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RH: Hill et al. ∙ Marshbirds and Invasive Cattail Control

**Marshbird Response to Herbicide Control of Cattail in Northwestern Minnesota**

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**ABSTRACT** Wetlands provide essential habitat for a wide variety of wildlife species. In the once wetland-rich Prairie Pothole Region and adjacent areas of central North America, many wetlands have been converted to agricultural production. Many remaining wetlands experience ecological change via the invasion and spread of non-native plant species, such as non-native (*Typha angustifolia*) and hybrid cattail (*Typha* x *glauca*), which spread aggressively and displace native vegetation, especially in large, impounded wetlands. Management of wetlands in these landscapes often includes broad-scale herbicide application intended to break up mats of cattail and restore areas to more wildlife-friendly 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 effects of cattail management on wetland wildlife, we conducted standardized call-broadcast surveys for 5 species of marshbirds at 9 study sites that included survey locations associated with treatment (herbicide application) and control (no herbicide application) sites in wetland impoundments in northwestern Minnesota, USA, using a before-after, control-impact study design. We surveyed American bitterns (*Botaurus lentiginosus*), least bitterns (*Ixobrychus exilis*), pied-billed grebes (*Podilymbus podiceps*), soras (*Porzana carolina*), and Virginia rails (*Rallus limicola*) during the breeding season prior to herbicide application (late summer and early autumn of 2015) and during the 3 breeding seasons after herbicide application (2016 – 2018). We modeled species counts using a generalized linear mixed model with year-by-treatment interactions as fixed effects and site as a random effect. Before herbicide application, expected mean counts did not differ between treatment and control survey locations. Three years post-treatment, we detected significant increases in expected mean counts at treatment compared to control survey locations for soras (*t*193 = 3.373, *P* = 0.020) and Virginia rails (*t*193 = 3.167, *P* = 0.037), and point estimates for all species except least bittern were higher at treatment survey locations. Overall, our results suggest that these marshbird species responded positively to herbicide control of invasive cattail and that breeding marshbirds in these and similar wetland systems may experience positive population response over a period of at least 3 years following treatment.

**KEY WORDS** American bittern, *Botaurus lentiginosus,* cattail, herbicide, invasive plants, marshbirds, pied-billed grebe, *Podilymbus podiceps,* *Porzana carolina,* Prairie Pothole Region, *Rallus limicola,* sora, Virginia rail, wetlands

The Prairie Pothole Region (PPR) and adjacent areas of central North America were once a vast complex of prairie grasslands and glaciated wetlands and lakes (Dahl 2014). Beginning in the late 19th and early 20th centuries, conversion of wetlands for agricultural uses and other purposes eliminated extensive areas of wetlands in the PPR. Drainage and other changes to hydrology 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, the 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 many wildlife species, although it is not well understood how some wildlife respond to altered characteristics of wetlands dominated by invasive vegetation or to efforts to restore native vegetation communities (National Research Council 1992, Ratti et al. 2001, Pulfer et al. 2014).

Land managers often aim to restore wetlands to conditions that existed prior to becoming dominated by invasive vegetation (e.g., Vacek and Friske 2012, Minnesota Prairie Plan Working Group 2018), 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 and altered wetlands in the PPR and adjacent landscapes often focuses on manipulating cattail (*Typha* spp.; Zedler 2000) via herbicide application or mechanical removal. Vegetation communities in many PPR wetlands have shifted from marshes with heterogeneous native species including broadleaf cattail (*Typha latifolia*), into dense, monotypic 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 reduced plant diversity and structural heterogeneity, and altered hydrology from sedimentation, 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). On large, impounded wetlands, herbicide application breaks up floating beds of cattail (Linz and Homan 2011), 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 plant species (Linz et al. 1994, Galatowitsch 2006, Linz and Homan 2011). These conditions are believed to be more favorable for breeding marshbirds (Bolenbaugh et al. 2011); however, few studies (e.g., Linz et al. 1994, Linz and Blixt 1997, Linz and Homan 2011, Anderson et al 2019) have investigated effects on wildlife from widespread herbicide application to control invasive wetland vegetation, and the response of breeding marshbird populations to changes that result from reducing large patches of cattail is unknown. In general, there is almost no published information on breeding marshbird response to herbicide application to control invasive cattail in PPR wetlands dominated by invasive cattail.

In general, breeding marshbirds prefer habitat with patches of emergent vegetation interspersed with open water or mudflats (e.g., Lor and Malecki 2006), high edge-to-interior ratio (e.g., Chabot et al. 2014), and plant communities with a range of canopy height and density (Johnson and Dinsmore 1986) that are generally free of woody vegetation (Bolenbaugh et al. 2011, Harms and Dinsmore 2013). However, marshbird-habitat relations in the context of wetland vegetation in the PRR are poorly documented (but see Fairbairn and Dinsmore 2001, Orr et al. 2020). Most marshbird 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, Fairbairn and Dinsmore 2001, Zimmerman et al. 2002, Orr et al. 2020). If treating areas dominated by cattail with herbicide restores these conditions, then breeding marshbird use and abundance may increase in wetlands following herbicide application (as reported for black terns [*Chlidonias niger*] by Linz and Blixt 1997).

To assess how herbicide application to control invasive cattail influences marshbird populations, we assessed an index of marshbird abundance (standardized counts) at large, impounded wetlands in northwestern Minnesota, USA, prior to and for 3 years following operational herbicide application to control invasive cattail. Our specific objectives were to (1) estimate responses of individual marshbird species based on change in counts (an index of 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]) and (2) assess the timing and magnitude of any observed responses. If marshbirds responded to changes in wetland conditions resulting from herbicide application, we expected to see changes in marshbird counts, although we anticipated the direction, magnitude, and timing of those changes may vary among species..

**Study Area**

We conducted surveys for breeding marshbirds at large (> 30 ha), impounded wetlands near the eastern edge of the PPR in northwestern Minnesota (Fig. 1), USA. This landscape has low relief and high water-holding capacity, resulting in large (>30 ha) 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). Climate in northwestern Minnesota is classified as warm-summer humid continental with mean annual precipitation from 51–56 cm, with a small proportion of the total coming as snow (MNDNR 2015). Extreme minimum temperatures are -40 – -43⸰C and extreme maximum temperatures are near 35⸰C (MNDNR 2015)

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 gated earthen embankments, and managed water levels. Generally, these wetlands and surrounding areas are managed to control water movement through the landscape and provide habitat for wildlife. However, the altered hydrology of deep wetland basins provides conditions that favor invasion by cattail (Zedler and Kercher 2004), which quickly becomes the dominant vegetation, resulting in large portions of surface area covered with floating mats (i.e., not rooted in the substrate) of cattail (Wiltermuth and Anteau 2016), with little open water. Land managers employ a variety of techniques to control cattail, including dredging, disking, mowing, 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 on wildlife management areas (WMAs) in northwestern Minnesota, USA that were managed to control invasive cattail by the Minnesota Department of Natural Resources (MNDNR). Wildlife management areas in this portion of Minnesota often include impounded wetlands, which are primarily managed by manipulating water levels to promote desirable vegetation (e.g., wild rice [*Zizania palustris*]) and wetland conditions that support breeding and migrating waterfowl and other birds. In 2015, the MNDNR implemented a program of herbicide application to control cattail in impounded wetlands in WMAs intended to return wetland conditions to those resembling a hemi-marsh (i.e., a mix of open water and emergent vegetation). 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; Table 1, Figs. 1 and 2; elevation = ~285 –355 m amsl) as priorities for herbicide application. Herbicide application (target rate of Rodeo® [with glyphosate as active ingredient] with surfactant at 3.77 liters/ha [maximum rate of 7.02 liters/ha] applied as a solution with a minimum volume of ~47 liters/ha) occurred via contract with third-party vendors in late summer and early autumn (5 August–6 September 2015 to 1,179 ha of cattail mats via aerial sprayers (on fixed-wing aircraft), and to another ~30 ha via ground application (via backpack sprayers and from amphibious vehicles).

**Methods**

We used a before-after, control-impact study design (Green 1979) to compare counts of marshbirds at WMAs within the herbicide program area during 4 spring breeding seasons, including the spring before herbicide treatment and 3 subsequent springs after treatment. We conducted surveys for marshbirds at wetlands that received herbicide application (treatment) and at paired wetlands that did not (control), and considered these pairs to be study sites. At each study site, we selected survey locations (locations where we conducted marshbird surveys, see below) on the edges of control (i.e., where no treatment was applied) and treatment (i.e., where herbicide was applied) wetlands that were similar in terms of initial vegetation composition, density, and interspersion (i.e., all survey locations were adjacent to monotypic invasive cattail). Where possible, we chose control survey locations within the same impounded wetland basin as the treatment survey locations. However, most similarly dense cattail-dominated wetlands within the same basin may also have been targeted for herbicide application, and at some basins suitable control survey locations were unavailable; in those cases, we chose alternative control survey locations in wetland basins adjacent to those with treatment survey locations. We conducted marshbird survey at 9 study sites that included paired treatment and control survey locations (1 study site per WMA, except at Roseau River, which was large enough to encompass 2 study sites; Fig. 1, Table 1). At 4 study sites, treatment and control survey locations were within the same basin (Fig. 2) and at 5 study sites, nearest treatment and control survey locations were separated by 1.1–5.7 km in a different wetland basin.

We established multiple treatment and control survey locations (Table 1) within study 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 the same marshbirds from multiple survey locations. To facilitate effective sampling and minimize potential influence of surveys on marshbird behavior, we placed survey locations where observers could stand to detect marshbirds aurally and visually, and with a broad view of the wetland basin, near the wetland edge, often along an embankment or management access road. We therefore sampled the edges of large, impounded wetlands at all survey locations. We established 28 treatment and 28 control survey locations across our 9 study sites (example in Fig. 2, Table 1). Wetland vegetation characteristics were similar among survey locations, with each adjacent wetland being dominated by invasive monotypic cattail (Fig. S1).

We conducted surveys for marshbirds during the early-spring breeding season (late May to mid-June; Table 2) before herbicide application (which occurred in late summer and early autumn 2015 and was coordinated by the MNDNR) and during the 3 springs following herbicide application. We conducted initial surveys in spring 2015 at all 9 WMA study sites (Table 1). We repeated surveys in 2016 at all 9 study sites and surveyed a subset of study sites in 2017 (*n* = 8) and 2018 (*n* = 6). We conducted surveys during crepuscular periods around sunrise (~0.5 hour before and up to 3 hours past sunrise) or around sunset (~3 hours before until ~0.5 hour after sunset) to incorporate diurnal variation in marshbird detectability. The same observer conducted surveys within paired sites at a WMA, and we conducted surveys at locations in the same order within individual sites. In 2015 and 2016, 2 observers (NMH and a field technician; a different field technician in each of those years) conducted surveys, and in 2017 and 2018, NMH conducted all surveys.

Upon arriving at a survey location, the observer recorded environmental conditions (e.g., ambient temperature, wind speed and direction, cloud cover; to confirm that conditions were within protocol parameters [Conway 2011]) and initial observations of all bird species; this first 3- to 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 involved passive observation without broadcasting marshbird vocalizations. The later 6 minutes were divided into 1-minute intervals (30 seconds of broadcast calls and 30 seconds with no broadcast calls) during which we broadcast calls of 6 marshbird species in order (Conway 2011): 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*). The recommended species, order of broadcast, and standardized recorded calls were obtained from the North American Marsh Bird Monitoring Program organizer (http://ag.arizona.edu/research/azfwru/NationalMarshBird/). We broadcast 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 in the target wetlands (i.e., the wetland in front of the observer when the observer was facing the treated or control wetland), regardless of distance from the observer, and recorded the estimated location of each detected (either visually or aurally) marshbird using printed aerial photographs that included indications of distances out to 400 m, a laser rangefinder, and compass. Recording estimated locations on printed aerial photographs helped observers track multiple individuals calling during surveys to reduce double-counting and verify that the individual was within the target wetland. Final data for analysis included only birds within 400 m from survey locations to minimize error associated with estimating distance beyond 400 m and to avoid including the same marshbirds in data for >1 survey location. The University of Minnesota Institutional Animal Care and Use Committee (IACUC protocol #1503-32456A) approved the protocol for this study.

To evaluate the response of marshbirds to herbicide application, we compared the expected mean marshbird counts at treatment versus control survey locations each year with a generalized linear mixed model (GLMM) with a Poisson distribution (R package glmmTMB, Brooks et al. 2017). We analyzed each marshbird species individually with study sites included as a random effect and with a treatment-by-year interaction (O’Donnell et al. 2015), considering only marshbirds detected during the 11-mintue survey period. Because we expected vegetation to respond to herbicide treatment in ways that could affect probability of detecting marshbirds visually (i.e., by decreasing vegetation density and increasing extent of open water) in the years following the year of treatment, we repeated our analyses using only aural detections and compared those results to results using both aural and visual detections to assess whether probability of visual detection changing through time. We evaluated significance with GLMM *t*-ratios and *P*-values with a Tukey-adjustment method based on contrasts on the log-scale and log-transformed expected mean counts and 90% confidence intervals for plotting using the emmeans package (Lenth 2022).

Our study design did not include measuring vegetation or other wetland metrics (e.g., extent of open water). Instead, *post hoc*, we assessed whether there was an evident change in vegetation within treated polygons (i.e., the areas of wetlands treated with herbicide) following herbicide application using remotely derived indices of greenness (Normalized Difference Vegetation Index [NVDI]) and water reflectance (Normalized Difference Water Index [NDWI]) from the period 2010–2020. To assess potential differences in land-cover-type composition between treatment and control survey locations prior to herbicide application, we used the Minnesota Land Cover Classification and Impervious Surface Area by Landsat and Lidar (2013 update - Version 2; <https://gisdata.mn.gov/dataset/base-landcover-minnesota>) to describe land cover within 400 m of survey locations in 2013 (2 years prior to the first year of our study).

**Results**

During 2015 – 2018, we conducted 202 surveys for marshbirds at herbicide application treatment survey locations at 8 WMAs (9 study sites). Observers recorded 524 detections of 5 marshbird species (Table 2). American bittern was the most commonly detected species (199 total observations), followed by sora (141), Virginia rail (78), pied-billed grebe (73), and least bittern (33), with no detections of yellow rails. We did not detect any differences in expected mean counts at treatment versus control survey locaitons during spring 2015 surveys (the spring prior to herbicide application) for any species (Table 2, Fig. 3).

After herbicide application, the expected mean counts varied among marshbird species and by year (Table 2, Fig. 3). In 2016, the first year after herbicide application, we detected no statistically significant difference in expected mean counts for any marshbird species (Table 2), although point estimates of expected mean counts were slightly lower at treatment survey locations for all 5 species (Fig. 3). There were no significant differences in expected mean counts for any marshbird species in 2017 (the second spring following herbicide application). In the third spring following herbicide application (2018), we observed higher expected mean counts at treatment versus control survey locations for soras (*t*193 = 3.373, *P* = 0.020) and Virginia rails (*t*193 = 3.167, *P* = 0.037; Table 2, Fig. 3), and point estimates of expected mean counts were higher for all marshbird species except least bitterns (Fig. 3). Patterns in expected mean counts from models using only aural detections were similar to those from models using both aural and visual detections.

Although we did not quantify changes in vegetation resulting from herbicide application as part of our study design, our observations of vegetation response were consistent with those described in the literature. The first spring after herbicide application, vegetation structure and condition were similar at both areas treated with herbicide and areas not treated with herbicide—large swaths of dead residual vegetation from the previous growing season. However, with the emergence of new growth, areas treated with herbicide had far less green vegetation density than did areas without herbicide application (see Fig. S2). Without renewed growth, residual vegetation in areas treated with herbicide decayed 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 differed both from the first year following treatment and from areas not treated with herbicide—live cattail was less vigorous and decay of residual stems of previous years’ growth resulted in vegetation with less structural complexity (see Fig. S3). Both NVDI and NDWI indicated a strong response following herbicide application (Fig. S4), with a decrease in NVDI beginning in 2016 and extending for several years and a similar decrease in NDWI, followed by an increase in NDWI beginning ~2–3 years following herbicide application. Land-cover-type composition surrounding survey locations was similar between treatment and control sites prior to the period during which we conducted our study (Fig. S5).

**Discussion**

Based on expected mean counts at paired treatment and control survey locations, none of the marshbird species (American bitterns, soras, Virginia rails, pied-billed grebes, and least bitterns) that we monitored exhibited a population-level response to herbicide application the spring following application to impounded PPR wetlands dominated by invasive cattail in northwestern Minnesota, USA. Of note, however, is that point estimates of expected mean counts of all 5 species were lower the spring following herbicide application, indicating a small, consistent decline in abundance among these species from the previous year (Fig. 3). There was no apparent population-level response for any of the 5 species 2 years following herbicide application, based on comparison of expected mean counts the spring prior to herbicide application. Three springs following herbicide application, soras and Virginia rails exhibited increased expected mean counts and point estimates of expected mean counts of 4 of the 5 marshbird species (all except least bittern) were higher at areas treated versus not treated with herbicide, indicating that herbicide application likely increased habitat quality for most of the marshbird species we monitored 3 years following herbicide application to control invasive cattail. Furthermore, although there appeared to be a slight decrease in marshbird abundance in areas treated with herbicide the first year after herbicide application, any potential negative effect of herbicide application on marshbirds appeared to be short-lived (Fig. 3). We note that our assessment of least bitterns, the only species that did not appear to exhibit an increase in abundance (indexed by counts), was likely constrained by small sample sizes resulting from their relatively quiet calls and consequent short detection distances (Benoît et al. 2009, Benoît et al. 2011). And, even though they were known to have previously been present in our study area (Sidie-Slettedahl 2013), we detected no yellow rails during our surveys, probably because they are scarce in the area, prefer shallow wetlands and meadows dominated by sedges (i.e., not similar to the wetlands we studied), and are most active at night (Bart et al. 1984, Martin et al. 2014, Sidie-Slettedahl et al. 2015), outside the periods we conducted surveys.

Because we observed no difference in expected mean counts of marshbird species we monitored in the breeding season prior to herbicide application between areas targeted for herbicide applicationand areas not targeted for herbicide application, our control survey locations likely served as an appropriate reference to evaluate the treatment. Furthermore, initial land-cover-type composition was similar between treatment and control survey locations prior to 2015, the first year we conducted surveys and prior to herbicide application (Fig. S5). However, we note that following herbicide application, changes in vegetation resulting from herbicide application could affect either marshbird behavior, which could make them more detectable by observers or increase the ability of observers to detect marshbirds (i.e., by increasing visibility resulting from changes in vegetation), or both. Because our models using only aural versus both aural and visual detections produced similar results, we did not find evidence for an influence of changes in visual detection probability on patterns in expected mean counts of marshbirds.

Our study design did not allow us to elucidate the mechanism(s) that resulted in increased expected mean counts of marshbirds. Although monitoring vegetation was not a component of our study, we posit that the timing of the marshbird response we observed likely reflected the timing and nature of the response of cattail to herbicide application. Generally, immediately after herbicide application, above-water-level portions of plants that experience direct contact with herbicide begin to die, and herbicide is translocated into roots and rhizomes of floating mats of vegetation and of vegetation rooted in the substrate. The first spring after herbicide application and during the period when marshbirds return to breeding locations in the PPR, vegetation structure and condition appear similar at both areas that received and did not experience herbicide application—large swaths of dead residual vegetation (either from plant senescence or the effects of herbicide) from the previous growing season. Later in spring, at the emergence of new growth, areas treated with herbicide have diminished green vegetation density and over time, residual vegetation decays 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 differs both from the first year following treatment and from areas not treated with herbicide—live cattail is less vigorous and decay of residual vegetation from the previous years’ growth results in vegetation with 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 and higher plant species diversity (Lishawa et al. 2015). We suspect that these changes in wetland vegetation in our study increased habitat quality for breeding marshbirds, especially for rails (Fairbairn and Dinsmore 2001, Orr et al. 2020).

Breeding marshbird expected mean counts in our study increased in response to control of cattail in impounded PPR wetlands, similar to increases in black tern abundance following chemical control of cattails reported by Linz et al. (1994) and Linz and Glixt (1997). Linz et al. (1994) observed in increase in black tern abundance the year following herbicide application to control monotypic cattail in PPR wetlands in North Dakota, USA in 1 of 2 experiments, and that response persisted for the 2 years they monitored wetlands following treatment. The immediate cause of this response was likely change in vegetation structure that increased habitat quality in both our study and that of Linz and Glixt (1997), such as decaying cattail mats broken apart by wind and wave action, exposing more open water and mud flat areas where marshbirds may forage. Similarly, Lehikoinen et al. (2017) observed an increase in waterbird abundance on Finnish wetlands managed to reduce dense, homogeneous areas of emergent vegetation, although they did not include herbicide application to control emergent vegetation as a treatment in their study. Conversely, Anderson et al. (2022) reported no association between herbicide application on monotypic cattail and marshbird abundance in east-central Minnesota, but concluded that marshbird community composition was related to wetland vegetation structure.

A common result of management actions to reduce dense vegetation is increasing the amount of irregular patch edges and openings of exposed shallow water or mudflat, which affords foraging areas and conditions where native hydrophytic plant species produce seed and harbor macroinvertebrates that comprise marshbird food. We suspect that herbicide application in the wetlands we studied increased marshbird breeding habitat quality within 3 years of treatment through altering vegetation and other wetland characteristics, and the effects that changes in vegetation, direct effects on food availability, or both, may have on nesting habitat or food availability. For example, food abundance and availability may be affected either directly or indirectly by herbicide application, but we do not know the extent or direction of these effects in the system we studies. In addition, we note that marshbird-habitat relations in the context of wetland vegetation and the response of breeding marshbirds to large-scale changes in vegetation and wetland characteristics are poorly documented in the PPR (but see Fairbairn and Dinsmore 2001, Orr et al. 2020) —future evaluation of the potential effects of food abundance and availability, factors that influence marshbird detection, and marshbird—habitat relations and the breeding home-range scale may provide further insight into the response of marshbirds to control of invasive vegetation in PPR and other wetlands. For impounded PPR wetlands, our results indicate a positive response of breeding marshbirds to chemical control of invasive, monotypic cattail vegetation, and that the effect of that control on marshbirds is evident 3 years following treatment and may extend further. Furthermore, although the general patterns among populations of marshbird species we monitored were similar, the strength and magnitude of responses varied among species, suggesting that other aspects of marshbird-habitat relations also likely influenced marshbird populations in the systems we studied.

**MANAGEMENT IMPLICATIONS**

We observed evidence of an increase in marshbird abundance 3 years following application of herbicide in late summer and early autumn to control invasive cattail that dominated impounded PPR wetlands in northwestern Minnesota, USA. Our study design did not allow us to elucidate the mechanism resulting in an increase in marshbird expected mean counts following herbicide application, but we hypothesize that marshbirds responded to the changes in vegetation and wetland characteristics that occur in cattail-dominated, impounded wetlands following herbicide application—increased structural and plant diversity and spatial heterogeneity that increased the amount of open water and edge between open water and emergent vegetation, and may have also affected food availability and nesting habitat quality. Our assessment suggests that breeding marshbirds respond positively following application of herbicides to control cattail in impounded PPR wetlands, but increases in abundance lag herbicide application by several years. Furthermore, any potential negative population-level effects are likely short-lived for the 5 marshbird species we studied (sora, Virginia rail, pied-billed grebe, American bittern, and least bittern). It is not clear how long the effects of herbicide application on marshbird abundance might last in the system we studied, but periodic herbicide treatments may be necessary to maintain vegetative conditions associated with increased breeding marshbird abundance.

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ETHICS STATEMENT: The University of Minnesota Institutional Animal Care and Use Committee (IACUC protocol #1503-32456A) approved the protocol for this study.

Literature Cited

Anderson, S. L., D. A. McGranahan, T. J. Hovick, and A. R. Hewitt. 2019. Passerine and secretive marsh bird responses to cattail management in temperate wetlands. Wetlands Ecology and Management 27:283–293.

Bansal, S., S. C. Lishawa, S. Newman, B. A. Tangen, D. Wilcox, D. Albert, M. J. Anteau, M. J. Chimney, R. L. Cressey, E. DeKeyser, K. J. Elgersma, S. A. Finkelstein, J. Freeland, R. Grosshans, P. E. Klug, D. J. Larkin, B. A. Lawrence, G. Linz, J. Marburger, G. Noe, C. Otto, N. Reo, J. Richards, C. Richardson, L. R. Rodgers, A. J. Schrank, D. Svedarsky, S. Travis, N. Tuchman, and L. Windham-Myers. 2019. *Typha* (cattail) invasion in North American wetlands: biology, regional problems, impacts, ecosystem services, and management. Wetlands 39:645–684.

Bart, J., R. A. Stehn, J. A. Herrick, N. A. Heaslip, T. A. Bookhout, and J. R. Stenzel. 1984. Survey methods for breeding yellow rails. Journal of Wildlife Management 48:1382–1386.

Benoît, J., R. Bazin, L. Maynard, A. McConnell, and J. Stewart. 2011. Least bittern (*Ixobrychus exilis*) survey protocol. The Birds of North America Online 34:225–233.

Benoît, J., L. Robillard, and C. Latendresse. 2009. Response of a least bittern (*Ixobrychus exilis*) population to interannual water level fluctuations. Waterbirds 32:73–80.

Beule, J. D. 1979. Control and management of cattails in southeastern Wisconsin wetlands. Technical Bulletin No. 112. Wisconsin Department of Natural Resources, Madison, Wisconsin, USA.

Blann, K. L., J. L. Anderson, G. R. Sands, and B. Vondracek. 2009. Effects of agricultural drainage on aquatic ecosystems: a review. Critical Reviews in Environmental Science and Technology 39:909–1001.

Bolenbaugh, J. R., D. G. Krementz, and S. E. Lehnen. 2011. Secretive marsh bird species co-occurrences and habitat associations across the Midwest, USA. Journal of Fish and Wildlife Management 2:49–60.

Bourdaghs, M., J. Genet, M. Gernes, and E. Peters. 2015. Status and trends of wetlands in Minnesota: vegetation quality baseline. Report wq-bwm-1-09. Minnesota Pollution Control Agency, St. Paul, Minnesota, USA.

Brooks M. E., K. Kristensen, K. J. van Benthem, A. Magnusson, C. W. Berg, A. Nielsen, H. J. Skaug, M. Maechler, and B. M. Bolker. 2017. glmmTMB balances speed and flexibility among packages for zero-inflated generalized linear mixed modeling. The R Journal 9:378–400. https://journal.r-project.org/archive/2017/RJ-2017-066/index.html .

Chabot, D., V. Carignan, and D. M. Bird. 2014. Measuring habitat quality for least bitterns in a created wetland with use of a small unmanned aircraft. Wetlands 34:527–533.

Conway, C. J. 2011. Standardized North American marsh bird monitoring protocol. Waterbirds 34:319–346.

Dahl, T. E. 2014. Status and trends of prairie wetlands in the United States 1997 to 2009. Ecological Services, U.S. Fish and Wildlife Service, Washington, D.C., USA. 67 pp.

Delphey, P. J., and J. J. Dinsmore. 1993. Breeding bird communities of recently restored and natural prairie potholes. Wetlands 13:200–206.

Elgersma, K. J., J. P. Martina, D. E. Goldberg, and W. S. Currie. 2017. Effectiveness of cattail (*Typha* spp.) management techniques depends on exogenous nitrogen inputs. Elementa: Science of the Anthropocene 5:19.

Fairbairn, S. E., and J. J. Dinsmore. 2001. Local and landscape-level influences on wetland bird communities of the prairie pothole region of Iowa, USA. Wetlands 21:41–47.

Galatowitsch, S. M. 2006. Restoring prairie pothole wetlands: does the species pool concept offer decision-making guidance for re-vegetation? Applied Vegetation Science 9:261–270.

Galatowitsch, S. M., N. O. Anderson, and P. D. Ascher. 1999. Invasiveness in wetland plants in temperate North America. Wetlands 19:733–755.

Glisson, W. J., R. S. Brady, A. T. Paulios, S. K. Jacobi, and D. J. Larkin. 2015. Sensitivity of secretive marsh birds to vegetation condition in natural and restored wetlands in Wisconsin. Journal of Wildlife Management 79:1101–1116.

Green, R. H. 1979. Sampling design and statistical methods for environmental biologists. Wiley, Chichester, England.

Harms, T. M., and S. J. Dinsmore. 2013. Habitat associations of secretive marsh birds in Iowa. Wetlands 33:561–571.

Johnson, R. R., and J. J. Dinsmore. 1986. Habitat use by breeding Virginia rails and soras. Journal of Wildlife Management 50:387–392. <http://www.jstor.org/stable/3801092>.

Kantrud, H. A., and W. E. Newton. 1996. A test of vegetation-related indicators of wetland quality in the Prairie Pothole Region. Journal of Aquatic Ecosystem Health 5:177–191.

Kloiber, S. M., and D. J. Norris. 2013. Status and trends of wetlands in Minnesota: wetland quantity trends from 2006 to 2011. Minnesota Department of Natural Resources, St. Paul, Minnesota, USA.

Lehikoinen, P., A. Lehikoinen, M. Mikkola-Roos, and K. Jaatinen. 2017. Counteracting wetland overgrowth increases breeding and staging bird abundances. Scientific Reports 7:41391.

Lenth R.V. 2022. emmeans: estimated marginal means, aka least-squares means. R package version 1.7.4-1. https://CRAN.R-project.org/package=emmeans

Linz, G. M., D. L. Bergman, D. C. Blixt, and W. J. Bleier. 1994. Response of black terns (*Chlidonias niger*) to glyphosate-induced habitat alterations on wetlands. Colonial Waterbirds 17:160–167.

Linz, G. M., and D. C. Blixt. 1997. Black terns benefit from cattail management in the northern Great Plains. Colonial Waterbirds 20:617–621.

Linz, G. M., and H. J. Homan. 2011. Use of glyphosate for managing invasive cattail (*Typha* spp.) to disperse blackbird (*Icteridae*) roosts. Crop Protection 30:98–104.

Lishawa, S. C., B. A. Lawrence, D. A. Albert, and N. C. Tuchman. 2015. Biomass harvest of invasive *Typha* promotes plant diversity in a Great Lakes coastal wetland. Restoration Ecology 23:228–237.

Lor, S., and R. A. Malecki. 2006. Breeding ecology and nesting habitat associations of five marsh bird species in western New York. Waterbirds 29:427–436.

Martin, K., N. Koper, and R. Bazin. 2014. Optimizing repeat-visit, call-broadcast nocturnal surveys for yellow rails (*Coturnicops noveboracensis*). Waterbirds 37:68–78.

Minnesota Prairie Plan Working Group. 2018. Minnesota Prairie Conservation Plan. Minnesota Prairie Plan Working Group, Minneapolis, Minnesota, USA. <<https://files.dnr.state.mn.us/eco/mcbs/mn_prairie_conservation_plan.pdf>>. Accessed 25 September 2018.

MNDNR. 2015. Ecological classification system. Minnesota Department of Natural Resources, Saint Paul, Minnesota, USA. <http://www.dnr.state.mn.us/ecs/index.html>. Accessed 13 Jun 2022.

Mulhouse, J.M., and S. M. Galatowitsch. 2003. Revegetation of prairie pothole wetlands in the mid-continental US: twelve years post-reflooding. Plant Ecology 169:143–159.

National Research Council, Committee on Restoration of Aquatic Ecosystems-Science. 1992. Restoration of aquatic systems: science, technology, and public policy. The National Academies Press, Washington, D.C., USA. <https://doi.org/10.17226/1807>

O’Donnell, K. M., F. R. Thompson III, and R. D. Semlitsch. 2015. Prescribed fire and timber harvest effects on terrestrial salamander abundance, detectability, and microhabitat use. Journal of Wildlife Management 79:766–775. <https://wildlife.onlinelibrary.wiley.com/doi/abs/10.1002/jwmg.884>

Orr, J. T., C. A. Duquette, T. J. Hovick, B. A. Geaumont, and T. M. Harms. 2020. Secretive marsh bird densities and habitat associations in the Prairie Pothole Region. Wetlands 40:1529–1538.

Pulfer, T. L., C. Hazzard, and D. Puric-Mladenovic. 2014. Understanding reptile and amphibian trends in relation to changes in wetlands: a pilot in the Essex Region Watershed. Conference Proceedings, 2014 Great Lakes Wetlands Day. Great Lakes Wetlands Conservation Action Plan, Toronto, Ontario, Canada.

Ratti, J. T., A. M. Rocklage, J. H. Giudice, E. O. Garton, and D. P. Golner. 2001. Comparison of avian communities on restored and natural wetlands in North and South Dakota. Journal of Wildlife Management 65:676–684.

Sidie-Slettedahl, A. M. 2013. Evaluating the use of autonomous recording units to monitor yellow rails, Nelson’s sparrows, and Le Conte’s sparrows. Thesis. South Dakota State University, Brookings, South Dakota, USA. <<http://pubstorage.sdstate.edu/wfs/thesis/Sidie-Slettedahl-Anna-2013-MS.pdf>> Accessed September 2015.

Sidie-Slettedahl, A. M., K. C. Jensen, R. R. Johnson, T. W. Arnold, J. E. Austin, and J. D. Stafford. 2015. Evaluation of autonomous recording units for detecting 3 species of secretive marsh birds. Wildlife Society Bulletin 39: 626–634.

Sojda, R. S., and K. L. Solberg. 1993. Waterfowl management handbook: 13.4.13. Management and control of cattails. Fish and Wildlife Leaflet 13.4.13. U.S. Fish and Wildlife Service, Washington, D.C., USA.

Spyreas, G., B. W. Wilm, A. E. Plocher, D. M. Ketzner, J. W. Matthews, J. L. Ellis, and E. J. Heske. 2010. Biological consequences of invasion by reed canary grass (*Phalaris arundinacea*). Biological Invasions 12:1253–1267.

Tuchman, N. C., D. J. Larkin, P. Geddes, R. Wildová, K. Jankowski, and D. E. Goldberg. 2009. Patterns of environmental change associated with *Typha* x *glauca* invasion in a Great Lakes coastal wetland. Wetlands 29:964–975.

Vacek S., and B. Friske. 2012. Habitat management plan for Morris Wetland Management District Morris, Minnesota. U.S. Fish and Wildlife Service Morris Wetland Management District, Morris, Minnesota, USA.

VanRees-Siewert, K. L., and J. J. Dinsmore. 1996. Influence of wetland age on bird use of restored wetlands in Iowa. Wetlands 16:577–582.

Weller, M. W. 1981. Estimating wildlife and other wetland losses due to drainage and other perturbations. Pages 337–346 *in* B. Richardson, editor. Selected proceedings of the Midwest Conference on Wetland Values and Management. Minnesota Water Planning Board, St. Paul, Minnesota, USA.

Wiken, E., F. Jiménez Nava, and G. Griffith. 2011. North American terrestrial ecoregions—Level III. Commission for Environmental Cooperation, Montreal, Québec, Canada. <ftp://ftp.epa.gov/wed/ecoregions/pubs/NA\_TerrestrialEcoregionsLevel3\_Final- 2june11\_CEC.pdf>.

Wiltermuth, M. T., and M. J. Anteau. 2016. Is consolidation drainage an indirect mechanism for increased abundance of cattail in northern prairie wetlands? Wetlands Ecology and Management 24:533–544.

Zedler, J. B. 2000. Progress in wetland restoration ecology. Trends in Ecology and Evolution 15:402–407.

Zedler, J. B., and S. M. Kercher. 2004. Causes and consequences of invasive plants in wetlands: opportunities, opportunists, and outcomes. Critical Reviews in Plant Sciences 23:431–452.

Zimmerman, A. L., J. A. Dechant, B. E. Jamison, D. H. Johnson, C. M. Goldade, J. O. Church, and B. R. Euliss. 2002. Effects of management practices on wetland birds: Virginia rail. U.S. Geological Survey Northern Prairie Wildlife Research Center Grasslands Ecosystem Initiative, Jamestown, North Dakota, USA.

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Figure Captions

Figure 1.Minnesota Department of Natural Resources (MNDNR) Wildlife Management Areas in northwestern Minnesota, USA, with large, impounded wetlands invaded by dense cattail (*Typha angustifolia* and *Typha* x *glauca*) stands. We evaluated the effects of glyphosate herbicide application by the MNDNR to control dense cattail on breeding marshbirds. We conducted standardized marshbird surveys at study sites that contained both treatment (treated with herbicide; red polygons) and control (not treated with herbicide and indicated by survey locations; located in the same or adjacent basin to treatment) areas and evaluated counts of marshbirds the spring before and up to 3 springs (2105–2018) after herbicide application.

Figure 2.Map of Elm Lake Wildlife Management Area in northwestern Minnesota, USA, indicating areas that received aerial glyphosate application to control cattail (*Typha angustifolia* and *Typha* x *glauca*) in late summer and early autumn 2015, and marshbird survey locations.

Figure 3.Expected mean marshbird counts (upper panel; error bars represent 90% confidence interval) within years 2015 to 2018 in 9 study sites that included both treatment (areas treated with herbicide) and control (areas not treated with herbicide) survey locations in northwestern Minnesota, USA, and the difference between expected mean marshbird counts (lower panel) for 5 species of marshbirds [American bittern (AMBI), least bittern (LEBI), pied-billed grebe (PBGR), sora (SORA), and Virginia rail (VIRA)]. We evaluated whether herbicide application affected mean marshbird counts by conducting surveys during spring breeding seasons at treatment and control survey locations and evaluated change in number of detections from the spring before to 3 springs after herbicide application (2015 – 2018). Statistical results are pairwise comparisons between the mean expected counts at control and treatment survey locations within study sites based on a generalized linear mixed model with plot as a random effect and a treatment-by-year interaction that was run for each individual species (log link, Poisson family). Asterisks represent significant treatment effects at *P* < 0.10.

**Summary for online Table of Contents**: Using a before-after, control-impact study design, we evaluated response of breeding marshbird abundance (indexed by standardized counts) to herbicide application to control invasive cattail in impounded wetlands in the eastern Prairie Pothole Region of northwestern Minnesota, USA. We observed evidence of increased abundance of marshbirds that lagged 3 years behind herbicide application.

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