American bittern	2015	21	0.19 (0.44)	37.10	0.29 (0.78)	0.16 (0.49)
	2016	42	0.33 (0.12)	140.42	1.10 (3.89)	0.33 (0.78)
Pied-billed grebe	2015	41	0.37 (0.94)	130.03	1.02 (3.03)	0.81 (4.12)
	2016	55	0.43 (0.6)	554.29	4.33 (22.48)	0.77 (2.52)
Sora	2015	97	0.88 (1.31)	678.51	5.30 (16.66)	1.75 (3.12)
	2016	107	0.84 (1.04)	1019.83	7.97 (18.30)	2.83 (5.19)
Virginia rail	2015	42	0.38 (0.81)	219.72	1.72 (8.15)	0.77 (2.06)
	2016	63	0.5 (1.09)	393.93	3.08 (14.22)	1.20 (2.80)

Figure 1.1. Variable wetland cover at survey locations

Example survey locations that illustrate how the same count of marshbirds within the same radial distance at 2 survey locations where distribution of wetland cover (light blue area) is different can lead to misinterpretations about the abundances of birds in relation to habitat. Two birds counted at a survey location where there is a low proportion of wetland cover in the surrounding landscape (yellow circle) represents a much different density than 2 birds counted at a survey location with a high proportion of wetland cover in the surrounding landscape (green circle).

Figure 1.2. Map of marshbird survey locations in west-central Minnesota

We assessed abundance of marshbirds at 128 survey locations across a portion of the Prairie Pothole Region of west-central Minnesota. We grouped 5-8 survey locations together into routes, named for the primary wildlife production area along each route.

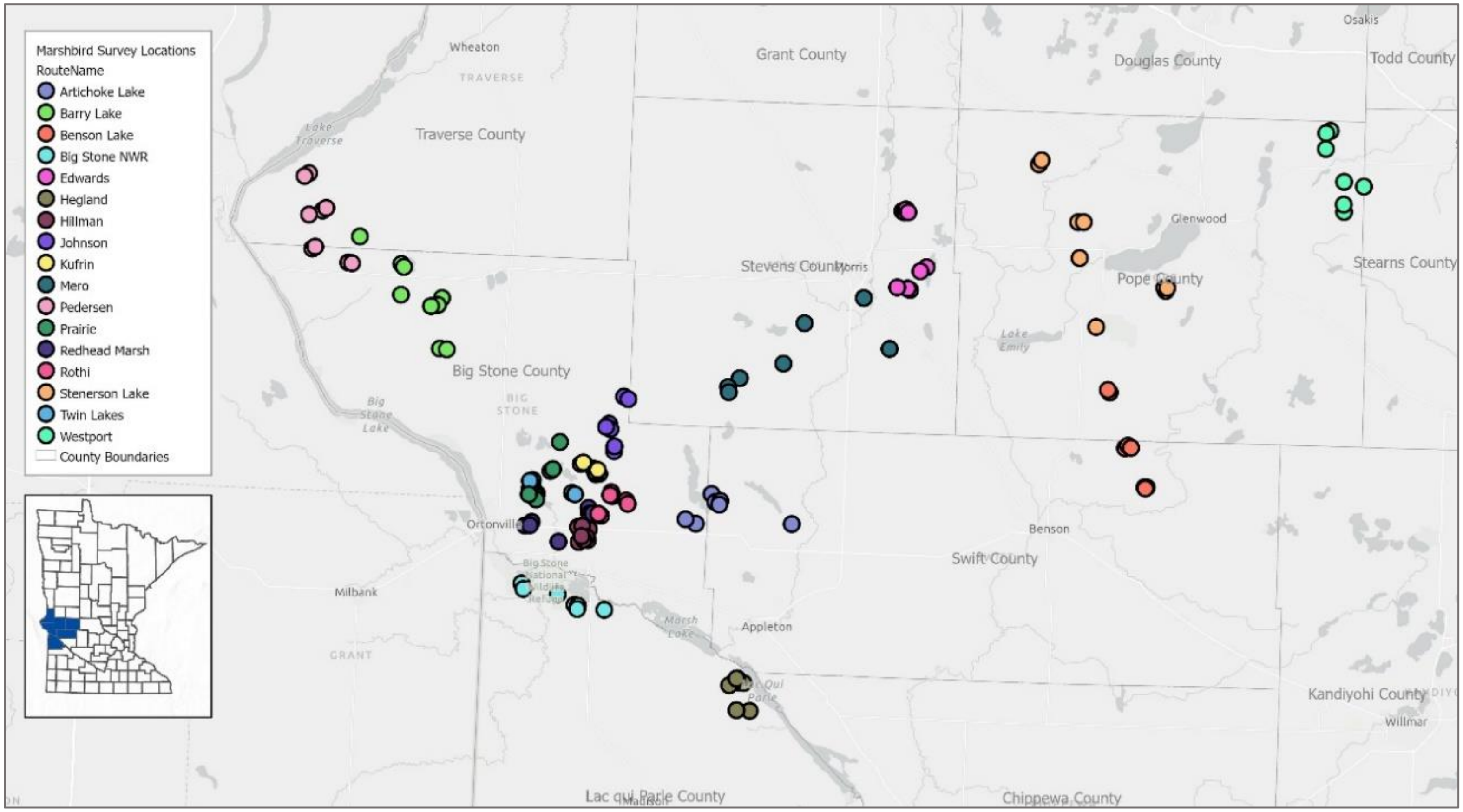


Figure 1.3. Wetland cover and marshbird detections by distance from observer

Area of wetland cover types (panel A) and sums of maximum counts (y-axis; panel B) within 23-m distance intervals (bins) from survey locations, derived from marshbird surveys in the Prairie Pothole Region of west-central Minnesota, U.S.A. We calculated truncation distance (dashed lines) as the distance from survey locations that contained 90% of observations for each species (American bittern = 230 – 253 m; pied-billed grebe = 322 – 345 m; sora = 184 – 207 m; Virginia rail = 92 – 115 m). We used maximum counts up to the truncation distance to derive estimates of density that accounted for the abundance and distribution of wetland cover around survey locations.

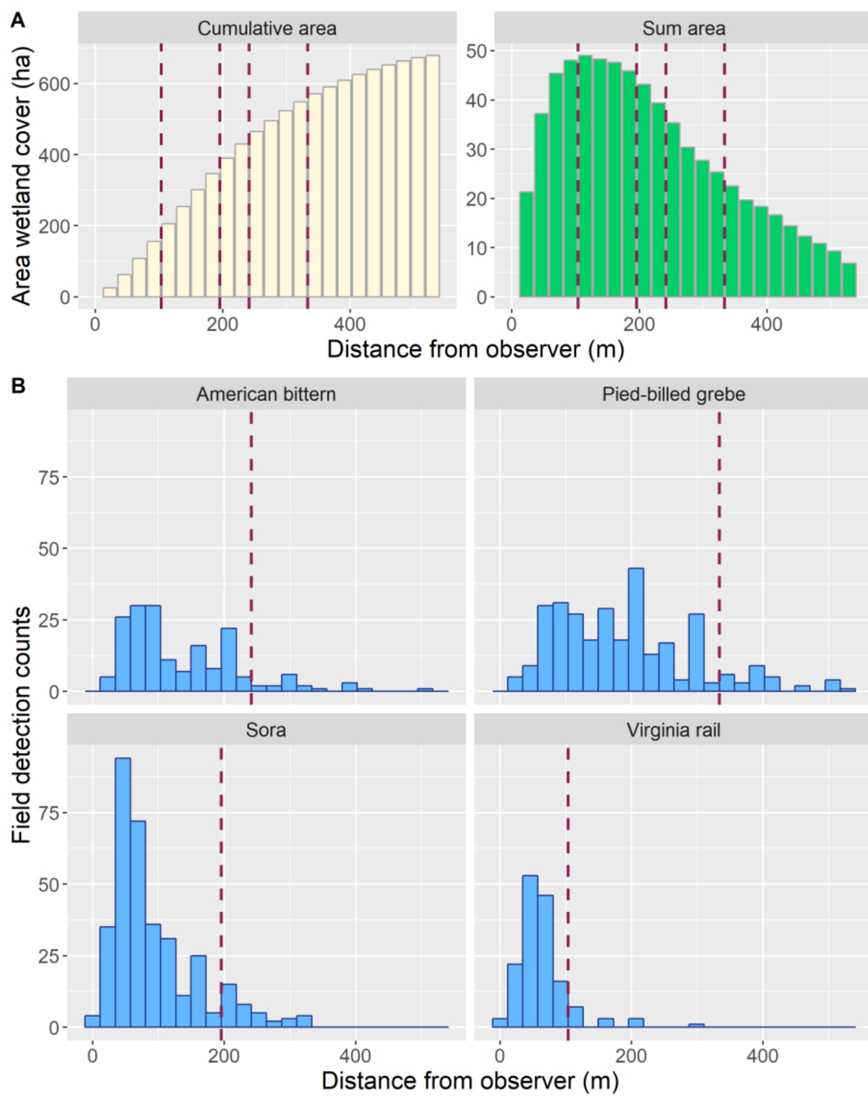
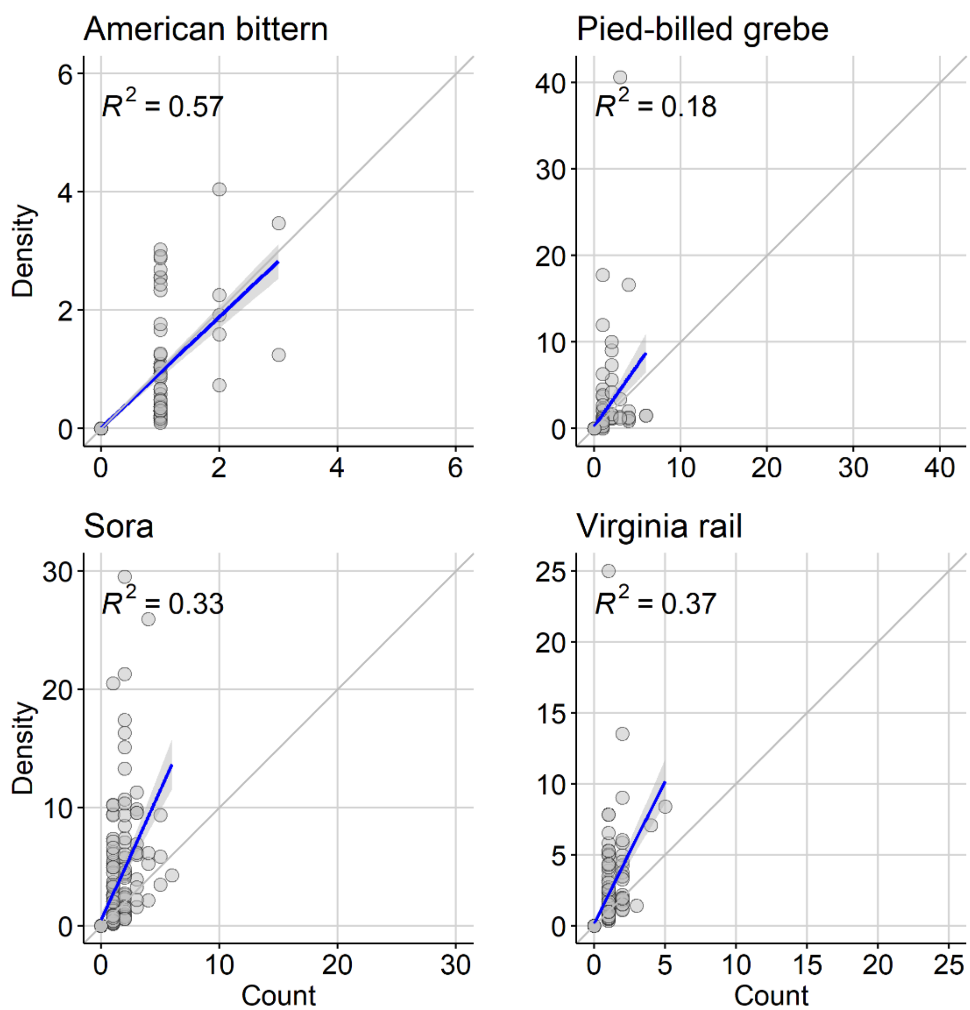


Figure 1.4. Comparing raw counts to habitat-informed densities

Maximum counts of marshbird detections from 2 visits during both 2015 and 2016 compared to estimated densities (birds per wetland ha) that account for detection probability and the distribution of wetland cover around survey locations at wetlands in west-central Minnesota, U.S.A. A least-squares regression line with slope = 1 suggested that accounting for detection probability and distribution of wetland cover around survey locations did not influence comparisons of abundances among survey locations (American bittern). Slope > 1 (sora, Virginia rail, and pied-billed grebe) indicated that counts underestimated abundances represented by density.



Chapter 2: Secretive marshbird response to prescribed burning and grazing in Prairie Pothole Region of Minnesota

Overview

Marshbirds are difficult to survey due their secretive nature and association with dense wetland vegetation. Recently developed standardized survey protocols can be used to monitor patterns of abundance, primarily at large spatial scales, but also can be used to assess marshbird response to management. We examined the abundances of 4 species of marshbirds (American bittern [*Botaurus lentiginosus*], pied-billed grebe [*Podilymbus podiceps*], sora [*Porzana carolina*], Virginia rail [*Rallus limicola*]) in relation to various histories of prescribed burning and grazing to assess the response of marshbirds to vegetation management in the Prairie Pothole Region of western Minnesota. We compared wetlands in 3 categories: those with little or no recent management activity, those with recent prescribed fire, and those with both recent prescribed fire and grazing. Marshbird abundance was not consistently related to management history, and patterns in abundance for the same species sometimes differed between the 2 years of the study (2015 and 2016), suggesting little influence of past management history. Marshbird abundance was related to wetland characteristics at both the scales of vegetation in individual wetlands and of the landscape in which wetlands were embedded. Pied-billed grebe abundance was positively associated with higher amounts of open water, soras were more abundant in wetlands with high ratios of open water to emergent vegetation, and Virginia rails were more abundant in wetlands with scrub-shrub wetland cover types. None of the covariates we considered was related to American bittern abundance. Our

results suggest that prescribed fire and grazing may not influence Prairie Pothole Region wetland characteristics enough to result in changes in marshbird abundance, but that abundance of marshbirds is primarily related to characteristics of individual wetlands that do not appear to respond to fire and grazing. Furthermore, models of marshbird abundance considering characteristics derived remotely or measured within individual wetlands suggested that different marshbird species were associated with either or both categories of covariates. However, model validation indicated that predictive ability of models was similar regardless of the suite of covariates considered.

***Key Words*:** grazing, marshbird, prescribed fire, wetland management, wetland management district

INTRODUCTION

The Prairie Pothole Region (PPR) consists of ~ 777,000 km^2 of the central North American continent (Smith et al. 1964), characterized by high densities of wetland complexes within a historically grassland-dominated landscape. This region plays a vital role in waterfowl production, provides habitat for a multitude of taxa, and delivers broad-scale ecosystem services to the landscape. However, over decades of land conversion to agriculture, and more recently to oil and gas exploitation, the PPR has suffered dramatic losses of native land cover, with direct elimination of wetlands in the region estimated at 92% (Dahl 2014). Widespread wetland loss has instigated hydrologic and ecological change that has often been detrimental to wildlife. As a result of habitat loss and decline in quality of remaining wetlands, many wildlife species, likely including secretive marshbirds, have suffered population declines (North American Bird Conservation Initiative 2016).

Current management efforts on protected natural areas are largely aimed at mitigating loss of system functionality; controlling invasive species, especially invasive vegetation; restoring wildlife habitat; and providing recreational opportunities. Remaining native ecosystems have experienced increased invasion and dominance by non-native plant species, which have been identified as one of the main contributing factors of low-quality wetland condition (Bourdaghs et al. 2015). In particular, non-native and hybrid cattail (*Typha* spp.) and reed canary grass (*Phalaris arundinacea)* can drastically alter wetland structure and establish feedback loops (Woo and Zedler 2002, Tuchman et al. 2009, Lishawa et al. 2010) that reduce suitability for wildlife over time (Zedler and Kercher 2004). Prescribed burning and grazing are management tools commonly used to combat non-native plant invasion and facilitate habitat restoration (Bansal et al. 2019; e.g., Vacek and Friske 2012). Although generally applied to upland areas surrounding wetlands, managed disturbances have both direct and indirect effects on wetlands. Repeated disturbances through prescribed burning and grazing can fragment mats of invasive cattails and, to a lesser extent, reed canary grass; remove thatch from previous years' growth; expose mud-flats to provide foraging habitat for birds and encourage native plant germination; and maintain open water patches that support the submerged aquatic base of the palustrine food chain (Hall and Zedler 2010, Wilcox et al. 2018, Bansal et al. 2019). Managed disturbances thereby create heterogeneous cover-type composition that supports higher wildlife and plant diversity (Bansal et al. 2019).

Although prescribed burning and grazing are effective tools for combating invasive plant species and are extensively applied across the region, their long-term effects on wetland-dependent wildlife are not well documented (Santisteban et al. 2011). This is particularly true for secretive marshbirds that are difficult to monitor.

To assess relationships between wildlife and management aimed at reducing invasive vegetation, we examined how the frequency of managed disturbances of prescribed burning and grazing related to the abundances of 4 species of PPR marshbirds (American bittern [*Botaurus lentiginosus*], pied-billed grebe [*Podilymbus podiceps*], sora [*Porzana carolina*], and Virginia rail [*Rallus limicola*]) in 2015 and 2016 in west-central Minnesota. In general, marshbirds exhibit higher abundances at large wetland complexes with individual wetlands of various sizes and depths (Naugle et al. 1999, Naugle et al. 2001). At the scale of individual wetlands, marshbirds exhibit higher abundances where there is a high diversity of vegetation structure, height, and density, and a high diversity of land cover types, and are positively associated with high interspersion of open water and emergent vegetation (Murkin et al. 1982, Harms and Dinsmore 2013) and negatively associated with invasive plant abundance (Blossey 1999, Glisson et al. 2015). Although recent development of standardized monitoring protocols and programs (Johnson et al. 2009, Conway 2011, Seamans et al. 2013) has increased understanding of these species' distributions and ecology (e.g., Bolenbaugh et al. 2011), there remains a lack of understanding of the relationships between marshbirds and habitat management and potential cumulative effects of management through time (Larkin et al. 2013).

Our specific objectives were to determine (1) if marshbird densities were related to management histories, categorized by various frequencies of prescribed burning and grazing, and (2) if marshbird densities were associated with wetland characteristics measured via remote sensing (e.g., ratio of open water to emergent wetland areas) or observed in the field (e.g., vegetation height, water depth). We hypothesized that repeated managed disturbances influenced wetland vegetation characteristics and that resulting vegetation and wetland characteristics would be related to marshbird abundance (Conway 1995). Based on that hypothesis, we expected that marshbird densities would be positively associated with frequency of managed disturbances aimed at reducing invasive plant abundance and would be related to associated wetland characteristics that correspond with a diversity of vegetation structures and heterogeneous landscapes. Finally, we explored how models of marshbird abundance based on wetland characteristics measured via remote sensing compared with models based on wetland characteristics observed in the field, and whether the additional effort required to obtain field-observed wetland characteristics improved inference.

STUDY AREA

We conducted surveys for marshbirds in west-central Minnesota at 76 Waterfowl Production Areas (and easements) managed by the U.S. Fish and Wildlife Service Morris Wetland Management District and Big Stone National Wildlife Refuge (45.5919° N, 95.9189° W). Our study area was characteristic of the PPR (Vacek and Friske 2012) in that land use consisted largely of row-crop agricultural production with altered hydrology that drains water through ditch systems into shallow permanent lakes. There are few remaining natural prairie grasslands and wet meadows among gently rolling hills. The small remnant and restored natural areas have high boundary-to-area ratios, making them vulnerable to invasion and dominance by aggressive, non-native plants (Matthews et al. 2009). Management objectives in this system focus on restoring and maintaining diverse native plant communities most commonly using prescribed burning and grazing (Vacek and Friske 2012). Throughout our study area it was common that both types of managed disturbances were applied to the same parcel in different years, typically prescribed burning once during a 3-to-5-year period, and grazing in 3 consecutive growing seasons during a 10-year period.

DATA COLLECTION

Marshbird Abundance

In 2015 and 2016 we conducted marshbird surveys at each survey location (see below) during the early part of the marshbird breeding season. In 2015, we conducted the first surveys during 14 – 19 May and the second surveys during 30 May – 17 June. In 2016, we conducted the first surveys during 5 – 9 May and the second surveys 20 – 27 May. In each year, we surveyed each location once during the period around sunrise (~0.5 hours before through up to 3 hours past sunrise) and once during the period around sunset (~3 hours before and ~0.5 hour after sunset). The same observer visited all survey locations on 1 route during a 3.5-hour crepuscular survey period, and survey locations on a route were visited in the same order on both survey occasions.

The observers (2 in 2015 and 2 in 2016, with 1 observer being the same individual in both years) conducted surveys following standardized protocol (Conway 2011). An observer approached the survey location on foot and allowed a 3-to-5-minute settling period during which she recorded information on environmental conditions (i.e., wind speed and direction, ambient noise level and disturbances) and recorded all bird species observed at or near the survey wetland. Following the settling period, we broadcasted recorded calls from a SanDisk Clip Sport mp3 player (SDMX24; SanDisk Corporation, Milpitas, CA) at 80-90 dB from 1 m away using a portable game speaker (Cass Creek Big Horn Remote Speaker, Cass Creek, Grawn, MI). In 2015, a survey lasted 10 minutes and included broadcasted calls in order: least bittern (*Ixobrychus exilis*), sora, Virginia rail, American bittern, pied-billed grebe (recommended species, order of broadcast, and standardized recorded calls obtained from program organizer http://ag.arizona.edu/research/azfwru/NationalMarshBird/). In 2016 a survey lasted 11 minutes using broadcasted calls similar to above, with the addition of yellow rail (*Coturnicops noveboracensis*) as the second species in the broadcast sequence. The observer recorded all aural and visual detections of these 6 marshbird species and estimated the compass bearing (degrees) and distance (meters) to the bird's location using a compass and laser rangefinder and by relating features in the landscape to aerial photographs with printed distance rings around the survey location. We summarized all aural and visual detections into species counts for each survey visit per year at each survey location. The protocol for this study was approved by the University of Minnesota Institutional Animal Care and Use Committee (IACUC protocol #1503-32456A).

To account for variable abundance and distribution of wetland cover surrounding survey locations, we transformed marshbird counts into estimates of density (birds per wetland ha) for each of the 4 most frequently detected species (see Chapter 1; we had insufficient counts to use in analyses for least bitterns and yellow rails). Briefly, we adjusted for detection probability as a function of distance from the observer to the bird and for the amount and distribution of wetland cover out to a distance within which we observed 90% of our counts. We used the maximum count at each survey location within a year to derive estimates of density. Unlike counts, estimates of marshbird density were directly comparable across survey locations, and we used density estimates to model marshbird abundance as a function of management history, and as related to measures of wetland characteristics (see below).

Management Histories

To assess whether marshbird abundance was related to vegetation management histories (prescribed burning and grazing), we sampled wetlands that varied in frequency and variety of managed disturbances. We assumed that as prescribed burning and grazing directly impacted wetland vegetation these management actions also indirectly influenced marshbird abundance, and that repeated and more recent managed disturbances had stronger effects than those farther in the past (Santisteban et al. 2011). We therefore classified management histories based on the frequency of prescribed burning that occurred during 2000 – 2014 into categories of low frequency burning (1 incident of prescribed burning; Low burn), high frequency burning (3 incidents of prescribed burning; High burn), and a third category with both high frequency burning and grazing ($\geq$3 incidents prescribed burning and $\geq$1 incidents grazing; Burn & graze).

We used a geographic information system (ESRI ArcGIS 10.2; Environmental Systems Research Institute, Redlands, CA) to examine U.S. Fish and Wildlife Service management records of the study area (Refuge Lands Geographic Information System; U.S. Fish and Wildlife Service 2008b) to identify wetlands that fit our management history categories. Typically, prescribed burning and grazing were applied to distinct areas (e.g., a portion of a specific management unit) that contained multiple wetlands. When this was the case, we randomly selected wetlands within the distinct management areas to survey for marshbirds. We included in our sample wet meadows, emergent wetlands, shallow marsh, and deep marsh wetland types of temporary, seasonal, and semi-permanent hydroperiod types (National Vegetation Classification Standard [U.S. Fish and Wildlife Service 2008a]; wetland types 2, 3, and 4 of the Circular 39 classification system). At each selected wetland, we established 1 survey location near the edge of the basin that had an unobstructed vantage point and that was ≥400 m from any other survey location. We grouped 5 – 9 spatially clustered survey locations together on a route, including as balanced a representation of wetland types among management history categories as possible (Appendix B), and surveyed all wetlands on a route on the same day. After the completion of marshbird surveys in 2015, both prescribed burning and grazing occurred in the study area that resulted in additional wetlands within our defined management history categories; we included 16 new survey locations in 2016. In total, we surveyed 113 wetlands to assess whether marshbird abundance was related to vegetation management histories.

Wetland Characteristics

To evaluate how marshbird abundance was related to wetland characteristics, we used both remotely sensed data and field-based methods to measure wetland attributes known to be associated with marshbird abundance. Of remotely sensed measures, wetland size is often identified as the most important factor related to marshbird abundance (Brown and Dinsmore 1986, Willard 2011, Tozer et al. 2010). Soras and Virginia rails are typically associated with shallow and seasonally flooded wetlands as they forage for macroinvertebrates in floating aquatic vegetation and mud flats. In contrast, American bitterns and pied-billed grebes are often associated with larger, permanent wetlands where they hunt for vertebrate prey in the water (Brown and Dinsmore 1986, Fairbairn and Dinsmore 2001, Monfils et al. 2012); in the PPR depth of wetland is positively associated with wetland area. Other studies (Brady and Paulios 2010, Santisteban et al. 2011) indicated that marshbird presence was related to diversity of wetland cover types, where American bitterns had higher occupancy at wetlands with large extent of emergent vegetation cover, and pied-billed grebes had higher occupancy at wetlands with large areas of open water and some areas interspersed with emergent vegetation. Marshbird abundance has strong relationships to edge-to-area ratio of emergent vegetation patches (Fairbairn and Dinsmore 2001, Chabot et al. 2014), and marshbird occupancy is typically negatively related to tree cover surrounding wetlands (Willard 2011, Tozer 2016). Using National Wetland Inventory data (Minnesota Department of Natural Resources 2016), we measured area (ha) of wetland cover types within each survey wetland (Circular 39 [Shaw and Fredine 1956] class types wet meadow, shallow marsh, deep marsh, open water, shrub swamp, wooded swamp), and calculated an index of wetland shape (edge length m / sqrt [area basin ha]) and the sum of all wetland cover area within a 600-m radius of each survey location. In addition, we classified and measured the area of all land cover types (Appendix C) within 1 km of survey locations using high-resolution aerial imagery collected by the U.S. Fish and Wildlife Service specifically for our study. Using object-based image analysis software (eCognition Developer; Trimble Geospatial, Sunnyvale, CA), we performed segmentation analysis and supervised classification of broad-scale cover classes. We transferred vector spatial data to GIS and refined cover class identification in overlay analyses for Waterfowl Production Areas with U.S. Fish and Wildlife Service National Vegetation Classification Survey data (U.S. Fish and Wildlife Service 2008a) and for non-protected lands with a 15-m resolution land cover raster dataset (Rampi et al. 2016). Finally, we summed areas of each cover type and calculated a ratio of open water cover class to emergent wetland cover types within a 1-km radius from each survey location.

We also quantified field-observed wetland characteristics to represent relationships between marshbird abundance and management (i.e., wetland characteristics that were related to marshbird abundance that also showed effects from prescribed burning and grazing). Of field-observed measures of wetland characteristics, marshbird species are commonly associated with vegetation density (Naugle et al. 1999, Naugle et al. 2001), vegetation height (Lor and Malecki 2006), extent or diversity of tall emergent vegetation patches (O’Neal et al. 2008), amount of litter (Johnson and Dinsmore 1986, Santisteban et al. 2011), amount of woody vegetation present

(Bolenbaugh et al. 2011), and water depth and proportional cover of standing water within the wetland basin (Lor and Malecki 2006). We conducted field observations at marshbird survey wetlands following completion of marshbird surveys in 2015 and 2016. In 2015 we assessed 111 wetlands during 1 July – 2 August. In 2016 we assessed wetlands at the 16 survey locations established in 2016 during 20 June – 28 July 2016, and randomly selected 40 wetlands from 2015 to resample. At each wetland we established ≥1 transect along a random bearing from the survey location into the emergent vegetation zone of the wetland. We sampled ≥5 plots spaced 10 m apart along the transect, beginning with a plot midway through the wetland-upland transition zone, continuing until either we encountered >1.5-m-deep open water or traveled out the far side of the wetland. If we sampled <5 plots on the first transect, we similarly established a second transect and sampled plots until reaching a minimum sample of 5 plots. At each plot we measured water depth and litter depth, and estimated vegetation densities along horizontal strata by counting the number of vegetation touches (from leaf blade or stem) within 0.5-m segments of a 1.5-m pole held vertically on the ground 90 degrees and 4 m away from the transect line (variation of the Step-Point Method; Evans and Love 1957). We identified and recorded additional plant species within a 1-m-wide plot from the 1.5-m pole to the transect. We identified plants to family or a lower taxonomic level when possible. Finally, from the 1-m height at the pole we measured low and high visual obstruction (Robel et al. 1970) looking back towards the transect line at a 3-m pole marked in 10-cm increments. We summarized measures across plots at each survey location into mean of vegetation height (m), variation (standard deviation) of vegetation height (m), density of cattails (mean touches on 1.5-m pole per plot), maximum depth of litter (m), mean depth of water (m), and proportion of plots where woody species (e.g., *Salix* spp.) were present. For our objective related to assessing marshbird abundance in relation to wetland characteristics, we used the same data as in addressing our first objective, with the addition of 14 survey locations for which we did not have management histories but did have estimates of marshbird densities and wetland characteristics. In total, we assessed 127 survey locations to evaluate relationships between marshbird abundances and wetland characteristics (Appendix B).

Both remotely sensed and field-observed measures of wetland characteristics were highly correlated. Therefore we input all measures into a Principal Component Analysis using Program R (PCA; R function prcomp [stats package]; R Core Team 2017) to help us identify and remove redundant variables, and considered a reduced set of variables in models of marshbird abundance that represented the variability across survey locations. We scaled and centered variables and compared resulting PCA scores; from each of the top Principal Components (PCs) that accounted for >60% of cumulative variation (Appendix D) we selected the single covariate with the highest PCA loading to consider in models of marshbird abundance.

ANALYSIS

Models of Marshbird Abundance and Management Histories

We examined relationships between marshbird densities observed in 2015 and 2016 and 3 management history categories (Low burn, High burn, and Burn & graze) with analysis of variance (ANOVA; R function aov [stats package]). We then directly compared densities among the 3 management history categories with Tukey's Honest Significant Differences (R function TukeyHSD [stats package]).

Models of Marshbird Abundance and Wetland Characteristics

To identify wetland characteristics associated with marshbird abundance and to evaluate whether the addition of field-observed covariates improved on models of abundance based solely on remotely sensed covariates, we examined relationships between marshbird densities and wetland characteristics derived solely from remote sensing, solely from field observations, and from field-observed covariates added to the best-supported model considering remotely sensed covariates. We randomly selected a portion of data to build models and reserved the remaining data to test models via cross validation. Briefly, we randomly selected 1 year of data from survey locations where we had 2 years of data, and 2/3 of data from survey locations where we had only 1 year of data (Appendix E). We used stepwise generalized linear model selection and identified the model with lowest AIC as best-supported (R functions glm [stats package] and stepAIC [mass package]).

Finally, we assessed performance of the best-supported models of marshbird abundance with cross validation. We split the reserved testing data for multiple rounds of cross validation based on our survey design. Briefly, the first set of reserved data was comprised of data from the alternate years from survey locations with 2 years of paired marshbird and wetland data. The second set of reserved data was comprised of 1/3 of data from survey locations where we only had 1 year of paired data. The third set of reserved data was comprised of data from survey locations in years where we had marshbird data but did not measure wetland characteristics in the same year and therefore substituted the other year's wetland data from those survey locations (Appendix E). We input reserved wetland characteristic covariate data into each of the 3 best-supported models (i.e., remotely sensed model, field-observed model, and combined-variable model) to predict marshbird densities for each species (R function predict [stats package]; 36 cross-validation tests total). We measured model performance by calculating the root mean squared error (RMSE; R function RMSE [caret package]) of the difference between marshbird densities predicted from our best-supported models and observed marshbird densities included in the reserved data, where lower RMSE values indicated better predictive performance.

RESULTS

Marshbird Abundance

During 2015 and 2016, we conducted 348 surveys and detected 483 marshbirds of the 6 species for which we broadcasted calls. Soras were the most commonly detected species (42.24% of detections; at 45.36% of survey locations), followed by pied-billed grebes (23.6%; at 24.74% of locations), Virginia rails (19.25%; at 48.45% of locations), American bitterns (13.25%; at 26.80% of locations), and least bitterns (1.66%; at 5.15% of locations; Appendix A); we did not detect yellow rails. We detected fewer marshbirds in 2015 (mean = 1.21 [SE=0.15] birds per survey) than in 2016 (mean = 1.55 [SE=0.11] marshbirds across 5 species for which we broadcasted calls; Fig. 2.1).

Wetlands we surveyed ranged in size from 0.18 to 35.30 ha (mean = 5.36 ha [SD=7.17]). Only 12% of survey locations included forested wetland or scrub/shrub wetland cover types. Across treatment categories, covariates derived from remotely sensed data were similar between High burn and Burn & graze management history categories (Appendix C); however, on average Low-burn wetlands had higher areas of row crops and lower areas of managed natural grass within 1-km radius around survey locations (Appendix C).

Models of Marshbird Abundance and Management Histories

We did not observe consistent relationships between marshbird densities and disturbance history categories for any of the 4 most-frequently detected marshbird species (Fig. 2.2). Pied-billed grebes had slightly higher densities in 2016 in wetlands of Low burn versus Burn & graze (Table 2.1, Fig. 2.2). Soras had higher densities in 2016 in wetlands with Low burn versus High burn (Table 2.1, Fig. 2.2). However, these patterns were not consistent between years (Table 2.1, Fig. 2.2).

Models of Wetland Characteristics

We considered 8 remotely sensed covariates and 6 field-observed covariates in models of marshbird densities (Table 2.2; Appendix D). The best-supported models of marshbird densities varied among species and we report cross validations from the first set of reserved data (for additional results see Appendix E). Overall, cross validation revealed little variation in RMSE among best-supported models that included remotely sensed vs. field-observed covariates vs. models of field-observed covariates added to the best-supported models of remotely sensed covariates (Table 2.2). American bittern densities were not related to remotely sensed covariates; however, for field-observed covariates, American bittern densities were negatively related to the presence of woody species at the survey wetland and standard deviation of vegetation height. The best-supported model of American bittern densities that involved field-observed covariates after considering remotely sensed covariates was the same as the best-supported model that considered only field-observed covariates. Pied-billed grebe densities were positively related to total wetland area (model considering remotely sensed covariates) and positively related to water depth (model considering field-observed covariates). The best-supported model of pied-billed grebe densities that sequentially considered remotely sensed covariates then field-observed covariates included both wetland area and water depth, although cross validation indicated that the overall best-supported model of pied-billed grebe density included only water depth (Table 2.2). Sora density was related positively to the ratio of open water to emergent wetland cover within 1 km and negatively to the area of all wetland cover within 1 km (model considering remotely sensed covariates). No covariates were related to sora density in models that considered field-observed covariates, but standard deviation of vegetation height (model considering field-observed covariates) was included in the best-supported model of sora density when field-observed covariates were considered along with remotely sensed covariates (Table 2.2). The null model of sora density had the lowest RMSE. Virginia rail density was positively related to presence of scrub-shrub wetland type at the survey wetland (model considering remotely sensed covariates) but not related to any field-observed covariates. Cross validation indicated that the best-supported model of Virginia rail density was the null model.

DISCUSSION

We explored potential relationships between marshbird abundance and management histories, specifically of various frequencies of prescribed burning and grazing, in the Prairie Pothole Region of western Minnesota, U.S.A. Although estimated densities varied for the 4 most-frequently detected marshbirds (American bitterns, pied-billed grebes, soras, and Virginia rails), overall we did not observe patterns in marshbird density related to management histories within or among species that were consistent in both years of our study. We suggest 2 potential reasons for a lack of measurable relationship between marshbird abundance and management history in this system. First, both prescribed burning and grazing have considerable effects on upland systems (Sojda and Solberg 1993, Kettenring and Adams 2011), but climate and geographic context cause variable extents of disturbance effects, which are generally less dramatic on the vegetation in wetlands within those systems. It may be that neither prescribed burning nor grazing substantively changed wetland conditions in this system. Studies investigating invasive wetland vegetation control methods often note that combining management types produces the best results (Kostecke et al. 2005, Mérő et al. 2015), and we attempted to evaluate the effects of both prescribed fire and grazing in our study (i.e., our Burn & graze category). However, there were fewer occurrences of grazing throughout our study area than of prescribed burns, and fewer still that occurred recently relative to when we conducted marshbird surveys, likely limiting our ability to assess the potential effect of grazing on marshbird abundance in our study. Second, we did not measure other factors that could influence marshbird abundance, such as food availability or predation pressure, which may influence marshbird abundance at individual wetlands during the breeding season.

Though our study did not examine the direct effect of management histories on wetland characteristics, our best-supported models indicated relationships of marshbird abundance to wetland characteristics that represented diversity of vegetation structures and heterogeneous landscapes, such as standard deviation of vegetation height and ratio of open water to emergent vegetation. Furthermore, none of the categories of models of marshbird densities we considered (i.e., models based on remotely sensed covariates, field-observed covariates, or a combination of the two sets of covariates) consistently resulted in the best-supported models. American bittern density was lower at wetlands with more woody vegetation (remotely sensed) and higher standard deviation of emergent vegetation height (field-observed). There is some suggestion that nesting marshbirds may avoid areas with tall, woody vegetation to decrease predation risk from avian and mammalian predators that use woody vegetation as hunting cover (Darrah and Krementz 2011). However, the null model of American bittern abundance had slightly lower RMSE than the other best-supported models, indicating that the covariates we considered, including those representing woody vegetation, did not explain most of the variation in American bittern abundance.

Best-supported models of pied-billed grebe densities indicated they were positively related to wetland size and water depth (Table 2.2), similar to associations reported elsewhere (Forbes et al. 1989, Lor and Malecki 2006). Grebes prefer larger wetlands within large wetland complexes where they can forage for prey underwater

(Darrah and Krementz 2010). Sora densities were related positively to the ratio of open water to emergent vegetation cover and negatively to the area of wetlands surrounding survey locations. Lor and Malecki (2006) and Bolenbaugh et al. (2011) reported that soras were found more often in smaller wetlands, but we did not observe a relationship between sora densities and size of wetlands. We also observed a positive relationship between open water in wetlands and sora densities, which may indicate a nonlinear relation—positive where emergent cover dominates and negative where open water dominates. Lor and Malecki (2006) and Bolenbaugh et al. (2011) both reported that soras were associated with wetlands with extensive emergent vegetation. Finally, Virginia rail densities were positively related to presence of scrub-shrub cover type (remotely sensed), inconsistent with evidence suggesting breeding marshbirds avoid wetlands with abundant woody vegetation, both shrubs and trees (Johnson and Dinsmore 1986, Darrah and Krementz 2010, Harms and Dinsmore 2013). However, this relationship was not evident in models based on field-observed covariates, and overall, the null model considering field-observed covariates had the lowest RMSE.

In summary, we found no consistent relationships between densities of the most commonly detected marshbird species and management histories of prescribed burning and grazing of the area surrounding and in wetlands in the PPR in western Minnesota. Relationships of marshbird densities to wetland covariates varied among species, and we observed no clear patterns when comparing models of densities considering wetland covariates measured from different-scale methods, i.e., via remote sensing or observed in the field. Although we did not observe influences of recent past management histories of prescribed burning and grazing on marshbird densities, American bitterns, pied-billed grebes, soras, and Virginia rails all appeared to use areas with heterogeneous grassland and wetland complexes with basins that provide a mix of open water and emergent vegetation, little cover of woody species, and that contain wetlands of variable size and water depths. Our results further suggest that models of marshbird densities in landscapes similar to the one we studied could be appropriately informed using measures derived from free and publicly available remotely sensed data, especially in situations where limited resources restrict the ability for field-based surveys.

Table 2.1. Testing marshbird densities at different management histories

Estimated marshbird densities and relationships to management history categories during 2015 and 2016 in the Prairie Pothole Region in west-central Minnesota, U.S.A. Mean densities of marshbirds per wetland ha for the 4 most commonly detected species are displayed for each year and by management history categories of Low burn (L; sites with 1 year prescribed burning during the previous 14 years), High burn (H; sites with 3 years of prescribed burning during the previous 14 years), and both Burn & graze (G; sites with >3 years of prescribed burning and grazing during the previous 14 years). We used analysis of variance (ANOVA) to detect differences in marshbird densities across management history categories and examined results between pair-comparisons of management history categories using Tukey's honest significance test. Relationships of mean densities to disturbance history categories are described with ANOVA *F* values [PR(>F)], and *P*-values of comparisons among the 3 categories.

		2015		**2016**			**2015**	**2016**
Species	**Management history category**	**Mean density**	***F* value**	**Mean density**	***F* value**	**Comparison**	***P***	***P***
American bittern	Low burn	0.06	1.86	0.38	1.31	H to L	0.28	0.46
	High burn	0.25	0.16	0.15	0.27	G to L	0.23	0.86
	Burn & graze	0.27		0.47		G to H	0.98	0.26
Pied-billed grebe	Low burn	0.11	0.04	0.22	3.06	H to L	1.00	0.23
	High burn	0.11	0.96	0.08	0.05	G to L	0.96	0.05
	Burn & graze	0.09		0.03		G to H	0.98	0.82
Sora	Low burn	1.60	1.63	3.92	3.02	H to L	0.80	0.06
	High burn	1.13	0.20	0.98	0.05	G to L	0.37	0.99
	Burn & graze	2.65		3.80		G to H	0.19	0.11
Virginia rail	Low burn	0.86	1.79	1.40	0.26	H to L	0.42	1.00
	High burn	1.67	0.17	1.39	0.78	G to L	0.65	0.79
	Burn & graze	0.26		1.06		G to H	0.15	0.83

Table 2.2. Wetland characteristic model selection and cross validation

Parameter estimates in best-supported models of marshbird densities considering remotely sensed covariates (Remote), field-observed covariates (Field), and adding field-observed covariates to the best-supported model considering only remotely sensed covariates (Both). We evaluated models using cross-validation and calculated root mean squared error (RMSE) of comparisons between model-predicted marshbird densities to observed marshbird density. Covariates in best-supported models varied among species, and across all species' models 6 of the 14 covariates considered were included. Cross validation using the most robust reserved test data revealed that models derived from different sets of wetland covariates had similar predictive performance, and null models had lowest RMSE for 3 of the 4 species.

Model	American Bittern Remote	American Bittern Field	American Bittern Both	Pied-billed grebe Remote	Pied-billed grebe Field	Pied-billed grebe Both	Sora Remote	Sora Field	Sora Both	Virginia rail Remote	Virginia rail Field	Virginia rail Both
Intercept Estimate	0.222	0.472	0.472	-0.009	0.013	-0.124	3.77	2.332	4.839	0.752	0.884	0.752
(Standard Error)	(0.069)	(0.134)	(0.134)	(0.07)	(0.042)	(0.078)	(0.937)	(0.406)	(1.165)	(0.246)	(0.239)	(0.246)
Remotely sensed covariates												
Survey wetl. area [log(ha)]	--	--	--	--	--	--	--	--	--	--	--	--
Shallow marsh proportion	--	--	--	--	--	--	--	--	--	--	--	--
Temp. flooded proportion	--	--	--	--	--	--	--	--	--	--	--	--
Scrub/shrub wetl. present	--	--	--	--	--	--	--	--	--	1.626 (0.86)	--	1.626 (0.86)
Wetland type 1 proportion	--	--	--	--	--	--	--	--	--	--	--	--
Hay / pasture proportion	--	--	--	--	--	--	--	--	--	--	--	--
Open water:emergent	--	--	--	--	--	--	0.051 (0.029)	--	0.051 (0.029)	--	--	--
All wetland types area (ha)	--	--	--	0.001 (0.001)	--	0.002 (0.001)	-0.023 (0.012)	--	-0.025 (0.012)	--	--	--
Field-observed covariates												
Veg height mean (m)	--	--	--	--	--	--	--	--	--	--	--	--
Litter depth max. (m)	--	--	--	--	--	--	--	--	--	--	--	--
Cattail present (mean plot)	--	--	--	--	--	--	--	--	--	--	--	--
Mean water depth (m)	--	--	--	--	0.473 (0.181)	0.534 (0.181)	--	--	--	--	--	--
Woody species present	--	-0.578 (0.405)	-0.578 (0.405)	--	--	--	--	--	--	--	--	--
Vegetation height std. dev.	--	-0.004 (0.002)	-0.004 (0.002)	--	--	--	--	--	-0.020 (0.013)	--	--	--
Statistics												
AIC	207.5	206.2	206.2	51.7	47.4	45.1	548.7	553.7	548.3	448	449.6	448
Adjusted R^2	--	0.03	0.03	0.01	0.06	0.09	0.07	--	0.08	0.03	--	0.03
RMSE	0.817	0.825	0.825	0.285	0.278	0.280	5.235	5.181	5.250	2.352	2.291	2.352

Figure 2.1. Marshbird densities varied in 2015 and 2016

Estimated marshbird densities (mean birds per wetland ha) derived from surveys at 111 wetlands in 2015 and 127 wetlands in 2016 in the Prairie Pothole Region in western Minnesota, U.S.A. Based on estimates of density, rails had higher abundance than the other species (soras = 3.08 birds per wetland ha across survey locations in 2016, and Virginia rails = 1.29 birds per wetland ha, American bitterns = 0.343 birds per wetland ha, pied-billed grebes = 0.13 birds per wetland ha).

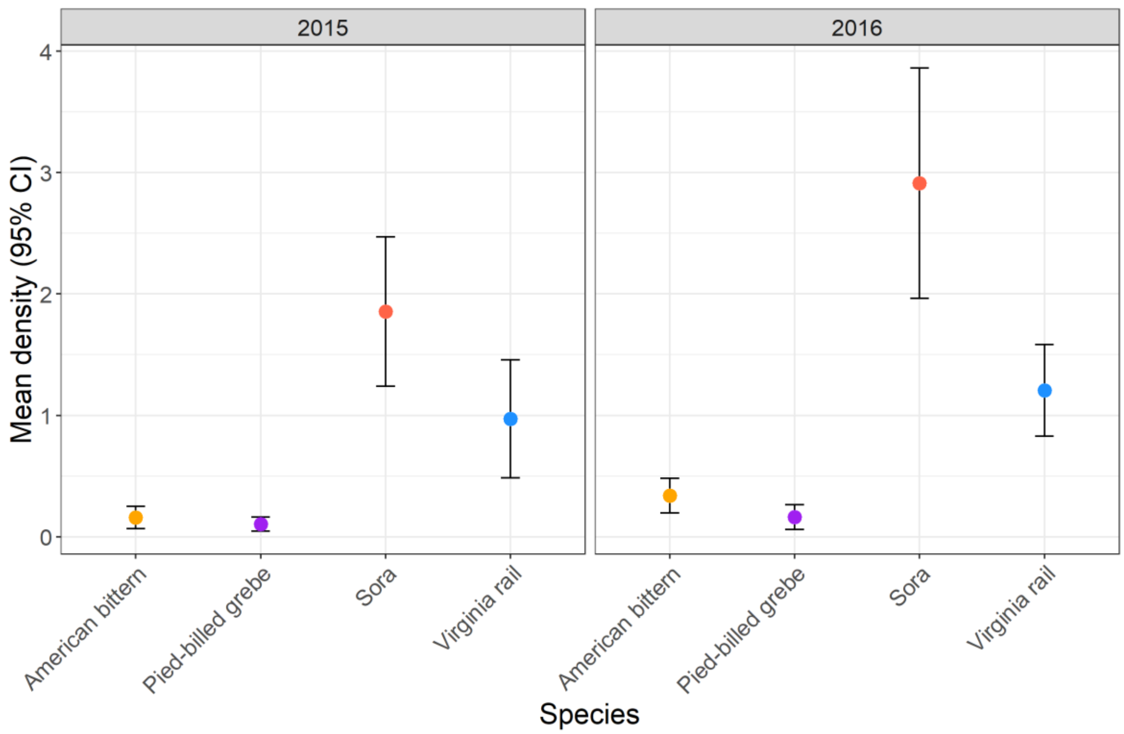
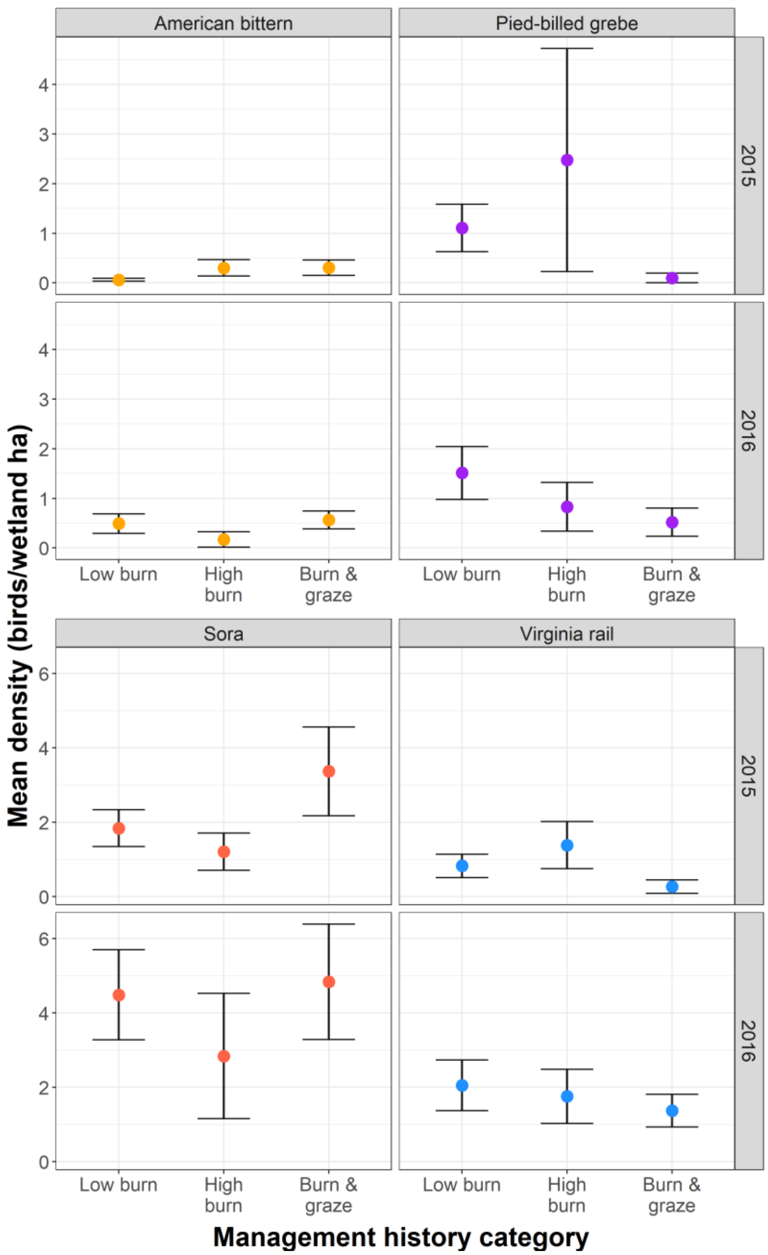


Figure 2.2. Comparing marshbird densities at different management histories

Estimated marshbird density by management history categories observed during the marshbird breeding seasons in 2015 and 2016 in Prairie Pothole Region wetlands in western Minnesota, U.S.A. Mean densities (birds per wetland ha) across survey locations (n = 111 in 2015; n = 127 in 2016) for the 4 most commonly detected species. We did not observe differences in densities among management history categories that were consistent across years.



Chapter 3: Secretive marshbird response to herbicide control of cattail in northwestern Minnesota

Overview

A high proportion of wetlands in the Prairie Pothole Region of central North America have been converted to agricultural production, and remaining areas of wetland continue to experience ecological change via the invasion and spread of non-native species. Non-native and hybrid cattail (*Typha angustifolia*, *Typha x glauca*) spread aggressively and displace native vegetation, especially in large, impounded wetlands. Management of wetlands often includes broad-scale herbicide application intended to break up mats of cattail and restore areas to hemi-marsh conditions. Although restoration of wildlife habitat is a common goal of such management, marshbird response to invasive cattail control is poorly understood. To evaluate the response of marshbirds to chemical control of invasive cattail, we conducted standardized call-broadcast surveys for 6 species of secretive marshbirds at treatment (herbicide application) and paired control (no herbicide application) sites within 9 wetland impoundments in northwestern Minnesota, United States. We conducted marshbird surveys during the breeding season prior to herbicide application in the fall of 2015 and during the 3 breeding seasons after herbicide application (2016 – 2018). We observed no change over the study period in abundance of least bitterns (*Ixobrychus exilis*). For the other marshbird species, mean differences in abundance did not change between treatment and control sites in the first year following herbicide application. However, abundances increased at treatment sites over the next 2 breeding seasons (2017 and 2018) for American bitterns (*Botaurus lentiginosus*), soras (*Porzana carolina*), and Virginia rails (*Rallus limicola*); pied-billed grebes (*Podilymbus* *podiceps*) showed a similar though not statistically significant pattern, indicating that these species responded positively to herbicide application to control invasive cattail.

INTRODUCTION

The Prairie Pothole Region (PPR) of central North America was once a vast complex of prairie grasslands and glaciated wetlands and lakes (Dahl 2014). Conversion of wetlands for agricultural uses and other purposes via drainage and other changes to hydrology have eliminated extensive areas of wetlands and disrupted and altered patterns of water flow and nutrient cycling, bolstering conditions favorable to invasion by non-native plants (Kantrud and Newton 1996; Zedler and Kercher 2004; Blann et al. 2009; Tuchman et al. 2009; Kloiber and Norris 2013). In PPR wetlands, dominance of invasive plants has resulted in reduced diversity of the wetland food web, reducing habitat quality for wildlife species that inhabit or otherwise depend on wetlands (Weller 1981; Johnson and Dinsmore 1986; Mulhouse and Galatowitsch 2003). Despite elimination and alteration of extensive areas, wetlands in the PPR remain important to an array of wildlife species, although it is not well understood how some wildlife respond to altered characteristics of wetlands dominated by invasive vegetation or how wildlife respond to efforts to restore native vegetation communities (National Research Council 1992; Ratti et al. 2001; Pulfer et al. 2014).

Land managers aim to restore wetlands to conditions that existed prior to becoming dominated by invasive vegetation (ex. Minnesota Prairie Plan Working Group 2018, Vacek and Friske 2012), under the assumption that such conditions will support a diversity of wetland-dependent wildlife (Delphey and Dinsmore 1993; VanRees-Siewert and Dinsmore 1996; Glisson et al. 2015). In particular, management of restored wetlands in the PPR often focuses on manipulating cattail (*Typha* spp.; Zedler 2000). Vegetation communities in many PPR wetlands have shifted from arrays of native species in diverse and complex spatial heterogeneity surrounding broadleaf cattail (*Typha latifolia*) marsh, into dense monocultural stands of more robust and aggressive non-native narrowleaf (*Typha angustifolia*) and hybrid (*Typha x glauca*) cattail (hereafter, cattail; Galatowitsch et al. 1999; Bourdaghs et al. 2015). As a result, many remaining wetlands have diminished ecological quality characterized by reduced plant diversity, structural homogeneity, and altered hydrology from accreted sediment, etc. (Tuchman et al. 2009; Spyreas et al. 2010). The most widespread approach to manage extensive areas dominated by cattail is direct application of herbicide (Bansal et al. 2019), which fragments monocultural patches of vegetation dominated by cattail (Linz and Homan 2011). On large, impounded wetlands, herbicide application breaks up floating beds of cattail, thereby creating a more heterogeneous mixture of open water and emergent vegetation with higher edge density among vegetation cover types, which favors regeneration of native emergent species (Linz et al. 1994; Galatowitsch 2006; Linz and Homan 2011). These conditions are believed to be more favorable for marshbirds (Bolenbaugh et al. 2011); however, there are few studies that investigate effects on wildlife from widespread herbicide application to control invasive wetland vegetation (Linz et al. 1994; Linz and Blixt 1997; Linz and Homan 2011), and the response of marshbird populations to changes that result from reducing large patches of cattail is unknown.

In general, marshbirds prefer habitat with patches of emergent vegetation interspersed with open water or mudflats (Lor and Malecki 2006), high edge-to-interior ratio (Chabot et al. 2014), and plant communities with varying structure of canopy height and density (Johnson and Dinsmore 1986), although generally free of woody vegetation (Bolenbaugh et al. 2011; Harms and Dinsmore 2013). Most species rely on seasonally dynamic water levels, and some species, particularly rails (Family Rallidae), are abundant in wetlands that include diverse vegetation structures and patchy interspersion of cover types (Johnson and Dinsmore 1986; Zimmerman et al. 2002). If treating areas dominated by cattail with herbicide restores these conditions, then marshbird use and abundance would likely increase in wetlands following herbicide application (Linz and Blixt 1997).

We evaluated the assumption that marshbird populations respond positively to changes in wetland condition from herbicide management of cattail at large impounded wetlands in northwestern Minnesota, United States. We conducted initial surveys for marshbirds following standardized protocols that included broadcasted calls to survey 6 marshbird species during the early breeding season of the spring prior to herbicide application and again during 3 successive breeding seasons following herbicide application. We measured responses of individual species based on change in abundance from before to after herbicide application, and compared between wetland areas that received herbicide application and nearby similar wetland areas that did not (before-after, control-impact study design [Green 1979]). If marshbirds responded to changes in wetland conditions resulting from herbicide application, we expected to see measurable changes in marshbird abundance, although we anticipated the direction and magnitude of those changes would vary among species because these species have different habitat associations.

STUDY AREA

We conducted surveys for marshbirds at large, impounded wetlands near the eastern edge of the PPR in northwestern Minnesota (Fig. 3.1). This landscape has low relief and high water-holding capacity, resulting in large pooled basins with slow overland water flow and peat bog conditions (Ecoregion Level 3: 5.2.2 glaciated plains of ancient Lake Agassiz and 9.2.2 northern peatlands; Wiken et al. 2011). Efforts in the late 1800s and early 1900s to farm this area included ditching, peat removal, and other attempts to drain water more quickly from the landscape (Bourdaghs et al. 2015). Subsequent protection and restoration of wetland areas has resulted in large, sloped basins impounded by earthen embankments, with gated and managed water levels. Generally, these wetlands and surrounding areas are managed to control water movement through the landscape and provide habitat for wildlife. The altered hydrology of deep wetland basins provides conditions that favor invasion by cattail (Zedler and Kercher 2004), where it quickly becomes the dominant vegetation, resulting in large portions of surface area covered with floating mats of cattail (Wiltermuth and Anteau 2016). Land managers employ a variety of techniques to control cattail, including dredging, disking, mowing, prescribed burning, grazing, water level manipulation, and herbicide application (Beule 1979; Sojda and Solberg 1993; Elgersma et al. 2017; Bansal et al. 2019). Long-term control strategies often involve broad-scale application of herbicide at approximately 10-year intervals combined with mechanical control in shallow areas and during the period between herbicide treatments (Galatowitsch et al. 1999; Zedler 2000).

We examined wetlands in northwestern Minnesota experiencing operational management to control cattail. In 2015, the Minnesota Department of Natural Resources identified large cattail-dominated wetlands across several state-owned properties (Wildlife Management Areas; WMAs) for a large-scale herbicide application project. Managers delineated areas with the highest density of cattail at 8 WMAs (Beaches Lake, East Park, Eckvoll, Elm Lake, Pembina, Roseau River, Thief Lake, and Twin Lakes WMAs; Figs. 3.1 and 3.2) as priorities for herbicide application. In total, the projects resulted in glyphosate herbicide application (target rate of Rodeo® with surfactant at 7.02 liters/ha) to 1,179 ha of cattail mats via aerial sprayers on fixed-wing aircraft, and to another ~30 ha via ground application (backpack sprayers and from amphibious vehicles).

METHODS

We used a before-after, control-impact study design (Green 1979) to compare abundances of marshbirds at WMAs within the herbicide project area across 4 spring breeding seasons. We conducted surveys for marshbirds at wetlands that received herbicide application (treatment) and at paired wetlands that did not (control). We selected control sites that were similar to treatment sites in terms of vegetation composition, density, and interspersion. Where possible, we chose control sites within the same impounded wetland basin as the treatment sites. However, most similarly dense cattail-dominated wetlands within the same basin had also been targeted for herbicide application, and at some basins suitable control sites were unavailable; in those cases we chose alternative paired control sites in wetland basins adjacent to those with treatment sites. We surveyed 9 pairs of treatment and control sites (1 pair per WMA, except at Roseau River, which was large enough to encompass 2 pairs; Fig. 3.1; Table 3.1).

We established multiple survey locations within treatment and paired control sites and conducted call-broadcast surveys based on the Standardized North American Marsh Bird Monitoring Protocol (Conway 2011). We positioned survey locations >400 m apart to minimize repeated detections of individual marshbirds from multiple points. We located survey locations where observers could stand to detect marshbirds aurally and with a broad view of the wetland basin, near the wetland edge, often along an embankment or management access road. We established 28 survey locations at treatment sites and 25 survey locations at control sites across our 9 paired treatment-control sites (Fig. 3.1; Table 3.1).

We conducted surveys for marshbirds during the early spring breeding season before herbicide application (which occurred in fall 2015 and was coordinated by the Minnesota Department of Natural Resources) and during the 3 years following herbicide application. We conducted initial surveys in spring 2015 at all 9 WMAs. We repeated surveys in 2016 at all 9 WMAs and surveyed a subset of the WMAs in 2017 (n = 8) and 2018 (n = 6). We conducted 2 surveys at each WMA in 2015 and 2016 and conducted 1 survey at each WMA in 2017 and 2018 (see Appendix G for visit dates). We conducted surveys during crepuscular periods around sunrise (~0.5 hours before and up to 3 hours past sunrise) or around sunset (~3 hours before and ended ~0.5 hour after sunset). In years when we conducted 2 surveys, we surveyed each point twice: once during the period around sunrise and once during the period around sunset to account for diurnal variation in marshbird detectability. The same observer conducted surveys within paired sites of a WMA, and we conducted surveys at locations in the same order within individual WMAs.

Upon arriving at a survey location, the observer recorded environmental conditions and initial observations of all bird species; this first 3-4-minute period after arrival also served as a settling period intended to minimize the influence of the observer on marshbird behavior. The observer then conducted an 11-minute survey. The first 5 minutes were passive observation without broadcasting marshbird vocalizations. The later 6 minutes were divided into 1-minute intervals (30 seconds of broadcasted calls and 30 seconds with no broadcasted calls) during which we broadcasted calls of 6 secretive marshbird species in order: least bittern (*Ixobrychus exilis*), yellow rail (*Coturnicops noveboracensis*), sora (*Porzana carolina*), Virginia rail (*Rallus limicola*), American bittern (*Botaurus lentiginosus*), and pied-billed grebe (*Podilymbus podiceps*; Conway 2011; recommended species, order of broadcast, and standardized recorded calls obtained from national program organizer http://ag.arizona.edu/research/azfwru/NationalMarshBird/). We broadcasted recorded calls from a SanDisk Clip Sport mp3 player (SDMX24; SanDisk Corporation, Milpitas, CA) at 80-90 dB from 1 m away using a portable game speaker (Cass Creek Big Horn Remote Speaker, Cass Creek, Grawn, MI). The observer recorded all aural and visual detections of marshbirds, regardless of distance from the observer, and recorded the estimated location of the vocalizing marshbird using aerial photographs, a laser rangefinder, and compass. Recording estimated locations helped observers track multiple individuals calling during surveys to reduce double-counting and verify whether the individual was within the area that had experienced herbicide application. The protocol for this study was approved by the University of Minnesota Institutional Animal Care and Use Committee (IACUC protocol #1503-32456A).

To evaluate the response of marshbirds to herbicide application we measured change in marshbird abundance during 1 spring before to abundance during each of 3 springs after herbicide application and evaluated whether there was a difference in abundance between treatment and paired control sites within WMAs. We assumed that marshbird breeding phenology was similar over years, and therefore compared abundance from surveys conducted during similar timing in the breeding season (second surveys of 2016 and 2015, and single surveys of 2017 and 2018). We recorded counts of individuals detected during surveys at a particular site (treatment or control) as our estimate of abundance. However, because the number of survey locations within pairs of treatment and control sites were not equal at all WMAs, and we did not survey at all WMAs in all years, counts were not comparable for analyses across WMAs over years. Therefore, we normalized each year's counts at the WMA-level by calculating the difference as the mean count of marshbirds per survey location at control sites minus the mean count of marshbirds per survey location at treatment sites (paired by WMA; $n = 9$). The difference for 2015 represented the baseline of unique and naturally occurring spatial distribution of marshbirds between treatment and control sites within a WMA before herbicide application. We tested difference in mean counts from spring 2015 before herbicide application against each subsequent year (i.e., 2015 to 2016; 2015 to 2017; 2015 to 2018) using t-tests ($\alpha = 0.05$) to evaluate if change over the year interval between difference in mean counts at treatment and paired control sites was different from zero. Therefore, we assessed how marshbird abundance may have shifted, in direction and magnitude, from the baseline and interpreted change as response to treatment. If there was evidence of a change over years in the difference of marshbird abundance between treatment and control, then direction and magnitude of change indicated the nature of the response of marshbirds to herbicide application. We expected responses to be species-specific, but overall that marshbirds would respond positively to changes in wetlands resulting from cattail control via herbicide application.

RESULTS

During 2015 – 2018, we conducted surveys for marshbirds at 309 herbicide application treatment sites and paired control sites across 9 WMAs. Observers recorded 1,050 detections of 5 marshbird species (Appendix G); American bittern was the most commonly detected species (39.3% of detections across 63.8% of surveys), followed by sora (31.1%, 46.9% of surveys), pied-billed grebe (14.1%, 27.5% of surveys), Virginia rail (11.1%, 25.2% of surveys), and least bittern (4.2%, 12.0% of surveys), with no detections of yellow rails. Surveys conducted in spring 2015 before herbicide application indicated that marshbird abundance varied across treatment and control sites (i.e., not all pretreatment differences in average counts between treatment and control sites were centered at zero; Fig. 3.3).

We excluded marshbirds detected during the early season surveys in 2015 and 2016 in our analyses to make data from those years directly comparable to data collected in 2017 and 2018, which resulted in using 528 detections of 5 species on 200 surveys in our assessment of changes in marshbird abundances. There was no change in marshbird abundances ($P > 0.05$ for all species; Fig. 3.3) related to herbicide application during 2016, the first breeding season following herbicide application; i.e., there was no measurable change in the differences in mean counts by treatment. From 2015 to 2017 (from before herbicide application to the second breeding season after herbicide application), American bitterns showed change in abundances, where mean counts increased more at treatment sites than at control sites (mean of differences = 0.86, t = 2.46, 7 df, P = 0.04; Fig. 3.3), but no change was evident for the other 4 species (t = -0.68, 7 df, P = 0.64 for least bitterns; t = 0.13, 7 df, P = 0.90 pied-billed grebes; t = 2.11, 7 df, P = 0.07 soras; t = 0, 7 df, P = 1.00 Virginia rails; Fig. 3.3; Appendix H). From 2015 to 2018, the third spring after herbicide application, abundances changed with mean counts increasing more at treatment than control (Fig. 3.3) sites for American bitterns (mean of differences = 1.03, t = 2.77, 5 df, P = 0.04), soras (t = 3.68, 5 df, P = 0.01), and Virginia rails (t = 2.88, 5 df, P = 0.04); point estimates of change in abundance increased for pied-billed grebes, but this difference was not as strong as for the other species (t = 2.27, 5 df, P = 0.07). There was no evidence of a change in least bittern abundance (Fig. 3.3) across the 3-year study period (2015 – 2017).

DISCUSSION

The response of secretive marshbirds to control of invasive vegetation in Prairie Pothole Region wetlands is not well understood. We evaluated how secretive marshbird abundance responded to herbicide application to control cattail within impounded wetlands in northwestern Minnesota, United States. Initially, marshbirds seemed to decrease at herbicide-treated sites. However, after a time lag of >1 year following herbicide application, we observed an increase in abundance of 4 of 5 marshbird species in this system in response to herbicide application. American bitterns exhibited an increase in abundance at treatment sites the second breeding season following herbicide application, but for soras and Virginia rails, response was not evident until the third breeding season following treatment. Pied-billed grebes exhibited a similar, albeit non-statistically significant, positive trend of abundance related to herbicide application. We detected least bitterns relatively infrequently during our study, in part likely due to their relatively quiet calls and short detection distances (Benoît et al. 2009; Benoît et al. 2011), and there was no evidence of an effect on their abundance related to herbicide application. Even though they are known to be present on our study sites (Sidie-Slettedahl 2013), we detected no yellow rails during our surveys, perhaps because they prefer shallow wetlands and meadows dominated by sedges, and because they are most active outside of the periods during which we conducted surveys (Bart et al. 1984; Martin et al. 2014; Sidie-Slettedahl et al. 2015).

The spatial variation in abundance of marshbirds we observed during initial (prior to herbicide application) surveys likely reflects spatial variation in habitat quality. In general, marshbirds were less abundant at treatment sites than at control sites prior to treatment, perhaps indicating that dense cattail stands are poor-quality habitat for marshbirds in the systems we studied. Following treatment, abundance increased at treatment sites relative to control sites, indicating that herbicide application increased habitat quality at treatment sites for most target marshbird species. However, the response in abundance of marshbirds was not immediate, taking 2 – 3 years to become evident. The timing of this response by marshbirds likely reflects the timing of the response of cattail to herbicide application. Generally, immediately after herbicide application (fall 2015 in our study), above-water-level portions of plants that experience direct contact with herbicide begin to die, and herbicide gets translocated into roots and rhizomes of floating mats. The first spring after herbicide application vegetation structure and condition appears similar at both treatment and control sites—large swaths of dead residual vegetation from the previous growing season. However, at the emergence of new growth, areas treated with herbicide have diminished green vegetation density. Without renewed growth, residual vegetation decays over time and floating mats begin to disintegrate through wave and wind action (Sojda and Solberg 1993; Linz et al. 1994). The second season after treatment, at areas treated with herbicide, vegetation is both different from the first year following treatment and different from control sites—live cattail is less vigorous and residual stems of previous years' growth results in vegetation that has less structural complexity. The weight of snow and freezing during winter causes cattail mats to disintegrate and sink, creating more edges and interspersion of open water, resulting in higher structural heterogeneity that promotes higher plant species diversity. In the systems we studied, marshbirds appeared to respond to these changes in wetland vegetation conditions 2 – 3 years following herbicide application.

Our results suggest that marshbird abundance increases in response to control of cattail in impounded PPR wetlands. The immediate cause of this response seems to be changes in vegetation structure that increase habitat quality, such as decaying cattail mats breaking apart by wind and wave action, exposing more open water and mud flat areas where birds forage for prey. Lehikoinen et al. (2017) observed an increase in waterbird abundance on wetlands managed to reduce dense, homogeneous areas of emergent vegetation, though they did not specifically test herbicide application. A common result of varied management actions to reduce dense vegetation is increasing the amount of irregular patch edges and openings of exposed shallow water or mudflat for birds to forage, and allows space for native hydrophytic plant species to produce seed and harbor macroinvertebrates that comprise marshbird food. Food abundance and availability is also likely affected by herbicide application, but we do not know the extent of these effects in our system. It is also possible that changes in vegetation resulting from herbicide application affect either the behavior of marshbirds that make them more detectable by observers or increase the ability of observers to detect marshbirds, or both. We controlled for these effects to the extent possible by using a before-after, control-impact study design. However, evaluating the potential effects of food abundance and availability and factors that influence marshbird detection may provide further insight into the response of marshbirds to control of invasive vegetation.

Table 3.1. Marshbird survey locations in northwestern Minnesota

Summary of secretive marshbird surveys on large cattail-dominated wetlands on Minnesota Department of Natural Resources Wildlife Management Areas (WMAs) in northwestern Minnesota, U.S.A. that were targeted for a large-scale glyphosate herbicide application to control cattail during autumn 2015. We evaluated whether herbicide application affected marshbird abundance by conducting surveys during spring breeding seasons at paired treatment and control sites and evaluated change in number of detections from before to 3 years after herbicide application (2015 – 2018).

	Area of WMA (ha)	Treatment area (ha)	No. survey locations		Years surveyed
			Herbicide	Control	
Beaches Lake	12,393	62.3	2	2	2
East Park	4,220	122.0	2	4	4
Eckvoll	2,626	121.9	4	2	3
Elm Lake	6,370	370.1	4	4	4
Pembina	2,638	186.8	4	4	4
Roseau River east	30,418	58.2	4	4	4
Roseau River west		213.0	2	2	4
Thief Lake	22,241	30.4	3	3	4
Twin Lakes	3,591	45.1	3	4	3

Figure 3.1. Map of marshbird survey locations across northwestern Minnesota

Minnesota Department of Natural Resources (MN DNR) Wildlife Management Areas in northwestern Minnesota, U.S.A. with large, impounded wetlands invaded by dense cattail stands. We evaluated the effects of glyphosate herbicide application by the MN DNR to control dense cattail on secretive marshbirds. We visited herbicide treatment (red polygons) and control (not pictured; located in the same or adjacent basin to treatment) sites and compared abundance of marshbirds before and up to 3 years after herbicide application.

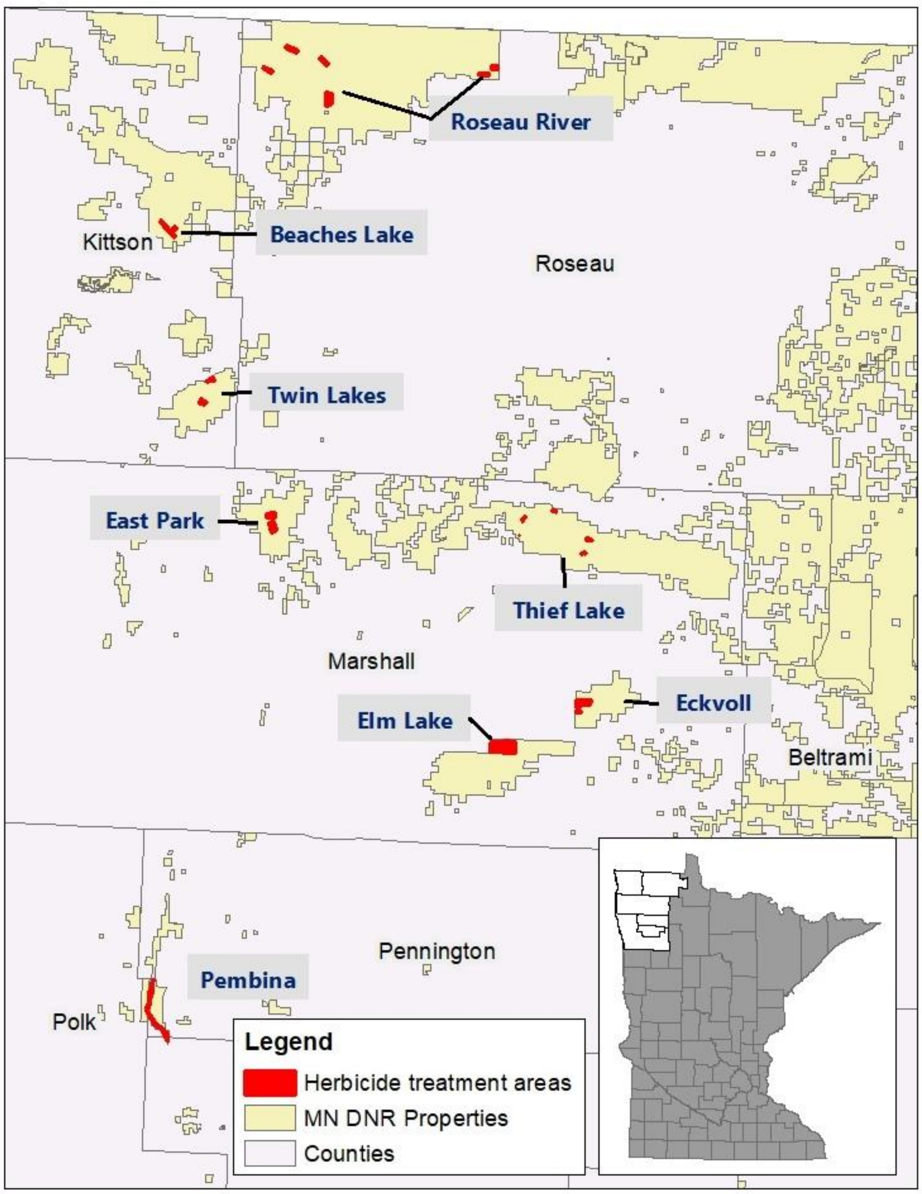


Figure 3.2. Map of herbicide and control areas at Elm Lake Wildlife Management Area

Map of Elm Lake Wildlife Management Area in northwestern Minnesota, U.S.A. indicating cattail treatment areas that received aerial glyphosate herbicide application in the fall of 2015, and marshbird survey point locations.

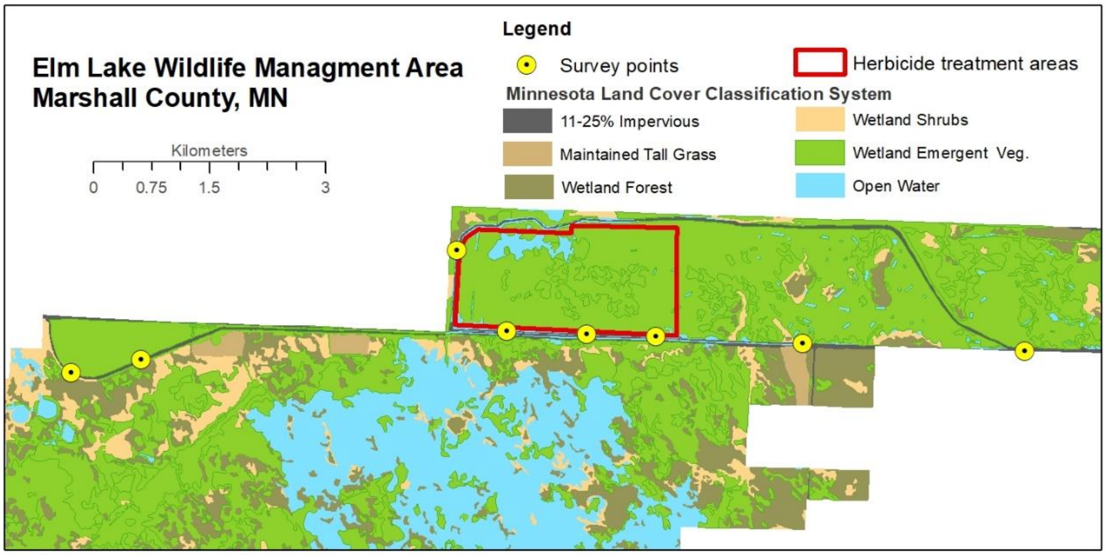
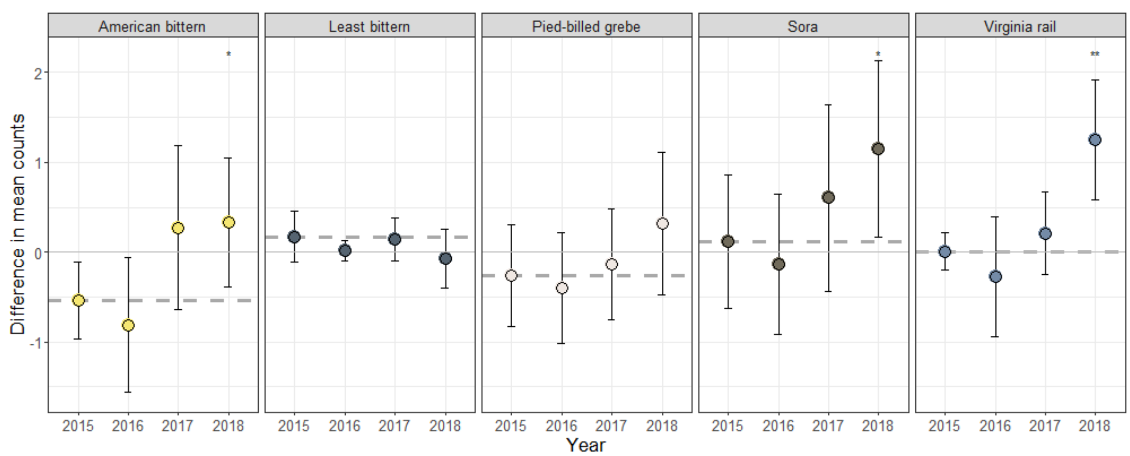


Figure 3.3. Comparing change in marshbird counts before and after herbicide

Differences within years 2015 to 2018 in mean marshbird counts (95% confidence interval) between treatment and control sites in northwestern Minnesota, U.S.A. for 5 species of secretive marshbirds. We used paired *t*-tests to compare mean differences between years from before (2015) to up to 3 years after herbicide application to control dense cattail (*Typha angustifolia* and *Typha* x *glauca*) and assess differences in marshbird abundance related to herbicide application. Baseline values of difference in mean counts before treatment in 2015 indicated by dashed line; subsequent years' values above dashed line indicate marshbird abundance increased at treated sites and/or decreased at control sites. Difference in mean abundance for 3 of the 5 species (American bitterns, soras, and Virginia rails) increased at treatment sites over years, although there was a lag in response. Pied-billed grebes showed a similar, although not statistically significant trend within the 3-year study period; we did not observe a change in least bittern abundance.



* statistically significant (paired *t*-test, $P = 0.05$) change in bird abundance.
** statistically significant (paired *t*-test, $P = 0.01$) change in bird abundance.

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APPENDICES

APPENDIX A. Detections of marshbirds in west-central Minnesota

Marshbirds detected (counts) during surveys in 2015 and 2016 on protected conservation lands managed by the U.S. Fish and Wildlife Service in west-central Minnesota, U.S.A. We conducted 2 surveys across all survey locations (*n*) following standardized protocol (Conway 2011) during the recommended period of early breeding season for this region in late May to early June.

Species	**2015**		**2016**		**Total**
	May 12 – 19 ***n*** **= 100**	**May 26 – June 17** ***n*** **= 115**	**May 2 – 9** ***n*** **= 126**	**May 20 – 27** ***n*** **= 126**	
American bittern	35	23	67	61	186
Least bittern	3	1	1	8	13
Pied-billed grebe	60	59	95	101	315
Sora	112	56	106	84	358
Virginia rail	27	30	48	52	157
Total	237	169	317	306	1,029

APPENDIX B. Wetland Selection

Distribution of wetland survey locations across spatially grouped routes in the Prairie Pothole Region of western Minnesota, U.S.A., tallied by management history category and wetland type. We defined management history categories based on the frequency of prescribed burning and grazing during 2000 – 2015. We surveyed marshbirds at type 2 and 3 (temporary and seasonal), and type 4 (semi-permanent) wetlands. We assessed marshbird abundance related to management history category (objective 1) at 113 survey locations and marshbird abundance related to wetland characteristics (objective 2) at the same 113 survey locations plus 15 additional survey locations that did not meet management history category criteria.

	Management history category							
	Low burn		**High burn**		**Burn & graze**			
Number and type of	**(≤ 1 Burn, 0 Graze)**		**(3 Burn, 0 Graze)**		**(≥ 3 Burn, ≥ 1 Graze)**			
management	**Wetland types**		**Wetland types**		**Wetland types**		***n* survey**	***n* additional**
Route Name	2,3	4	2,3	4	2,3	4	**locations for**	**survey locations**
	Count	Count	Count	Count	Count	Count	**objective 1**	**for objective 2**
Artichoke Lake	1	2	0	0	1*	0	4	3
Barry Lake	4	2	2	0	0	0	8	
Benson Lake	3	0	3	0	1	1	8	
Big Stone NWR	1	0	2	1	2	1	7	
Edwards	1	1	3*	3*	1	0	9	
Hegland	2	1	0	0	3*	0	6	1
Hillman	0	0	0	0	4*	0	4	4
Johnson	1	3	0	0	2	1	7	
Kufrin	2	0	2	1	2	0	7	
Mero	2	1	0	1	0	1*	5	2
Pedersen	4	1	4*	1*	0	0	10	
Prairie	1	2	2	1	0	0	6	
Redhead Marsh	1*	0	0	0	3*	3*	7	
Rothi	1	2	1	1	2	0	7	
Stenerson Lake	2	1	1	0	2*	2*	8	1
Twin Lakes	2	0	1*	0	2	0	5	2
Westport	2	2	1	0	0	0	5	2
Totals	30	18	22	9	25	9	113	15
	48		31		34		**Total = 128**	

*includes survey locations added in 2016

APPENDIX C. Land Cover Types Surrounding Survey Locations

Figure C1.

Area (ha) of land-cover types within 1 km of marshbird survey locations, grouped into spatially clustered routes, in the Prairie Pothole Region of western Minnesota, U.S.A. We surveyed locations during the marshbird breeding seasons in 2015 (n = 111) and 2016 (n = 127). We performed segmentation and supervised classification analyses of aerial imagery provided by the U.S. Fish and Wildlife Service using supplemental data from National Vegetation Classification Standard and 15-m-pixel raster land cover from Rampi et al. (2016). Bars indicate mean across all survey locations; points indicate means by Route.

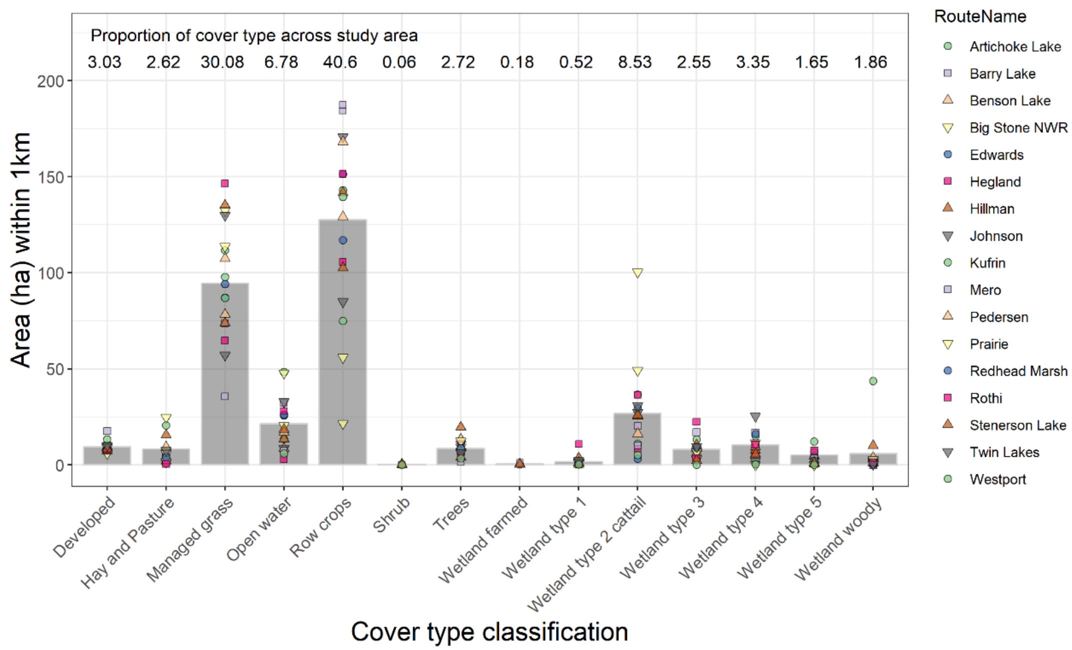
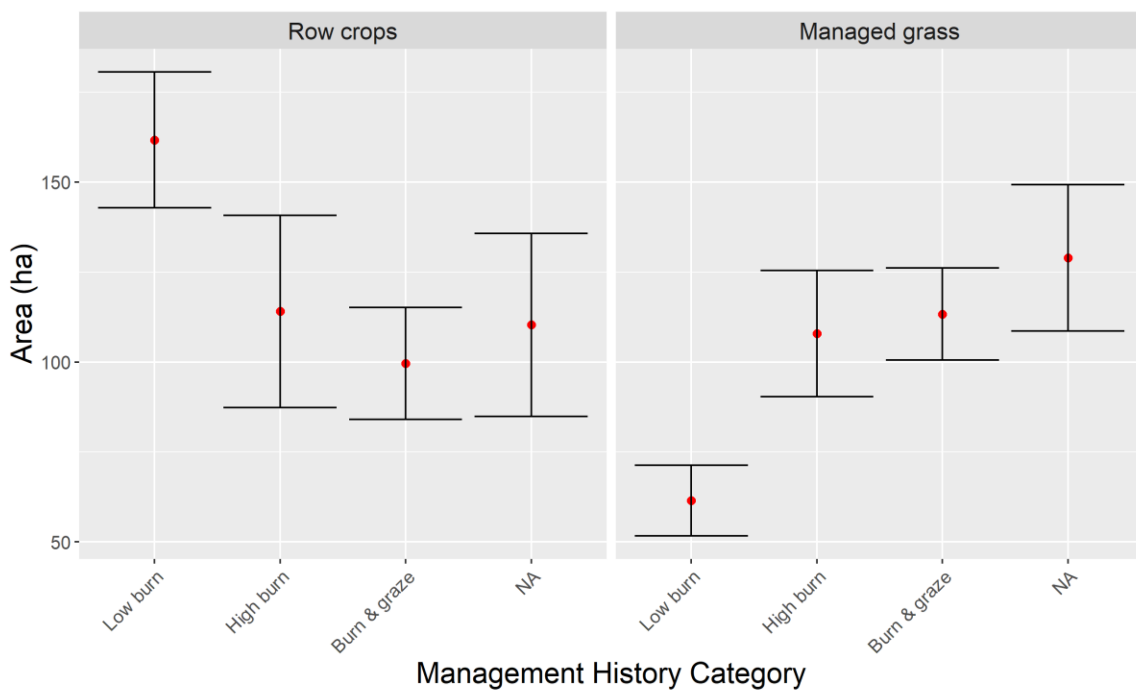


Figure C2.

Mean area (95% confidence interval) of row crops and managed-grass cover types within 1 km of survey locations across management history categories in Prairie Pothole Region of western Minnesota, U.S.A. Cover types derived using object-based image analysis, supervised classification, and overlay analysis. Areas of cover types (e.g., developed, open-water cover types) were similar across survey locations, except for differences of row crops and managed-grass cover types at survey locations that experienced low burn (sites with 1 prescribed burn during the previous 14 years; *n* = 47), high burn (sites with 3 prescribed burns during the previous 14 years; *n* = 31), and both burn & graze (sites with >3 prescribed burns and grazing during the previous 14 years; *n* = 33); we evaluated cover types surrounding additional wetland locations (*n* = 14) that did not match management history categories (*NA* category).



APPENDIX D. Principal Component Analysis of Wetland Characteristics

We assessed wetland characteristics at survey locations that experienced different management histories (Low burn [sites with 1 prescribed burn during the previous 14 years; *n* = 47], High burn [sites with 3 prescribed burns during the previous 14 years; *n* = 31], and both Burn & graze [sites with >3 prescribed burns and grazing during the previous 14 years; *n* = 33]), and at additional wetland locations that did not match management history categories (*n* = 14).

The covariates of wetland characteristics from remotely sensed and field-observed measures were highly correlated, so we assessed their similarity via Principal Component Analysis. We selected the covariates with the highest loading from each of the top Principal Components (PCs) that accounted for >60% of cumulative variation in the wetland characteristics datasets. The Principal Component Analysis revealed collinear variables within the remotely sensed dataset, and we selected covariates to consider in models of marshbird abundance based on PCs 1-8. The PCs 1, 5, 6, and 8 were highly correlated with variables representing the survey wetland basin. The PCs 2, 3, 4, and 7 were related to land cover within a 1-km radius of survey locations. In general, density of emergent vegetation was greatest in the tallest (1-1.5 m) section of the step-point pole, and we observed taller standing residual cattail at wetlands of Low burn and Burn & graze wetlands, but higher litter depth on the ground at High burn wetlands. Cattail was the dominant species (most frequently recorded species at visual obstruction in plots at survey location) at 73.8% survey location wetlands, and there were no differences across management history categories; other dominant plant species included reed canary grass and burreed (*Sparganium* sp.). We selected the single top-correlated variable with PCs 1-6 to consider in models of marshbird abundance. The PCs 1, 4, and 6 were related to mean of maximum height of emergent vegetation across plots (cumulative proportion of variance explained = 21.0%), standard deviation of the mean of cattail density (i.e., variation in mean number of cattail hits per plot at step-point measure; 52.53%), and standard deviation of maximum height of emergent vegetation (62.52%). The PC 2 was related to the gradient of deep water to dry (indicated by presence of litter and shorter vegetation height; 37.5%) and the top-correlated variable was mean water depth; PC 3 was related to the depth of litter (45.0%); and PC 5 was related to woody species present at the survey wetland.

From remotely sensed wetland characteristics only 3 of 8 variables were included in best-supported models of marshbird abundance: presence of scrub/shrub wetland type, index representing complexity of wetland shape (edge length m / sqrt[area ha]), and sum area of wetland cover within 1 km. From the field-observed wetland characteristics, 3 of the 6 variables were included in best-supported models: mean water depth, proportion of plots with woody species present, and variance in height of emergent vegetation. Best-supported models that considered field-observed covariates added to best-supported models based on remotely sensed covariates, resulted in models with both remotely sensed and field-observed covariates for pied-billed grebes and soras, but not for American bitterns and Virginia rails.

Table D1.

Principal component analysis of remotely sensed and field-observed wetland characteristics. We selected covariates that accounted for >60% of cumulative variance explained to consider in models of marshbird abundance.

Remotely sensed covariate	Principal component	Proportion of variance explained	Cumulative proportion of variance explained	Standard deviation
Log area of survey wetland	1	18.34%	18.34%	3.90
All wetland types sum area 1 km	2	12.62%	30.96%	3.24
Wetland type 1 proportion in 1 km	3	6.43%	37.39%	2.31
Hay and pasture proportion in 1 km	4	6.05%	43.44%	2.24
Shallow marsh proportion of survey wetland	5	5.18%	48.62%	2.07
Temp. flooded proportion of survey wetland	6	4.83%	53.44%	2.00
Ratio of open water to emergent vegetation	7	4.37%	57.82%	1.90
Scrub/shrub present in survey wetland	8	3.68%	61.50%	1.75

Field-observed covariate	Principal component	Proportion of variance explained	Cumulative proportion of variance explained	Standard deviation
Maximum vegetation height (m) mean	1	21.39%	21.39%	3.24
Litter depth (m)	2	16.74%	38.13%	2.86
Cattail density (mean touches per plot)	3	7.69%	45.82%	1.94
Water depth (m)	4	6.72%	52.53%	1.81
Woody species present	5	5.14%	57.67%	1.58
Maximum vegetation height (m) variance	6	4.85%	62.52%	1.54

APPENDIX E. Data Groups for Cross Validation

For cross validation of models of marshbird abundance related to wetland characteristics, we grouped the datasets based on survey locations for which we had associated, within-year marshbird densities and measures of wetland characteristics, and whether we resampled those measures (Table 2). The first group was the most complete as it included paired within-year marshbird and wetland measures for both years (2015 and 2016) at survey locations. The second group included paired within-year marshbird and wetland measures for only 1 year at survey locations (i.e., paired 2015 measures of survey locations which were [sampled for marshbirds in 2016 but] not resampled for wetland characteristics in 2016, and the paired 2016 measures of newly established survey locations in 2016 [i.e., which lacked resampled 2015 wetland measures]). The third group was the least robust as it was unpaired 2016 marshbird measures (i.e., from survey locations of the second group that were resampled in 2016 for marshbirds but for which we did not repeat measurement of wetland covariates). We randomly selected paired data from the first and second groups for model building; we lumped randomly selected 1 year of paired measures from the first group of survey locations (n = 42) with randomly selected 2/3 of paired measures from the second group (n = 57). We did not include marshbird measures from the third group in model building because they did not have paired wetland measures. We used the reserved data of the 3 groups in separate series of cross validation model testing: the other-year paired measures of the first group (n = 42), 1/3 of the second group (n = 28), and the unpaired 2016 marshbird measures paired with the previous year's (2015) wetland measures (n = 69).

Table E1.

The number of survey locations for which we split data for cross validation, based on where we estimated marshbird density and measured field-observed wetland characteristics for 2015 and 2016. Group 1 was the most robust with both estimates of marshbird density and measures of wetland covariates within the same year, for both years. Group 2 had data points for estimates of marshbird density and measures of wetland covariates within the same year, for only 1 year. Group 3 had data points for estimates of marshbird density but lacked measures of wetland covariates within the same year; we did not use Group 3 in model building, but used this group in cross validation by pairing with measures of wetland covariates at the survey location in a different year. We split the data into 3 groups and split Groups 1 and 2 again into Build and Test subsets. We pooled the 2 Build subsets together (n = 99) to derive models of marshbird density, and then evaluated models independently in cross validation with Test 1, Test 2, and Test 3 subsets.

		(data collection types at no. survey locations 2015)		(data collection types at no. survey locations 2016)					
	No. survey locations in study area	Marshbird densities	Wetland characteristics	Marshbird densities	Wetland characteristics	Total survey locations-year datapoints	**Subset for model building**	**Reserved for model testing**	
***n* survey locations with marshbird + wetland measures in same year for 2 years**	42	42	42	42	42	84	42	42	**“Test 1”**
***n* survey locations with marshbird + wetland measures in same year for 2015**	69	69	69	69*	0	85	57	28	**“Test 2”**
***n* survey locations with marshbird + wetland measures in same year for 2016**	16	0	0	16	16				
	leftover unpaired marshbird measures with substituted paired wetland measures from previous year:					69*	0	69*	**“Test 3”**
Total	127	111	111	127	58	238	99		

APPENDIX F. Model performance in cross validation

Best-supported models of marshbird density based on wetland characteristics (models considering remotely sensed covariates versus field-observed covariates versus best-supported models considering remotely sensed covariates and adding field-observed covariates). We compared across model types in 3 rounds of cross validation analyses using 3 subsets of reserved data. Lower root mean squared error (RMSE) indicated better model performance.

	American Bittern			Pied-billed grebe			Sora			Virginia rail		
Model	**Remote**	**Field**	**Both**	**Remote**	**Field**	**Both**	**Remote**	**Field**	**Both**	**Remote**	**Field**	**Both**
Intercept	0.222	0.472	0.472	-0.009	0.013	-0.124	3.77	2.332	4.839	0.752	0.884	0.752
(Standard Error)	-0.069	-0.134	-0.134	-0.07	-0.042	-0.078	-0.937	-0.406	-1.165	-0.246	-0.239	-0.246
AIC	207.5	206.2	206.2	51.7	47.4	45.1	548.7	553.7	548.3	448	449.6	448
Adjusted R^2	--	0.03	0.03	0.01	0.06	0.09	0.07	--	0.08	0.03	--	0.03
RMSE test 1	0.817	0.825	0.825	0.285	0.278	0.280	5.235	5.181	5.250	2.352	2.291	2.352
RMSE test 2	0.408	0.441	0.441	0.169	0.174	0.165	2.205	2.341	2.335	3.232	3.179	3.232
RMSE test 3	0.695	0.698	0.698	0.760	0.755	0.761	5.441	5.495	5.355	2.150	2.100	2.150

APPENDIX G. Marshbird detection counts in Northwestern Minnesota

Counts of 5 species of marshbirds detected at herbicide-treated and non-treated control sites. We conducted surveys during late May – early June, with 2015 surveys establishing the baseline marshbird abundance (i.e., before herbicide treatment), from which we assessed changes in abundance in subsequent years. We visited survey locations 2 times in 2015 and 2016, and 1 time in 2017 and 2018. Timing of surveys for this area were based on recommendations of local biologists and the National Marshbird Survey Monitoring Program (http://ag.arizona.edu/research/azfwru/NationalMarshBird) to coincide with the marshbird breeding season across northwestern Minnesota, U.S.A.

		2015		2016		2017	2018	
	Visit date	20 – 26 May	8 – 13 June	15 – 17 May	26 May – 7 June	30 May – 3 June	28 May – 4 June	Total
American bittern	Herbicide	61	29	67	55	22	8	413
	Control	41	13	44	31	29	13	
Least bittern	Herbicide	3	4	0	4	3	2	45
	Control	7	11	2	2	6	1	
Pied-billed grebe	Herbicide	20	11	26	18	13	4	148
	Control	14	6	12	7	7	10	
Sora	Herbicide	35	24	49	16	15	2	327
	Control	45	21	57	9	31	23	
Virginia rail	Herbicide	8	7	17	8	9	1	117
	Control	5	12	9	4	13	24	
Total	Herbicide	127	75	159	101	62	17	1,050
	Control	112	63	124	53	86	71	

APPENDIX H. Change in marshbird abundance from before to 3 years after herbicide application

Mean differences in marshbird abundance between paired reference and treatment (application of glyphosate to control cattail [*Typha* spp.]) wetlands in northwestern Minnesota, U.S.A. and comparisons of abundance from the spring in the year (2015; *n* = 9) prior to glyphosate herbicide application to spring in each year after (2016 [*n* = 9], 2017 [*n* = 8], 2018 [*n* = 6]). Positive values within a year indicate higher abundance at treatment wetlands. Positive values in comparisons with 2015 indicate a greater increase in abundance at treatment wetlands.

	Mean difference between treatments within a year					**Change in mean difference from before treatment (2015) to after**		
Species	2015	2016	2017	2018		1st year	2nd year	3rd year
American bittern	-0.54	-0.81	0.27	0.33	change	-0.27	0.88	1.09
					95% CI	-0.99 - 0.44	-0.22 - 1.97	0.11 - 2.08
					t (*P*-value)	-0.88 (0.41)	1.89 (0.14)	2.85 (0.04)
Least bittern	0.17	0.01	0.15	-0.07	change	-0.16	-0.08	-0.37
					95% CI	-0.46 - 0.15	-0.47 - 0.31	-0.87 - 0.14
					t (*P*-value)	-1.18 (0.27)	-0.47 (0.65)	-1.88 (0.12)
Pied-billed grebe	-0.26	-0.40	-0.14	0.32	change	-0.14	0.16	0.57
					95% CI	-0.94 - 0.67	-0.83 - 1.15	-0.17 - 1.3
					t (*P*-value)	-0.39 (0.71)	0.38 (0.72)	1.97 (0.11)
Sora	0.12	-0.14	0.60	1.15	change	-0.25	0.32	0.89
					95% CI	-1.15 - 0.64	-0.98 - 1.62	0.11 - 1.67
					t (*P*-value)	-0.66 (0.53)	0.58 (0.58)	2.94 (0.03)
Virginia rail	0.01	-0.27	0.21	1.25	change	-0.28	0.17	1.15
					95% CI	-0.9 - 0.34	-0.44 - 0.77	0.47 - 1.83
					t (*P*-value)	-1.05 (0.32)	0.65 (0.53)	4.35 (0.01)