Final Report:

WATERBIRD RESPONSE TO CONSERVATION GRAZING IN WESTERN MINNESOTA TALLGRASS PRAIRIES

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EXECUTIVE SUMMARY

Conservation grazing is becoming a popular management tool in grassland systems that evolved with disturbance. While it has been found to be an effective tool for maintaining upland prairie cover, its impacts on wetlands and the species that rely upon wetlands is not well-studied. Our objective was to investigate the impacts of conservation grazing on two groups of waterbirds: secretive marsh birds and ground-nesting waterfowl. As an economically important group of birds, much of the public land in Minnesota is managed for the production of waterfowl. Meanwhile, secretive marsh birds are believed to be an indicator species, and their response to grazing could tell us how the whole system is responding to management. In 2017 and 2018, we conducted waterfowl pair and brood counts and call-response surveys for secretive marsh birds. We estimated abundance of Blue-winged teal and Mallard pairs in response to grazing frequency and years since a site was grazed using generalized linear models and estimated Blue-winged Teal brood abundance using generalized linear mixed effects models. Insufficient observations of Mallard broods prohibited analysis of brood abundance, but we combined all dabbler broods together to estimate overall brood abundance in relation to grazing. Using robust design occupancy models in Program MARK, we determined the relationship between grazing and marsh bird occupancy. Grazing explained variation in occupancy of American Bittern, Pied-billed Grebe, Sora, and Virginia Rail. American Bittern, Sora, and Virginia Rail responded positively to grazing. Pied-billed Grebe experienced a modest increase in local extinction one year after grazing but showed no response to increased grazing frequency. Grazing had no effect on Blue-winged Teal pair or brood abundance or dabbler brood abundance. Mallard abundance increased marginally in response to increased time since a site was grazed but showed no response to increased grazing frequency. Our study suggests that the use of frequent grazing at

low intensities to manage upland cover should neutrally or positively impact occupancy of secretive marsh birds and abundance of ground-nesting waterfowl pairs and broods.

INTRODUCTION

Until fairly recently, a widely-held belief was that grazing by livestock provided few benefits to grassland systems (Schieltz and Rubenstein 2016). This viewpoint likely stemmed from the harmful effects caused by unsustainable grazing practices during recent centuries (Briske et al. 2011). Year-round grazing at a high intensity led to a homogenous, overgrazed landscape that inhibited native plant species growth and species diversity (Briske et al. 2011, Richmond et al. 2012, Morris and Reich 2013, Minnesota Prairie Plan Working Group 2018). Overgrazing also negatively impacts wetlands, resulting in increased erosion and sedimentation, eutrophication, and the removal of emergent vegetation (Kantrud 1986, Harrison et al. 2017). However, in the tallgrass prairie of Minnesota, native prairie plants evolved in an ecosystem with high levels of ungulate grazing pressure (Sampson and Knopf 1994, Grace 1998, Heisler et al. 2003), and sustainable grazing practices can promote structural diversity of upland and wetland vegetation to benefit a greater diversity of wildlife species (Kantrud 1986, Morris and Reich 2013).

North American prairies evolved with and were maintained by frequent disturbances that influenced the architecture of the land, including fire and grazing (Collins 1990, Fuhlendorf and Engle 2004). Regular wildfires ensured that fire-intolerant species, notably trees and shrubs, were unable to establish themselves on the prairie (Anderson 1990, Grace 1998). Fires burned unevenly, creating a mosaic of successional stages on the landscape that supported greater species diversity (Collins 1990, Fuhlendorf and Engle 2004). Recently burned patches were preferred by grazing ungulates, enhancing their populations and further disturbing and shaping vegetation communities (Collins and Gibson 1990, Minnesota Prairie Plan Working Group 2018).

Additionally, both fire and grazing played an important role in controlling excessive accumulation of dead plant litter, which decreases primary productivity (Collins 1990).

While we now recognize the critical role that frequent disturbance plays in grassland ecosystems, humans spent the last 200 years removing most disturbance from our prairie. We have eradicated our native grazing ungulates, most notably bison (*Bos bison*), and actively suppressed wildfires as they pose a significant threat to the livelihood and survival of those living and working on the land (Collins 1990, Kantrud 1986). In the absence of disturbance, trees, shrubs, and non-native and invasive species have infiltrated prairie systems, out-competing native prairie species and altering the ecosystem (Sampson and Knopf 1994, Grace 1998, Bakker 2003, Heisler et al. 2003, Minnesota Prairie Plan Working Group 2018). An estimated 99.6% of Minnesota's native tallgrass prairie has been lost since European settlement (Sampson and Knopf 1994, Minnesota Prairie Plan Working Group 2018) as a direct result of widespread cultivation and urban development (Samson and Knopf 1994, Thompson et al. 2016), and unfortunately, what prairie remains is often highly degraded due to the suppression of natural disturbances (Sampson and Knopf 1994, Grace 1998, Bakker 2003, Heisler et al. 2003). The fate of Minnesota’s prairies extends to the wetlands embedded within them. In the last 200 years, 91.9% of wetlands in Minnesota’s Prairie Parkland Region have disappeared, primarily due to agricultural drainage (Minnesota Prairie Plan Working Group 2018). Meanwhile, the Minnesota Pollution Control Agency has reported that, of the wetlands remaining, 50% are rated to be in poor condition (Genet 2015). This loss in quality is due in large part to the proliferation of invasive emergent vegetation species (Genet 2015, Minnesota Prairie Plan Working Group 2018), which have been able to establish themselves in the absence of regular disturbances (Minnesota Prairie Plan Working Group 2018). Today, undisturbed, shallow wetlands quickly become choked with invasive hydrophytes, particularly hybrid cattails (*Typha X glauca* Godr. and *T. angustifolia*) and reed canary grass (*Phalaris arundinacea*) (Kantrud 1986, Genet 2015, Minnesota Prairie Plan Working Group 2018). Hybrid cattail form monocultures that dominate entire wetlands, leaving little to no open water and negatively impacting plant species and structural diversity (Kantrud 1986).

Because wildfires still pose a significant economic threat, they continue to be actively suppressed. And with our native grazers gone, remnant prairies would go undisturbed without human intervention. Today, in the absence of natural disturbances, land managers must recreate these phenomena as best they can if they hope to preserve what remains of Minnesota's native prairie. Notable objectives of mimicking disturbance include suppression of woody and non-native species, encouragement of native plant species growth, reduction of litter buildup, and promotion of a structurally diverse plant community (Collins 1990, Devries and Armstrong 2011). Whereas prescribed burning is widely-regarded as an effective disturbance tool on the prairie (Collins and Gibson 1990), it is expensive, logistically complicated, and difficult to perform at the intensity and frequency that are needed (Thompson et al. 2016). Because woody vegetation can benefit from infrequent burning (Grace 1998, Heisler et al. 2003), it is important that land managers consider other disturbance tools if they are unable to perform burns as frequently as needed.

One practical alternative to burning is conservation grazing (Minnesota Prairie Plan Working Group 2018). Widespread grazing by bison (*Bos bison*) and other ungulate species once acted as a secondary natural disturbance regime in the Great Plains (Grace 1998, Heisler et al. 2003), and conservation grazing using domestic livestock attempts to mimic this regime (Fuhlendorf and Engle 2004). Past studies have established that grazing can positively impact upland cover, controlling dead litter buildup, promoting native species growth, and creating structural diversity of prairie plant communities (Ryan et al. 2006, Morris and Reich 2013). However, the impacts of conservation grazing on wetlands and wetland-dependent wildlife is less well-known (U.S. Fish and Wildlife Service 2017). Intensive levels of grazing are recognized as being harmful to wetland health (Kantrud 1986, Harrison et al. 2017), but conservation grazing is performed at more sustainable levels (Minnesota Prairie Plan Working Group 2018). If conservation grazing is going to be used by land managers in the tallgrass prairies on the Midwest, it is important that we consider how grazing in this ecosystem is impacting prairie wetlands and the species that rely upon them for habitat.

Secretive marsh birds are rarely seen and infrequently vocalize (Conway and Gibbs 2005, Conway 2011). Though one of the least studied avian groups in North America, many populations are believed to be in decline (Conway 2011). They include rails, bitterns, and grebes, and although they are not taxonomically related, they all rely on similar wetland habitat that involves a juxtaposition of dense emergent vegetation and open water (Lor and Malecki 2006). Emergent vegetation that becomes too thick can be a hinderance to water birds, impeding their ability to move and forage (Ma et al. 2010, Johnson 1984). Additionally, monocultures of hybrid cattail create a structurally homogenous vegetation community that is unattractive to many waterbird species (Kantrud 1986, Minnesota Prairie Plan Working Group 2018). Monitoring the productivity of secretive marsh birds as it relates to wetland management practices could reveal how the entire system is reacting to manipulations (Conway 2011).

Ground-nesting waterfowl are in the unique position of relying on both upland and wetland habitat during the breeding season. Upland vegetation is important for providing cover for nests (Reynolds et al. 2001, Stephens et al. 2005n), whereas emergent wetland vegetation provides cover to flightless broods (Raven et al. 2007). Nevertheless, too much emergent cover may reduce the attractiveness of prairie wetlands to some dabbling duck broods (Walker et al. 2013). Because waterfowl are economically important game species for the region, how ground-nesting ducks respond to grazing will likely be important to land managers considering conservation grazing as a disturbance tool. Although studies have assessed the impact of grazing on waterfowl, results have been inconsistent (Duebbert et al. 1986, Warren et al. 2008, Harrison et al. 2017). Furthermore, the majority have focused on the impacts of high-intensity grazing. These studies typically compared grazed to ungrazed sites without considering other factors (i.e. time of year, intensity, frequency) (Briske et al. 2011, Schieltz and Rubenstein 2016). The few studies that have investigated waterfowl breeding success across a range of intensities have found benefits at low to medium levels of grazing, but harm at high levels (Warren et al. 2008, Harrison et al. 2017). But we are unaware of any studies that have investigated how the frequency of grazing affects waterfowl productivity. How often a site should be disturbed might be as important as how intensely it should be disturbed (Devries and Armstrong 2011).

Our objective was to evaluate waterbird response to conservation grazing in the tallgrass prairie of western Minnesota. Collectively, the U.S. Fish and Wildlife Service (USFWS), Minnesota Department of Natural Resources (DNR), and non-profit conservation organizations such as The Nature Conservancy (TNC) manage thousands of acres of remnant native prairie throughout western Minnesota (Johnson 1997). With so much land to be managed and regularly disturbed, conservation grazing is a potentially valuable tool for managers who are unable to burn as often as needed or who are unable to burn at all. With so few of the state’s original wetlands still present on the land, and even fewer of good quality, it is important to understand how conservation grazing practices are impacting a vulnerable group of birds that rely on wetlands and emergent vegetation during a critical time in their life cycle.

METHODS

Study Area

We surveyed wetlands on Waterfowl Production Areas (WPAs) in the Morris Wetland Management District (WMD) in western Minnesota. Historically, this was a productive tallgrass prairie system that provided important habitat for migrating birds (both as breeding habitat and migration stop-overs). Since European settlement, however, the land has seen widespread conversion to row-crop agriculture and a significant loss of prairie and wetland habitat (U.S. Fish and Wildlife Service 2012). Within Morris WMD, just 10% of the district's historical wetlands remain and an estimated 1% of its native prairie (U.S. Fish and Wildlife Service 2012, U.S. Fish and Wildlife Service 2017). Despite the prairie and wetland loss, this region's remnant prairie still provides important habitat to prairie species.

These public lands are managed primarily for the production and conservation of migrating waterfowl by the United States Fish and Wildlife Service (USFWS) (U.S. Fish and Wildlife Service 2017). The Morris WMD is in Minnesota's Prairie Parkland Region and lies within the eastern-most extent of the Prairie Pothole Region (PPR). The district is characterized by a landscape transitioning from relatively flat tallgrass prairie in the west to rolling hardwood forests in the east (U.S. Fish and Wildlife Service 2017). (U.S. Fish and Wildlife Service 2017). These distinct topographies impact the characteristics of the wetlands on the land. Wetlands in the western counties are smaller and shallower with more emergent vegetation, while wetlands in the eastern counties are larger and deeper with more open water.

To meet their objective of maximizing waterfowl production, the Morris WMD largely focuses on upland habitat enhancement. They actively burn, graze, and hay their lands to promote growth of native prairie plants and combat woody and nonnative species encroachment (U.S. Fish and Wildlife Service 2017). The district has an extensive grazing program and collaborates with cattle producers to provide pastures in exchange for a grazing fee. Their grazing objectives include reducing accumulation of biomass, engineering a more heterogeneous landscape, and promoting the growth of native grasses through reduced competition from invasives. To achieve these objectives, they focus efforts on short-duration, late spring grazing. Grazing contracts typically stipulate 30-day grazing periods between May 1 and July 31 at a stocking rate of 1 cow-calf pair per acre (U.S. Fish and Wildlife Service 2017).

Site Selection

We selected sites with a variety of grazing histories to investigate waterbird response to grazing. Larger WPAs were often divided into smaller management units with variable management histories. In these instances, we often had more than 1 site on a WPA. Using management histories of the district's WPAs, we grouped sites by the number of years since they were last grazed: 0 (currently being grazed), 1, 2, 3, 4, 5, 6, 7+ years. We considered sites that had no form of disturbance (grazing, burning, or haying) in over 6 years undisturbed. Due to the limited number of sites with grazing as the only form of recent disturbance management, selecting sites randomly was not an option. Because the district actively manages with fire as well as grazing, it was impossible to select a large enough sample of grazed WPAs that did not have a recent burn history. As a result, we incorporated some sites that had been burned as well as grazed in the last 6 years. Some 2017 sites were dropped in 2018 if they were burned or mowed between field seasons, while some previously unsurveyed sites were added in 2018.

Secretive Marsh Birds

Field Methods: Depending on the size of the site, we randomly selected ≥1 wetland per site that would be assigned a marsh bird survey point. To minimize the chance of counting the same bird at multiple survey points, we spaced points ≥400 meters apart (Conway 2011). On larger sites that allowed for more than 1 wetland to be surveyed, we randomly selected a second wetland to survey. Following the protocol of Conway (2011), we placed survey points along the edges of selected wetlands where emergent vegetation was present. We grouped marsh bird points into survey routes based on proximity to each other and maintained those routes and the order in which wetlands were surveyed across each field season (Conway 2011). Because some 2017 sites were dropped and others added in 2018, not all survey routes remained the same in 2017 and 2018.

We adapted the Standardized North American Marsh Bird Monitoring Protocol (Conway 2011) to survey 6 species of secretive marsh birds: American Bittern (*Botaurus lentiginosus*), Least Bittern (*Ixobrychus exilis*), Pied-billed Grebe (*Podilymbus podiceps*), Sora (*Porzana carolina*), Virginia Rail (*Rallus limicola*), and Yellow Rail (*Coturnicops noveboracensis*). Unlike the standardized protocol, which specifies 3 visits during the marsh bird breeding season, we surveyed marsh bird routes twice, choosing to prioritize a larger sample size over additional visits. We surveyed each route once in the morning (30 minutes prior to sunrise to 0930 hours) and once in the evening (1800 hours to 30 minutes after sunset) (Bolenbaugh et al. 2011) using different observers for each visit.

At the beginning of each survey, we documented time and estimated wetland inundation and percent emergent vegetation covering the wetland using both aerial images of the wetland and various vantage points around the wetland. When wind impeded our ability to detect calls (wind ≥ 24 km/h; 15 mph), we halted surveys. Additionally, we paused or halted surveys in the event of continuous precipitation. If a survey route could not be completed, we started the survey route over on the next available day (Conway 2011).

Surveys were 11 minutes long, consisting of a 5-minute passive survey followed by 6 minutes of broadcasting common breeding and territorial calls of our 6 target species to elicit responses from secretive marsh birds. Each of these latter minutes consisted of 30 seconds of broadcasting various calls of 1 species followed by 30 seconds of silence. Upon hearing one of the focal species, the surveyor recorded the species, type of call, minute detected, whether the bird was seen and/or heard, and estimated location to help determine if a bird heard later in the survey was a new bird.

Statistical methods: We assessed the impact of grazing and other variables on occurrence of secretive marsh birds with robust design occupancy models (McKenzie et al. 2003) in Program MARK (White and Burnham 1999) using a limited, *a priori* set of candidate models (Ahlering et al. 2018). Robust design occupancy allowed us to incorporate multiple visits (i.e., 2 visits per year, for 1 or 2 years) without assuming an individual site maintained the same occupancy status throughout all 4 surveys. Because our primary focus was on occupancy, we parameterized the model by estimating occupancy during all 4 survey periods (ψ_t) and estimating local extinction (ε_t) between survey periods (McKenzie et al. 2003). We expected detection probabilities (p) of each species to differ in their response to playback and occupancy to differ in response to grazing, and so we modeled each species separately.

We first modeled vocalization probability for each species using an intercept-only model for ψ. We built vocalization probability models that considered how broadcasted calls impacted the probability of a species calling. We reasoned that the probability of calling could be influenced by survey minute, survey period, and survey year. Starting with survey minute, we held visit and year constant and reasoned that there were 4 likely vocalization behaviors in response to broadcasted calls. All four assumed a constant detection probability throughout the initial 5-minute passive survey. First, we considered that the broadcasted calls did not influence the probability of calling; therefore, the model had a constant detection probability throughout the 11-minute survey. Second, we considered that only conspecific calls influenced vocalization probability, thus all heterospecific calling minutes and silent minutes were held constant while the conspecific minute had a separate (presumably greater) detection probability. The third and fourth models considered that both conspecific and heterospecific calls influenced vocalization probability. The third model included 1 vocalization probability for all heterospecific calling minutes and 1 for the conspecific calling minute, while the fourth model allowed for variation in vocalization probability among all 6 calling minutes. Using second-order Akaike's Information Criterion (AIC_c), we determined the best performing model that explained each species' response to broadcasted calls. We then constructed models to investigate whether vocalization probability was influenced by survey period or survey year. As a final step, we incorporated a suite of 5 variables that have been documented to impact detection probability, including observer, noise level, AM vs. PM, time since sunrise/sunset, and survey day.

After identifying an AIC_c -best detection model for each species, we then fit companion occupancy models. Because the primary objective of our project was to investigate the impact that grazing might have on occupancy of secretive marsh birds, our analysis was designed to assess whether the inclusion of grazing variables altered predictions about site-level occupancy. Accordingly, we built a base occupancy model that included influential site- and landscape-level characteristics that past studies have shown to be correlated with marsh bird occupancy. The site-level characteristics we included were 1) area of the surveyed wetland, 2) percent of emergent vegetation covering the wetland, 3) wetland permanence class (Shaw and Fredine 1956), and 4) the number of years since a site was burned. The landscape-level variable we included was total wetland area within a 200-meter buffer. To calculate this last variable, we created 200-meter buffers around the central points of surveyed wetlands and overlaid all wetland types intersecting the buffers. We then calculated the total area in hectares of the intersecting wetlands within each buffer.

To assess whether the addition of a grazing variable to the base model improved model performance, we fit 4 grazing models for each species with occupancy as the response variable. While we selected survey sites based on the number of years since a site was grazed (years since grazed), we created an additional grazing variable to illustrate grazing frequency: the number of times a site had been grazed in the last 3 years. Because these 2 grazing variables were highly correlated, we considered their effects in separate models. We fit 1 model with grazing frequency, 1 model with a linear variable for years since grazed, and to account for potential non-linear responses, we added a quadratic term for years since grazed to a third model. To determine if grazing caused a site that was occupied in year 1 to become unoccupied, we fit a fourth model that included a binary covariate on the extinction parameter ($\varepsilon_{2\text{-}3}$) to determine if sites that were grazed the previous year were more likely to become unoccupied. We compared the 4 grazing models to the base model described above using AIC_c.

Ground-nesting Waterfowl

Field methods: We conducted four rounds of waterfowl surveys in 2017 and 2018. During the first two rounds, our sampling methods were influenced by concurrent surveys of secretive marsh birds, which are performed in the mornings and evenings. Given that past studies have endorsed both morning and evening waterfowl surveys (Pagano and Arnold 2009*a*), we conducted visual surveys for waterfowl pairs in the morning (visible light – 1030) and evening (1700 – dark). Pair surveys were conducted from May 12 – June 3 in 2017 and May 15 – May 29 in 2018. We missed the optimum window for pair counts of early-nesting waterfowl (i.e. Mallard, Northern Pintail) in 2017; however, a much colder than average April and a late iceout (Waterfowl Population Status 2018) may have pushed back that window for early-nesters in 2018. We conducted three rounds of brood surveys between June and the end of July. Because we were no longer surveying for marsh birds during the third and fourth visits, all surveys were conducted during the morning. During the third visit, we conducted same-day replicate surveys to obtain an estimate of detection probability. We completed replicate visits at least one and no more than four hours after the first visit.

To survey for pairs and broods at each site, we conducted visual surveys of every wetland with visible open water. We rotated observers between visits to minimize impacts of observer variation. At the beginning of each survey, we documented time, temperature, cloud cover, and wind. We also estimated wetland inundation and percent emergent vegetation covering the wetland (Walker et al. 2013, Carlson et al. 2018) using both aerial images of the wetland and various vantage points around the wetland. When waterfowl were observed, we recorded species and social category: lone pair, lone male, lone female, grouped pairs, group of ≤5 males, group of >5 males, grouped females, group of 2 males and 1 female, other mixed sex groups, and broods

(Dzubin 1969). When broods were observed, we counted the number of ducklings and estimated their age (Gollop and Marshall 1954).

To ensure observers were able to competently survey an entire wetland, we excluded wetlands greater than 5 ha. On vegetated wetlands and larger wetlands, surveyors used multiple vantage points to adequately survey an entire wetland, spending at least 5 minutes per wetland to maximize detection probability (Pagano and Arnold 2009a). We did not return to wetlands that were dry or completely occluded by vegetation. We halted surveys when wind exceeded 50 km/hour, precipitation persisted, or fog impeded our ability to see. If a site could not be completed due to inclement weather, we started it over on the next available day.

Statistical methods: To evaluate waterfowl pair and brood response to grazing, we used a limited, *a priori* set of candidate models (Ahlering et al. 2018). Because the objective of our project was to investigate the impact that grazing may have on abundance of ducks, our analysis was designed to assess whether the inclusion of grazing variables altered predictions about site-level abundance. Accordingly, we built base models for pair and brood abundance that included influential site- and landscape-level characteristics that past studies have shown to be correlated with dabbling duck abundance. The site-level variables were longitude, wetland inundation, percent of emergent vegetation covering the wetland, area of surveyed wetland, wetland permanence (Shaw and Fredine 1956), and the number of years since the site was last burned. The landscape-level variables included were 1) the proportion of land cover classified as wetland habitat within a buffer (2000-m for pairs and 500-m for broods) of the surveyed wetland (National Wetlands Inventory 2018) and 2) the proportion of land cover classified as agricultural within those buffers (National Agricultural Statistics Service 2017). We also included observer and time of day because previous studies have shown that they can impact detection probability

(Pagano and Arnold 2009). Lastly, we included survey day and year as additional variables in our base model. Using z-transformation, we re-scaled our quantitative variables prior to analysis.

To assess whether the addition of a grazing variable to the base model improved model performance, we fit three grazing models for each set of analyses. We fit 1 model with grazing frequency, 1 model with a linear variable for years since grazed, and to account for potential non-linear responses, we added a quadratic term for years since grazed to a third model. We compared the 3 grazing models to the base model described above using AIC. If a grazing model had a lower AIC value than its corresponding base model, we concluded that grazing had a demonstrable effect on waterfowl abundance.

We modeled pair abundance using generalized linear models with a negative binomial distribution and indicated breeding pairs of a particular species as our response variable. We used the social categories explained earlier to calculate indicated breeding pairs: lone pairs + grouped pairs + max(lone males or lone females) + 2M:1F + max(grouped males ≤5 and grouped females). Our base model for each species included the same set of independent variables described above, which was compared to the three grazing models to determine if grazing was having a noticeable impact on pair abundance. To determine the effect of grazing on brood abundance, we fit generalized linear mixed effects models (glmer in lme4 package in R) with a Poisson distribution and number of broods of a species as our response variable.

Emergent Wetland Vegetation

To explore the potential impacts of grazing on emergent wetland vegetation, we fit linear models with percent emergent vegetation as the response variable and grazing as an indicator variable. We considered our 2 grazing variables (years since grazed and grazing frequency) in separate models, as we did for our marsh bird analysis. The first 2 models we fit investigated the response by emergent wetland vegetation to our grazing variables. Recognizing that there was likely a relationship between percent emergent wetland vegetation and wetland permanence, we fit a third model with wetland class as a predictor variable. We combined wetland classes 1 and 2, as there were insufficient numbers of each to analyze on their own. Lastly, we fit models with an additive and an interactive effect between class and our two grazing variables. With each of our grazing variables, we fit 1 model with grazing and wetland class as predictor variables and 1 model with an interaction between the two. We compared the AIC values of the 7 models to determine the top model and if grazing had any effect on percent emergent wetland vegetation.

RESULTS

Fifty sites were surveyed across 37 WPAs in 2017 and 53 sites across 41 WPAs in 2018. Between 2017 and 2018, we surveyed 85 sites that had a recent grazing history and 18 sites that had not been disturbed in 7+ years (Table 1). Thirty-two sites had been burned in the previous 6 years.

Table 1. Grazing histories of sites surveyed for secretive marsh birds and waterfowl in 2017 and 2018 in western Minnesota.

Years Since Last Grazed	**No. of 2017 Sites**	**No. of 2018 Sites**	**Total**
0	2	1	3
1	14	14	28
2	13	7	20
3	4	11	15
4	7	4	11
5	1	4	5
6	2	1	3
7+	7	11	18
Total	50	53	103

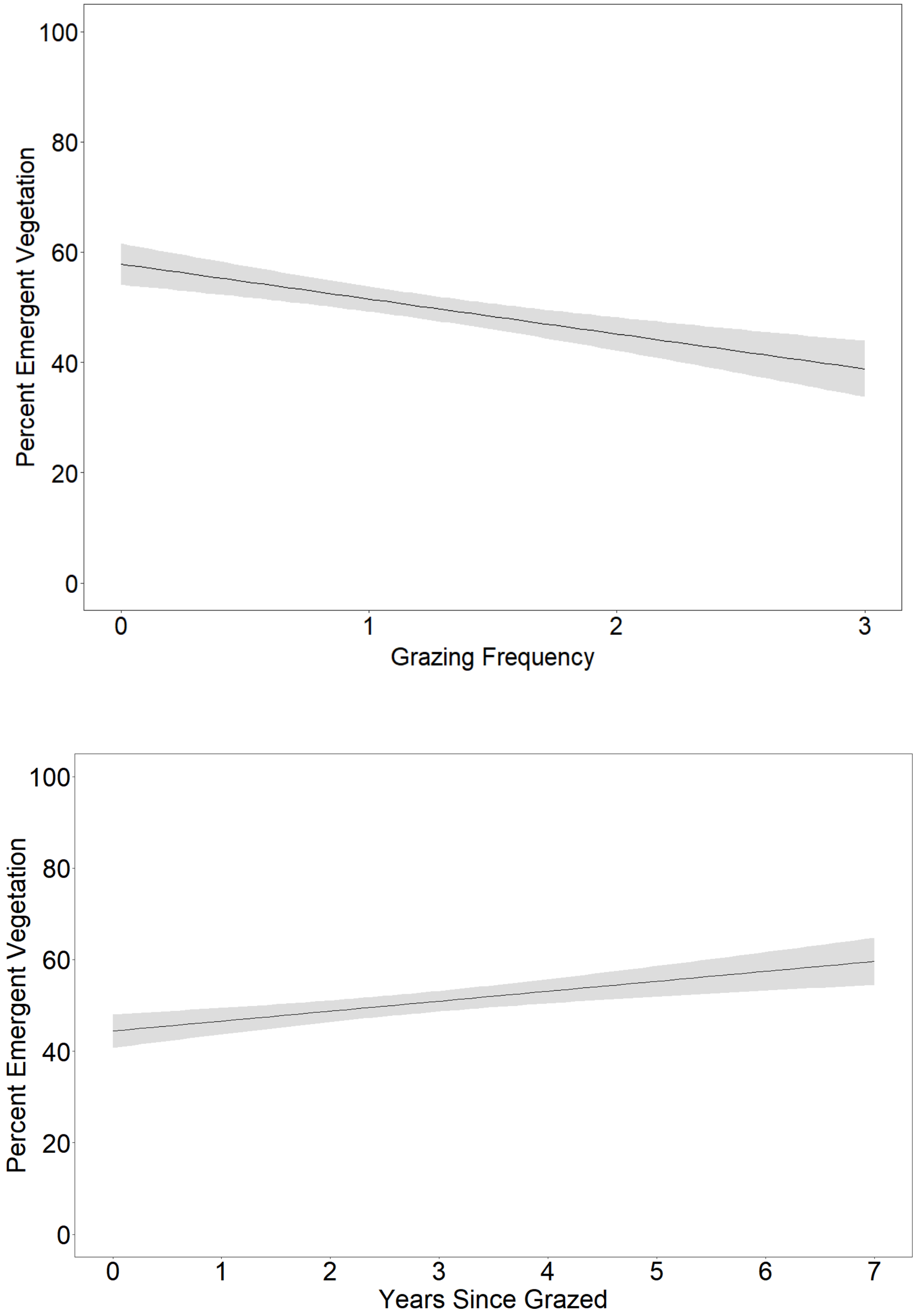
Emergent Wetland Vegetation

The top model predicting percent emergent vegetation included additive effects of wetland class and grazing frequency (Table 2). Percent emergent wetland vegetation was negatively associated with increasing grazing frequency (Figure 1). The model with wetland class and years since grazed was also competitive (ΔAIC = 2.12), indicating that recentness of grazing was also an important predictor of percent emergent vegetation, with percent vegetation showing a positive relationship to years since grazed. There is some evidence that grazing effects vary among wetland classes, but while it is a competitive model, the grazing effect was not strong enough to overcome the 4-parameter penalty to AIC.

Table 2. Model results for percent emergent vegetation analysis including number of parameters (K) and ΔAIC.

Model	**K**	**ΔAIC**
Grazing frequency + Wetland class	6	0
Years since grazed + Wetland class	6	2.124
Grazing frequency * Wetland class	10	3.176
Years since grazed * Wetland class	10	3.562
Wetland class	5	16.744
Grazing frequency	2	128.034
Years since grazed	2	132.606

Figure 1. Relationship between percent emergent vegetation and grazing. Top figure presents response of emergent cover to grazing frequency (the number of times a site was grazed in the last 3 years), whereas bottom figure shows response to time since grazed. The shaded area represents an 85% confidence interval.



Secretive Marsh Birds

We surveyed 86 wetlands at 50 sites across 37 WPAs in 2017 and 89 wetlands at 53 sites across 41 WPAs in 2018. We completed 350 marsh bird surveys between the 2 field seasons. We detected 579 individuals of our target species with Soras detected on 76 sites, Pied-billed Grebes on 57, Virginia Rails on 45, American Bitterns on 25, Least Bitterns on 11, and Yellow Rail on 0. We were able to model detection probability and occupancy of Sora, Pied-billed Grebe, Virginia Rail, and American Bittern, but there were insufficient detections of Least Bittern and Yellow Rail to perform analysis.

Vocalization Probability: The 4 species of marsh birds responded differently to broadcasted calls (Table 3). Detection probability of American Bittern varied between survey periods and years, but not among survey minutes, with a constant probability of calling during each survey minute. The top vocalization probability model for Pied-billed Grebe allowed for variation within survey minute, survey period, and survey year. Within surveys, Pied-billed Grebe detection was greatest following their own broadcast call and consistently low during both the initial passive listening period and all other species' calls. The top vocalization probability model for Sora included variation within survey minutes, with 7 modeled vocalization probabilities within a survey: detection remained consistently low during the passive listening period and varied across the 6 minutes of broadcasted calls and was greatest following their own calls. The top model for Virginia Rail included variation within survey minute and survey period, with 7 vocalization probabilities within a survey, showing a similar response to broadcasted calls as Soras.

Table 3. Vocalization probability models for the four species of secretive marsh birds analyzed. Each model parameter represents a different vocalization probability and varied by minute, visit, and/or year.

Species	**Equation**
American Bittern	$\beta_0 + \beta_{visit} + \beta_{year}$
Pied-billed Grebe	$\beta_0 + \beta_{min\ 1\text{-}9,11*visit*year} + \beta_{min\ 10*visit*year}$
Sora	$\beta_0 + \beta_{min1\text{-}5} + \beta_{min6} + \beta_{min7} + \beta_{min8} + \beta_{min9} + \beta_{min10} + \beta_{min11}$
Virginia Rail	$\beta_0 + \beta_{min1\text{-}5} + \beta_{min6} + \beta_{min7} + \beta_{min8} + \beta_{min9} + \beta_{min10} + \beta_{min11}$

Occupancy: Grazing helped explain variation of occupancy in all 4 species of marsh birds (Table 4). Both years since grazed and grazing frequency improved model performance for American Bittern, though AIC indicates that years since grazed was a better predictor of occupancy. Sites were less likely to be occupied by American Bittern the longer they went without grazing (Figure 2), and more frequently grazed sites were more likely to be occupied (Figure 3). Virginia Rail responded positively to increased grazing frequency, and Sora responded negatively to increasing years since grazed. The grazing variable on the extinction parameter ($\varepsilon_{2\text{-}3}$) was only present in the top model for Pied-billed Grebe (Figure 4). We could not include the parameter in the Virginia Rail model as it was inestimable, likely a result of a lack of data. Sites that had been grazed the previous year were more likely to experience local extinction by Pied-billed Grebe between 2017 and 2018.

Table 4. Model results for occupancy of four species of secretive marsh birds including number of parameters (K) and ΔAIC_c.

Species	**Model**	**K**	**ΔAIC_c**
American Bittern	Base model + Years since grazed	23	0.0
	Base model + Years since grazed^2	24	0.5037
	Base model + Grazing frequency	23	3.2852
	Base model	22	4.693
	Base model + Grazed last year	23	6.9523
Pied-billed Grebe	Base model + Grazed last year	27	0.0
	Base model	26	0.4624
	Base model + Years since grazed^2	28	1.9581
	Base model + Years since grazed	27	2.7736
	Base model + Grazing frequency	27	2.7751
Sora	Base model + Years since grazed	26	0.0
	Base model	25	2.1655
	Base model + Years since grazed^2	27	2.3423
	Base model + Grazing frequency	26	3.2717
	Base model + Grazed last year	26	4.409
Virginia Rail	Base model + Grazing frequency	33	0.0
	Base model	32	2.4258
	Base model + Years since grazed^2	34	3.6400
	Base model + Years since grazed	33	3.9607

Figure 2. Marsh bird occupancy versus time since grazed, using the best supported linear (AMBI, SORA) or quadratic model (PBGR, VIRA). Lines represent mean predicted occupancy, whereas dots represent predicted values for each survey site, given other unique combinations of covariates.

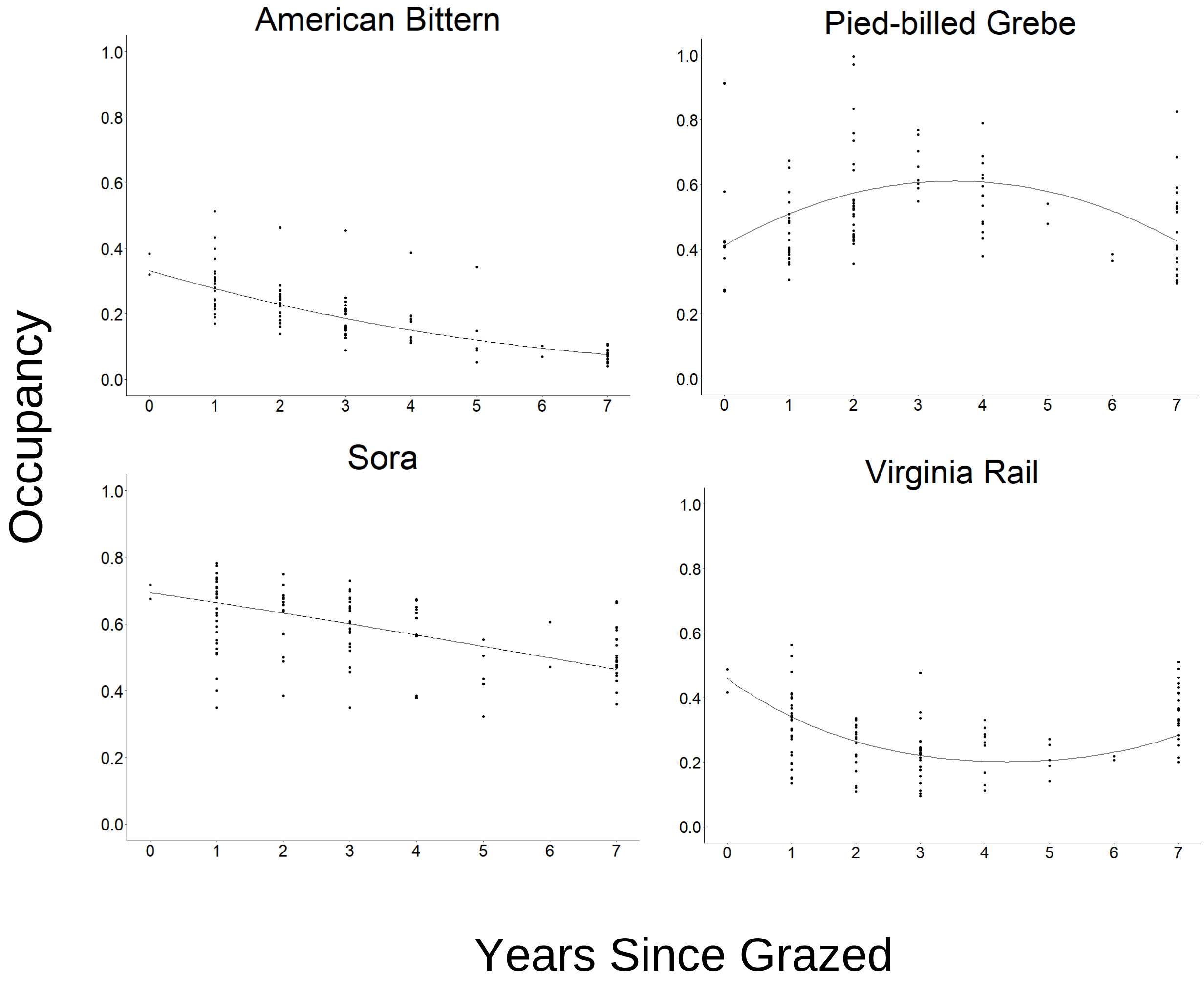


Figure 3. Marsh bird occupancy response to grazing frequency (number of times grazed in the last three years). Lines represent mean predicted occupancy, whereas dots represent predicted values for each survey site, given other unique combinations of covariates.

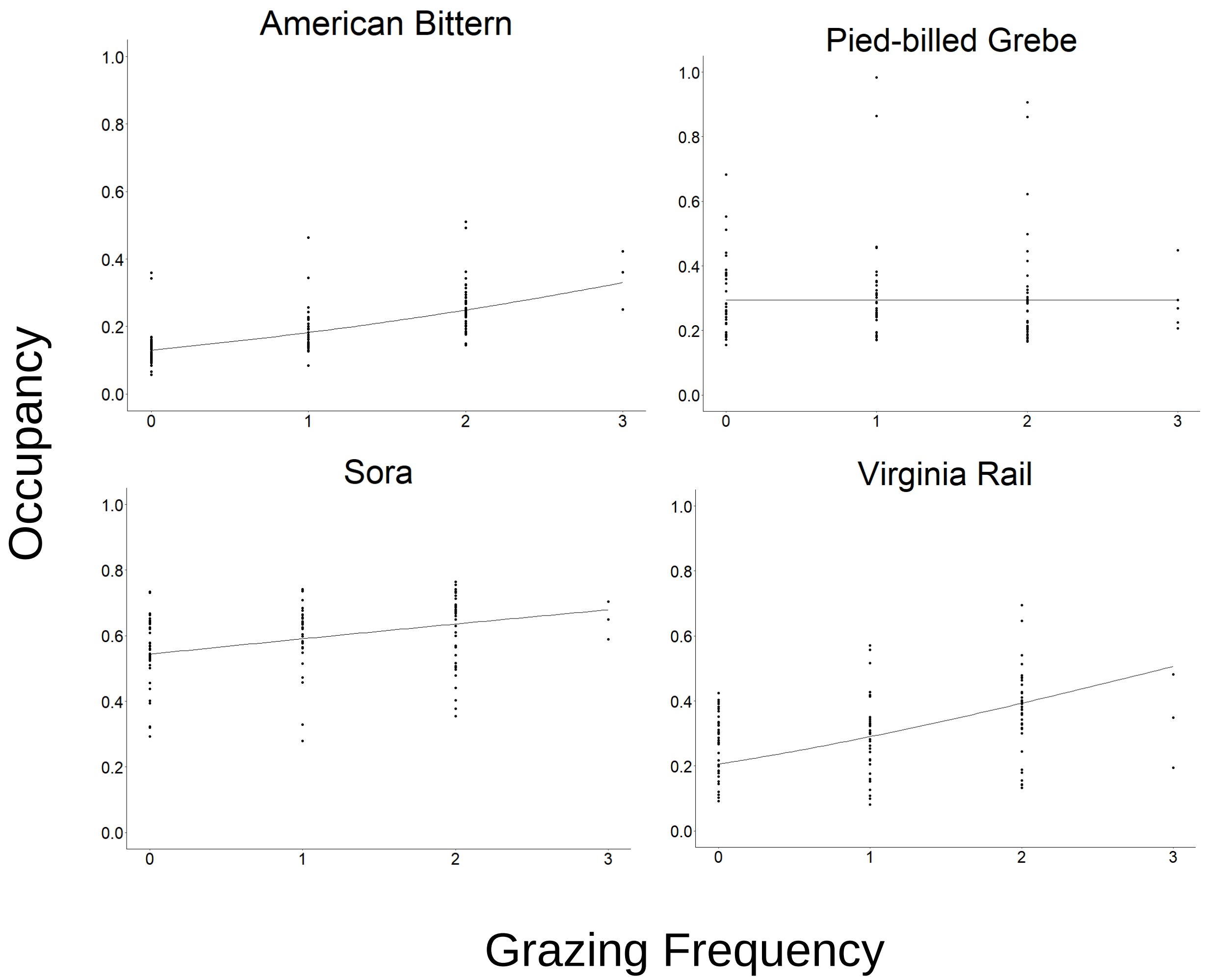
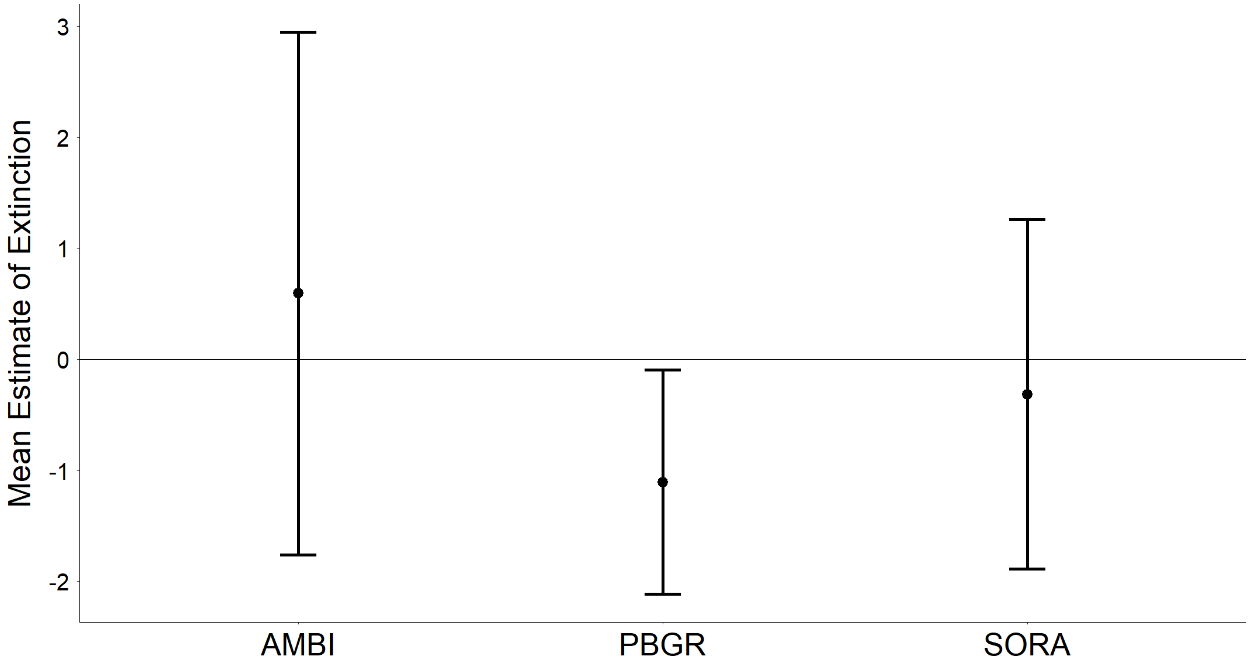


Figure 4. Effect of grazing in the previous year on between-year local extinction probability. Negative estimates indicate a reduction in extinction probability when sites were grazed last year. VIRA not included, because parameter was inestimable.



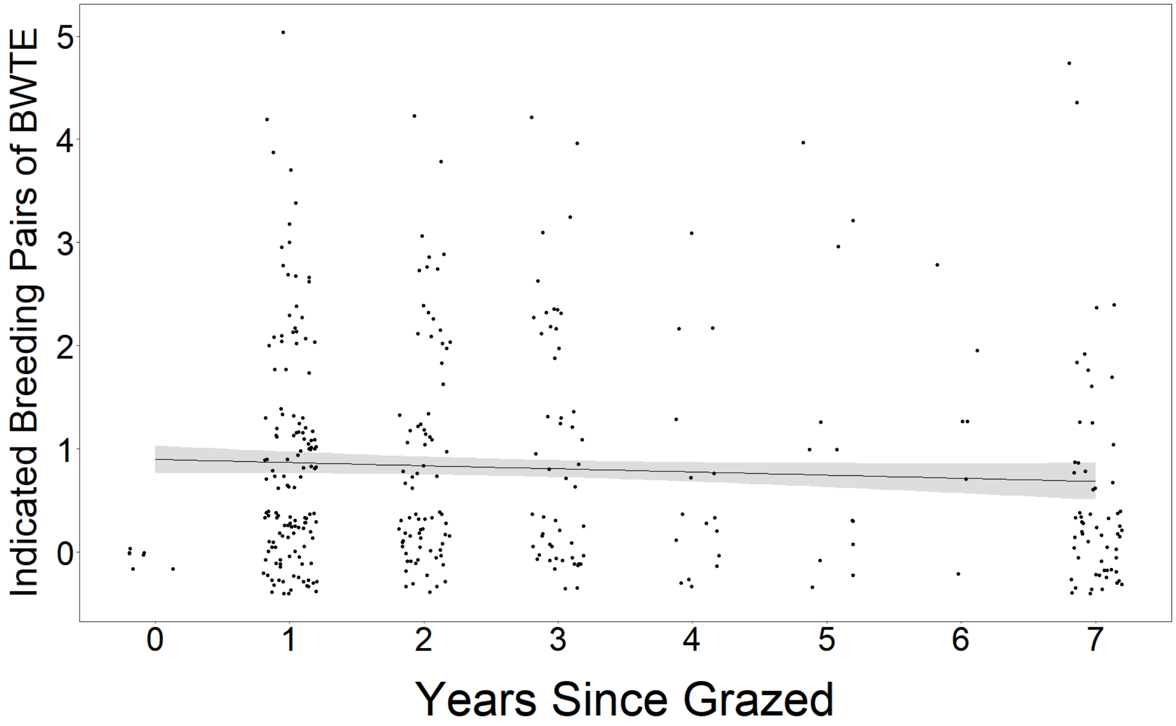
Ground-nesting waterfowl

We detected 916 Blue-winged Teal, 705 Mallard, 60 Gadwall, 55 Northern Shovelers, and 17 Green-winged Teal indicated breeding pairs and 70 Blue-winged Teal, 30 Mallard, 4 Gadwall, and 13 unknown broods. We were able to model pair abundance of Blue-winged Teal and Mallard, but there were insufficient detections of other species to perform analysis. Grazing helped explain variation in abundance for Mallard pairs with years since grazed included in the top model (Table 5). Mallard pair abundance increased as years since grazed increased (Figure 5). Grazing did not improve model performance for Blue-winged Teal pairs, and the top model was the base model. Grazing frequency had no effect on pair abundance of either species (Figure 6). We only had sufficient data to analyze Blue-winged Teal broods; however, we combined all dabbler broods observed (Blue-winged Teal, Gadwall, Mallard, and unknown dabblers) and fit our brood models to the whole group. Grazing did not help explain variation in brood abundance for Blue-winged Teal or our combined group of all dabblers (Table 5).

Table 5. Model results for abundance of two species of indicated breeding pairs including number of parameters (K) and ΔAIC.

Species	**Model**	**df**	**ΔAIC**
Blue-winged Teal pairs	Base model	15	0.0
	Base model + Years since grazed	16	1.3721
	Base model + Grazing frequency	17	1.9913
	Base model + Years since grazed^2	16	3.3356
Mallard pairs	Base model + Years since grazed	15	0.0
	Base model + Years since grazed^2	16	0.2965
	Base model	14	1.8512
	Base model + Grazing frequency	15	2.7131
All dabbler broods	Base model	12	0.0
	Base model + Years since grazed	13	1.3129
	Base model + Grazing frequency	13	1.9968
	Base model + Years since grazed^2	14	3.2943
Blue-winged Teal broods	Base model	12	0.0
	Base model + Years since grazed	13	1.5837
	Base model + Grazing frequency	13	1.7986
	Base model + Years since grazed^2	14	3.3581

Figure 5. Blue-winged Teal and Mallard pair response to years since grazed, using the best-supported linear model. Lines represent mean predicted abundance, whereas dots represent predicted values for each survey site, given other unique combinations of covariates.



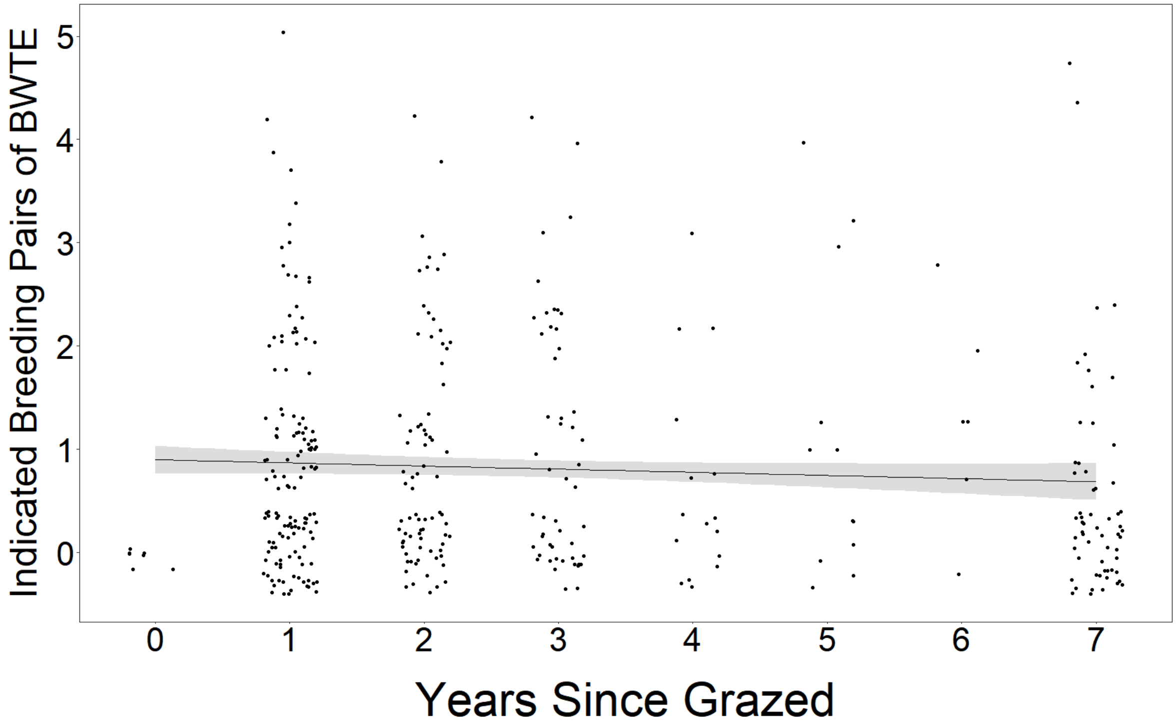
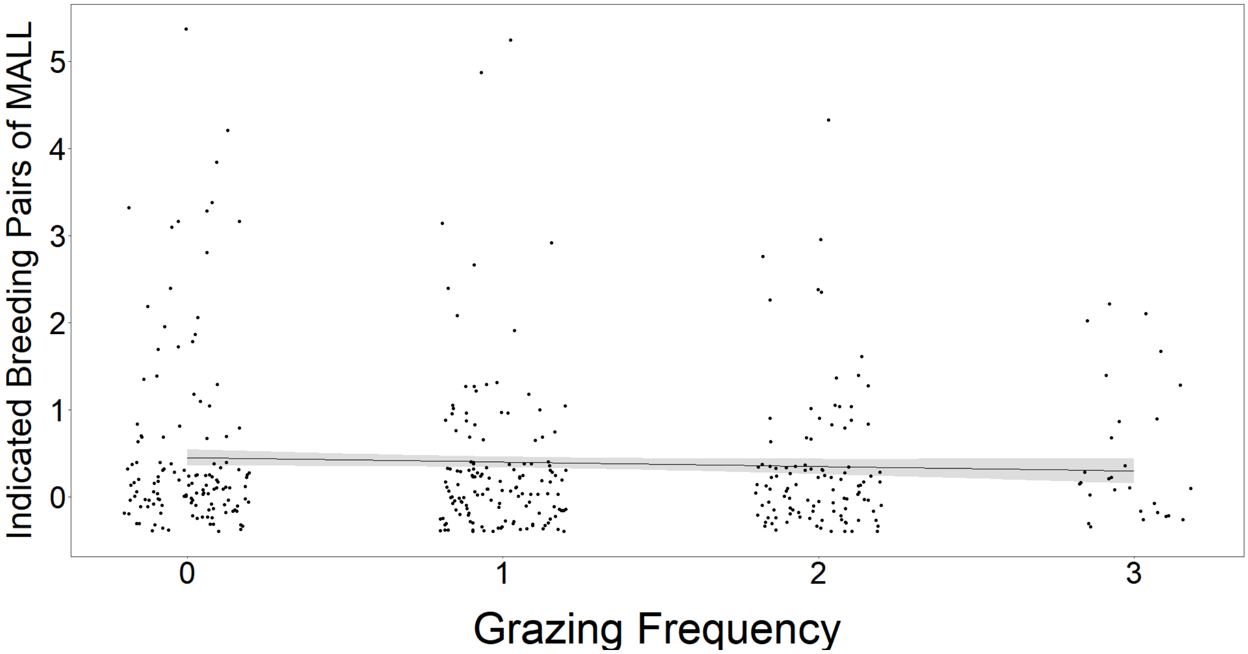
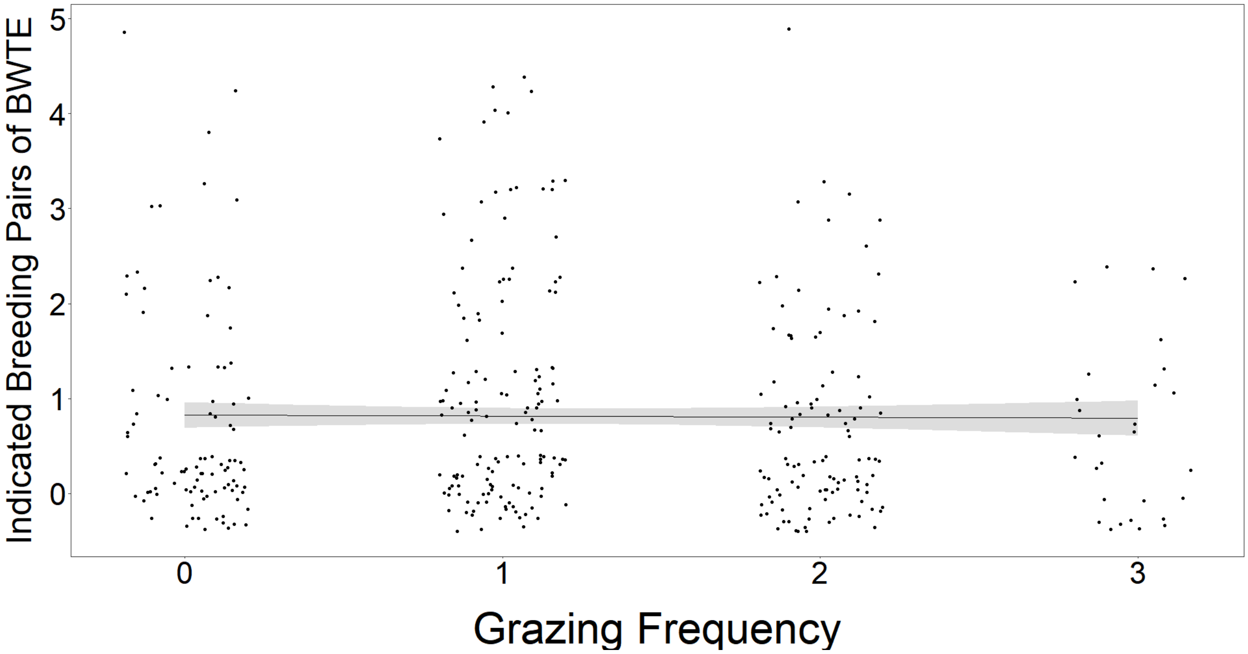


Figure 6. Blue-winged Teal and Mallard pair response to grazing frequency (number of times grazed in the last three years). Lines represent mean predicted abundance, whereas dots represent predicted values for each survey site, given other unique combinations of covariates.





DISCUSSION

Conservation grazing decreased the percentage of emergent vegetation in study wetlands. In addition, the longer a site had gone undisturbed by grazing, the more vegetated wetlands became. While this change is statistically significant, our results show that conservation grazing at this low intensity is unlikely to drastically alter the percent of emergent vegetation covering a wetland. A modest reduction can be expected in response to trampling and grazing on emergent vegetation, exposing open water and increasing interspersion of water and vegetation. Additional research into the impact of grazing frequency on water quality would give us more information on how wetlands respond to conservation grazing.

A modest decrease of emergent vegetation is likely appropriate for most secretive marsh bird species, who prefer wetlands dominated by vegetation (Gibbs et al. 1991, Lor and Malecki 2006). Both years since grazed and grazing frequency improved model performance beyond the base model for American Bitterns, though years since grazed was a slightly more important occupancy predictor. While very little is known about the ecology of this secretive marsh bird, we do know that they prefer shallower wetlands with significant habitat interspersion (Lowther et al. 2009). Grazing that opened up heavily choked wetlands and increased interspersion would understandably be beneficial for these birds. While we do not have evidence to support that Soras were significantly affected by increased grazing frequency, they did show decreased probability of occupancy the longer a site went undisturbed by grazing. Given that the results of our emergent vegetation model showed that grazing only modestly reduced wetland vegetation even when performed multiple years in a row, it is unlikely that 2 or 3 consecutive years of light grazing reduces wetland vegetation cover to such a degree that wetlands would become unattractive to Soras. Meanwhile, idle wetlands become increasingly choked by dense vegetation and may become less attractive to Soras. While our 2 grazing variables are highly correlated, only grazing frequency was an important predictor of Virginia Rail occupancy. This could be a result of their preference for earlier-successional wetlands that do not have an accumulation of residual standing biomass (Conway 1995). Grazing multiple years within a short span likely increases the impact of trampling and grazing on emergent vegetation, more effectively controlling accumulation of biomass and maintaining a wetland that does not impede movement of Virginia Rails.

Pied-billed Grebes have remarkably different habitat preferences than the other 3 species of marsh birds analyzed in this paper. They prefer densely vegetated wetlands interspersed with a considerable amount of open water (Weller and Spatcher 1965, Bolenbaugh et al. 2011). As diving marsh birds, Pied-billed Grebes forage for fish and invertebrates and escape danger by diving and thus prefer deeper wetlands (Muller and Storer 1999, Baschuk et al. 2012). Given that these deeper wetlands are less at risk of being dominated by invasive emergent vegetation, it is unsurprising that grazing frequency or years since grazed did not influence occupancy of Pied-billed Grebe. Since we did see a modest impact of grazing on percent emergent vegetation, it is possible that grazing at higher intensities could trample perimeter vegetation that grebes rely upon for cover. However, our results indicate that, at a low intensity, increased grazing frequency and years since grazed did not impact Pied-billed Grebe occupancy. We did report a reduction in local extinction at sites that were grazed the previous year. While we cannot explain why we saw this effect, ΔAIC for the base model was only 0.4624, indicating that the grazing effect was minor. Given that years since grazed was not an important occupancy predictor, perhaps the negative grazing effect was short-lived.

Grazing did not impact abundance of Blue-winged Teal pairs, suggesting that at this low intensity of grazing, cattle were not altering vegetation to a degree that wetlands become more or less attractive to teal breeding pairs. And while Mallard pair abundance increased the longer a site went undisturbed, increased grazing frequency had no observable effect. When we looked at Blue-winged Teal individually or all dabbler species combined, we saw no evidence that grazing frequency or years since grazed influenced brood abundance. Our study suggests that grazing at least once every three years at a low intensity does not negatively affect pair or brood abundance of dabbling ducks.

While our results indicate that increased grazing frequency at a low intensity is not harmful to marsh bird occupancy or ground-nesting duck pair and brood abundance, a few caveats are in order. Drier grassland ecosystems recover from grazing more slowly and so our results should be applied cautiously in more arid habitats such as mixed or short-grass prairies. Additionally, our study only analyzed effects of late-spring, low intensity grazing, and we cannot assume similar results on the tallgrass prairie when different grazing prescriptions are applied. This suggests that our results apply primarily to conservation land managers. For livestock producers, these low stocking rates are likely impractical. Additional research is needed to investigate how grazing frequency and years since grazed influences waterbirds at higher stocking rates. Past studies have found that mesic grasslands that evolved with heavy grazing pressures can sustainably support moderate levels of grazing (Fuhlendorf and Engle 2001), so it is possible that Minnesota's tallgrass prairie could support higher stocking rates without negatively influencing waterbird populations; however, this requires additional research.

Management Implications

Management of restored and native prairies frequently focuses on upland management (U.S. Fish and Wildlife Service 2017). But in grassland systems like the Prairie Pothole Region, wetland habitat is vital to the survival of many species, and upland management could have unintended impacts. More land managers are using conservation grazing as an alternative or supplement for burning to enhance upland prairie habitat for wildlife. While much is still unknown about how grazing at lower intensities is impacting wetlands, our study suggests that secretive marsh birds and ground-nesting waterfowl are not negatively impacted by increased grazing frequency. If land managers in the tallgrass prairie are unable to burn as regularly as needed, then our results suggest that the use of frequent grazing at low intensities to manage upland cover should neutrally or positively impact occupancy of secretive marsh birds and abundance of ground-nesting waterfowl pairs and broods. Additionally, the longer land managers go without a disturbance on the land, the more dominated wetlands become by vegetation, negatively impacting some species of marsh birds. If land managers are considering secretive marsh birds in their management plans, we do not recommend idling pastures for 6+ years.

Literature Cited

Ahlering M. A., D. H. Johnson, and L. H. Elliott. 2018. Land Ownership and Use Influence Grassland Bird Abundance. The Journal of Wildlife Management 83(2):343–355.

Anderson, R. C. 1990. The historic role of fire in the North American grassland. Pages 8 – 18 in S. L. Collins and L. L. Wallace, editors. Fire in North American Tallgrass Prairies. The University of Oklahoma Press, Norman, USA.

Bakker, K.K. 2003. The effect of woody vegetation on grassland nesting birds: An annotated bibliography. The Proceedings of the South Dakota Academy of Science 82:119−141.

Baschuk, M. S., N. Koper, D. A. Wrubleski, and G. Goldsborough. 2012. Effects of water depth, cover and food resources on habitat use of marsh birds and waterfowl in boreal wetlands of Manitoba, Canada. Waterbirds 35:44-55.

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

Briske, D. D., J. D. Derner, D. G. Milchunas, and K. W. Tate. 2011. An evidence-based assessment of prescribed grazing practices. Pages 23–74 *in* D. D. Briske, editor. Conservation benefits of rangeland practices: assessment, recommendations, and knowledge gaps. United States Department of Agriculture, Natural Resources Conservation Service, Washington, D.C., USA.

Collins, S. L. 1990. Introduction: fire as a natural disturbance in tallgrass prairie ecosystems. Pages 3 – 7 *in* S. L. Collins, and L. L. Wallace, editors. Fire in North American Tallgrass Prairies. The University of Oklahoma Press, Norman, USA.

Collins, S. L., and D. J. Gibson. 1990. Effects of fire on community structure in tallgrass and mixed-grass prairie. Pages 81 – 98 *in* S. L. Collins, and L. L. Wallace, editors. Fire in North American Tallgrass Prairies. The University of Oklahoma Press, Norman, USA.

Conway, C. J. 1995. Virginia Rail (*Rallus limicola*). The Birds of North America (A. F. Poole and F. B. Gill, Ed.). Cornell Lab of Ornithology, Ithaca, NY, USA. <https://doi.org/10.2173/bna.173>. Accessed 15 June 2019.

Conway, C. J., and J. P. Gibbs. 2005. Effectiveness of call-broadcast surveys for monitoring marsh birds. The Auk 122(1):26–35.

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

Devries, J. H., and L. M. Armstrong. 2011. Impact of management treatments on waterfowl use of dense nesting cover in the Canadian Parklands. The Journal of Wildlife Management 75:1340-1349.

Fuhlendorf, S. D., and D. M. Engle. 2004. Application of the fire-grazing interaction to restore a shifting mosaic on tallgrass prairie. Journal of Applied Ecology 41:604-614.

Gibbs, J. P, J. R. Longcore, D. G. McAuley, and J. K. Ringelman. 1991. Use of wetland habitats by selected nongame water birds in Maine. U.S. Fish and Wildlife Service. Fish and Wildlife Resource Publication 9, Washington, D.C., USA.

Grace J. B. 1998. Can prescribed fire save the endangered Coastal Prairie ecosystem from Chinese Tallow invasion? Endangered Species Update 15:70-76.

Genet, J. 2015. Status and trends of wetlands in Minnesota: depressional wetland quality assessment (2007–2012). Minnesota Pollution Control Agency, Saint Paul, MN.

Harrison R. B., W. M. Jones, D. Clark, B. A. Heise, and L. H. Fraser. 2017. Livestock grazing in intermountain depressional wetlands: effects on breeding waterfowl. Wetlands Ecology and Management. DOI: 10.1007/s11273-017-9529-z

Heisler J. L., J. M. Briggs, and A. K. Knapp. 2003. Long-term patterns of shrub expansion in a C4-dominated grassland: fire frequency and the dynamics of shrub cover and abundance. American Journal of Botany 90:423–428.

Johnson, R. R. 1984. Breeding habitat use and postbreeding movements by Soras and Virginia Rails. Thesis, Iowa State University, Ames, USA.

Johnson D. H. 1997. Effect of fire on bird populations in mixed-grass prairie. Pp. 181-206 in Ecology and Conservation of Great Plains Vertebrates (Knopf FL, Samson FB, eds.). Springer, New York.

Kantrud, H. A. 1986. Effects of vegetation manipulation on breeding waterfowl in prairie wetlands—a literature review. U.S. Fish and Wildlife Service. Fish and Wildlife Technical Report 3, Washington, D.C., USA.

Lor S., and R. A. Malecki. 2002. Call-response surveys to monitor marsh bird population trends. Wildlife Society Bulletin 30:1195–1201.

Lowther, P. E., A. F. Poole, J. P. Gibbs, S. M. Melvin, and F. A. Reid. 2009. American Bittern (*Botaurus lentiginosus*). The Birds of North America (A. F. Poole, Ed.). Cornell Lab of Ornithology, Ithaca, NY, USA. <https://doi.org/10.2173/bna.18>. Accessed 15 June 2019.

Ma, Z., Y. Cai, B. Li, and J. Chen. 2010. Managing wetlands habitats for waterbirds: an international perspective. Wetlands 30:15-27.

McKenzie, D. I., J. D. Nichols, J. E. Hines, M. G. Knutson, and A. B. Franklin. 2003. Estimating site occupancy, colonization, and local extinction when a species is detected imperfectly. Ecology 84:2200-2207.

Minnesota Prairie Plan Working Group. 2018. Minnesota Prairie Conservation Plan. Minnesota Prairie Plan Working Group, Minneapolis, Minnesota, USA.

Muller, M. J., and R. W. Storer. 1999. Pied-billed Grebe (*Podilymbus podiceps*). The Birds of North America (A. F. Poole, and F. B. Gill, Ed.). Cornell Lab of Ornithology, Ithaca, NY, USA. <https://doi.org/10.2173/bna.410>. Accessed 14 June 2019.

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.

Morris, K., and P. Reich. 2013. Understanding the relationship between livestock grazing and wetland condition. Arthur Rylah Institute for Environmental Research Technical Report Series No. 252. Heidelberg, Victoria: Department of Environment and Primary Industry.

Pederson, R. L. 2006. Disturbance and grassland ecosystems. Pages 16-34 *in* B. Hohman, editor. Migratory bird responses to grazing. U.S. Department of Agriculture, Natural Resources Conservation Service Technical Report 190–54.

Richmond O. M. W., J. Techkin, and S. R. Beissinger. 2012. Impact of cattle grazing on the occupancy of a cryptic, threatened rail. Ecological Applications 22:1655–1664.

Samson F. B., and F. L. Knopf. 1994. Prairie conservation in North America. Bioscience 44:418-421.

Schieltz, J. M., and D. I. Rubenstein. 2016. Evidence based review: positive versus negative effects of livestock grazing on wildlife. What do we really know? Environmental Research Letters 11:1-18.

Shaw, S. P., and C. G. Fredine. 1956. Wetlands of the United States. U. S. Fish and Wildlife Service, Circular 39.

Thompson S. J., T. W. Arnold, J. Fieberg, D. A. Granfors, S. Vacek, and N. Palaia. 2016. Grassland birds demonstrate delayed response to large-scale tree removal in central North America. Journal of Applied Ecology 53:284–294.

U.S. Fish and Wildlife Service. 2012. Morris Wetland Management District Habitat Management Plan, Morris, Minnesota, USA.

U.S. Fish and Wildlife Service. 2017. Morris Wetland Management District Annual Narrative Report: January 1, 2016 – December 31, 2016, Morris, Minnesota, USA.

Weller M. W., and C. S. Spatcher. 1965. Role of habitat in the distribution and abundance of marsh birds. Agriculture and Home Economics Experiment Station Special Report 43. Ames, Iowa: Iowa State University of Science and Technology.

White, G. C., and K. P. Burnham. 1999. Program MARK: survival estimation from populations of marked animals. Bird Study 46:120-138.