CONSERVATION GRAZING WITH CATTLE: EVALUATING CONTROL OF WOODY ENCROACHMENT, PLANT COMMUNITY CHANGE AND GRASSLAND BIRD HABITAT

by

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Chapter 1: Conservation grazing with cattle and control of woody encroachment in temperate cool-season grasslands in Wisconsin

Abstract

Woody encroachment threatens grasslands worldwide, which in turn threatens grassland obligate wildlife. Mowing and herbicides are among the tools used to combat encroachment. Livestock grazing is another potential tool, although grazing methods vary widely and can sometimes exacerbate encroachment. We tested the potential to reduce woody plant cover and density with rotational grazing of cattle at low stocking density in three temperate cool-season grasslands. We implemented a completely randomized design on each of two pre-treatment levels of woody cover. Seasonal grazing was applied alone and in combination with two one-time woody suppression techniques (mowing and herbicide) applied in 2016. Each subsequent spring (2017 through 2019) we measured the response in total woody cover and stem size and density of individual woody species. We analyzed the most common species at each site: Quaking aspen (*Populus tremuloides* Michx.), white meadowsweet (*Spiraea alba* Du Roi), gray dogwood (*Cornus foemina* Mill.), hybrid bush honeysuckle (*Lonicera* x *bella* Zabel) and prickly ash (*Zanthoxylum americanum* Mill.). In high initial woody cover (20 to 50%), foliar herbicide followed by rotational grazing significantly reduced woody cover at every site, mowing followed by rotational grazing reduced it at two sites and grazing alone reduced it at one site after two to three years (cover ≤ 17%; $P < 0.03$). In low initial woody cover (5 to 20%), foliar herbicide followed by rotational grazing and mowing followed by rotational grazing reduced woody cover

at only one site after two years (cover ≤ 3%; $P < 0.01$). Stem densities of common woody species responded differently to treatments highlighting the need for vigilant, species level monitoring of woody plants following treatment. Conservation grazing by cattle in combination with herbicides has potential to augment woody control efforts in temperate cool-season grasslands, but further evaluation is needed.

Introduction

Grasslands are critical habitat for grassland obligate wildlife, such as grassland birds. In areas of North America, more than 99% of the original grasslands (existing prior to European settlement) have been lost, primarily to intensive agricultural use (Samson and Knopf, 1994) and grasslands are considered a critically endangered ecosystem (> 98% loss in the United States) (Noss *et al.*, 1995). In tandem with this loss, grassland bird populations have declined steeply and continue to decline (Rosenberg *et al.*, 2019). Some of these declining species require large grasslands (> 100 ha) (Sample and Mossman, 1997) and many land managers recognize the need for large, contiguous grasslands; however, there is evidence that even the largest remaining grasslands in North America may not be adequate to halt population declines (With *et al.*, 2008). This indicates the need to preserve existing grasslands, expand them where possible, and restore degraded ones to maximize the area of contiguous grassland habitat.

In the Upper Midwest region of the United States, many declining species, such as the Henslow's sparrow (*Ammodramus henslowii* Audobon) and the bobolink (*Dolichonyx oryzivorus* L.) rely on grasslands for habitat. In Wisconsin the greater prairie-chicken (*Tympanuchus cupido*

L.), a state threatened species, has one of its largest remaining populations in the Buena Vista Wildlife Area, which is one of the most extensive conservation grasslands east of the Mississippi River (5,000 ha). We define conservation grasslands as land dominated by herbaceous grass vegetation with minimal presence of woody plants, where managers prioritize wildlife habitat and preserving biodiversity over agricultural production.

One major threat to conservation grasslands is the encroachment of woody plants, including both indigenous and exotic woody species (Archer, 1995). Grasslands originated under the influence of periodic drought, frequent fire, and mammalian grazers (Anderson, 2006) and human activity has and continues to alter all of these factors, thereby altering the grassland ecosystem. While it is difficult to disentangle these influences on this altered ecosystem and identify direct causes of woody encroachment, several factors have been implicated or associated with increases in woody cover, including reduced fire frequency (Nowacki and Abrams, 2008), heavy livestock grazing (Archer, 1995), a warming climate (Buffington and Herel, 1965), and increased atmospheric CO_2 (Van Auken, 2000). The spread of exotic and invasive woody plants has also altered the grassland ecosystem. In the Midwest, for example, exotic shrubs such as common buckthorn (*Rhamnus cathartica* L.) and hybrid bush honeysuckle (e.g. *Lonicera x bella* Zabel) exhibit earlier leaf emergence in the spring and extended leaf longevity in the fall compared to indigenous plant species (Harrington *et al.*, 1989) likely giving them a competitive advantage in grassland habitats. These species and others are already widespread in forests and are likely to continue to spread into grasslands (Fan *et al.*, 2018).

Grassland bird species are negatively impacted by increases in woody vegetation on the landscape (Ribic and Sample, 2001; Kates, 2005); however, slowing or reversing woody encroachment is complex and difficult. Ratajczak et al. (2014) found that the transition from mesic grassland to shrubland was non-linear, with abrupt increases over time, and suggested keeping shrub cover < 6% to prevent conversion to shrublands. Positive feedback likely plays a role in woody encroachment (especially for deep-rooted and clonal species) because woody growth reduces herbaceous growth, thereby reducing subsequent fire intensity (Ratajczak *et al.*, 2011) and some have observed that once shrubs become established in conservation grasslands, their cover increases even under frequent application of fire (Heisler *et al.*, 2003; Briggs *et al.*, 2005). Birds also contribute to positive feedback in woody encroachment via dispersal because they use existing woody plants as perching sites and cause 'seed rain' of additional woody species (Prather *et al.*, 2017). This indicates that there may be a threshold beyond which it becomes increasingly difficult to remove woody species from grasslands and suggests that management in low cover, before woody plants become well established, is preferable.

Given that the rate of encroachment may increase with increasing woody cover, the aggressive growth of exotic and invasive woody plants and the needs of declining grassland wildlife species, identifying management techniques that can halt or reverse increases in woody cover are critically important. Fire, herbicide and mowing are widely-used and effective strategies for reducing woody cover (Bragg and Hulbert, 1976; Lett and Knapp, 2005), yet woody encroachment continues. This may be due to logistical and resource constraints in applying these management methods at an appropriate frequency and scale, woody plant cover that is already beyond the threshold, or a combination of these and other factors. Irrespective of the cause, identifying additional effective management methods that can be used alone or in combination with existing methods would be beneficial.

Grazing has been proposed as such a method because it can be used to meet defined vegetation and landscape goals. When timing, frequency, intensity and selectivity of grazing are closely managed and optimized, the term often used is ‘targeted grazing’ (Launchbaough and Walker, 2006). Targeted grazing has been proposed as an alternative to mowing, fire and herbicide (Bailey *et al.*, 2019). ‘Conservation grazing’ is similar to targeted grazing in its focus on vegetation and landscape goals, but is a less intensive type of management generally applied to natural or seminatural ecosystems, where preservation of biodiversity and wildlife habitat are the primary management goals (Bailey *et al.*, 2019). While targeted grazing alone may be able to accomplish some conservation goals—especially those related to woody plant suppression—the intensive level of management involved and reliance on browsing livestock species such as goats and sheep instead of cattle (which are more readily available) make it difficult to apply at a large scale under resource constraints. Furthermore, targeted grazing raises animal welfare concerns. High stocking densities are often used to encourage consumption of targeted plants, which may be toxic to the animal due to secondary metabolites (Estell, 2010). Conservation grazing at a low stocking density may avoid this issue and present lower risk to the livestock owner. These factors make conservation grazing with cattle a practical option to apply at a larger scale.

There is little evidence, however, to show that cattle grazing alone can reduce woody plants. One study across northern temperate grasslands found that long-term cattle grazing reduced woody cover, though this effect was only observed on mesic sites and not on dry-mesic grasslands (Lyseng *et al.*, 2018). A study in southern Wisconsin using Scottish Highland cattle—a breed known to browse woody vegetation—found that grazing resulted in lower woody stem densities after two years, though the response of individual woody species to grazing was not consistent. The authors concluded that grazing could be an important supplement to management with fire but not replace it (Harrington and Kathol, 2009). While these studies document woody suppression with cattle grazing, many exceptions were presented. This suggests that conservation grazing with cattle needs to be employed in conjunction with additional vegetation management strategies in order to consistently reduce woody vegetation. This is not a new idea. Masters and Sheley (2001) suggest that a multi-faceted approach that uses many management methods (e.g., herbicide, grazing, mowing, tillage, root removal, fire and reseeding) over time is ideal for meeting conservation goals. Conservation grazing, when used in combination with other vegetation management methods, could also ameliorate some of the drawbacks of other methods such as reducing the cost or reducing non-target impacts (e.g. herbicides) (Shepard *et al.*, 2004). Though conservation grazing requires an initial investment of time and resources (e.g. installing fencing and water sources) it could reduce costs over time if use of other management methods is reduced.

In the Midwest there is interest in reducing woody species in conservation grasslands. However, many conservation grassland managers cannot use prescribed fire as a management method due to factors such as budget and time constraints. Stakeholders have interest in exploring if rotational grazing can replace this management technique. Rotational grazing— rather than continuous or set-stock grazing— is of interest because it could facilitate regrowth of palatable forage species (Paine *et al.*, 1999), benefit soil health (Teague *et al.*, 2011), and potentially allow for homogenous trampling or browsing (i.e. impact) on woody species.

While grazing and woody encroachment has been studied extensively in the western and central regions of the United States, there is little evidence of the impacts of rotational conservation grazing on temperate cool-season grasslands in the Midwest and whether it can provide woody plant control when combined with mowing or herbicides. The objective of this study was to test whether one-time woody plant control treatments followed by rotational conservation grazing of cattle could reduce woody plant cover and density over two to three years in Wisconsin grasslands. This was tested on two levels of woody encroachment.

Materials and Methods

Site descriptions

Research was conducted on three Wisconsin Department of Natural Resources (WI-DNR) wildlife areas that are challenged by woody encroachment. These were Buena Vista Wildlife Area (BV) in central WI (44°21'47" N, 89°35'05" W), Hook Lake Wildlife Area (HL) in southern WI (42°56'23" N, 89°19'11" W) and the Johnson East Tract of the Western Prairie Habitat Restoration Area (WP) in northwestern WI (45°12'31.8" N, 92°25'14.4" W). Soil types were mucky to mucky loamy sand at BV, fine-silty to fine-loamy at HL, and loam to silt loam at WP. All sites had a history of agricultural use but were managed as conservation grasslands for at least 15 years prior to the inception of the study. The climate of Wisconsin is temperate, with cold winters and hot summers. Across our sites, the 30-year average of annual precipitation ranges from 802 to 900 mm and average of annual temperature ranges from 6.4 to 8.2°C.

The herbaceous component of the plant community was dominated by cool-season grasses (cover > 85%) across all sites, with predominantly Kentucky bluegrass (*Poa pratensis* L.) at WP and HL and smooth brome (*Bromus inermis* Leyss.) at BV. The most prevalent forb across all sites was Canada goldenrod (*Solidago canadensis* L.) with cover > 20%. Other forb species were not common at BV, but at HL, additional common forbs (cover > 15%) were stiff goldenrod (*Solidago rigida* L.), orange hawkweed (*Hieracium aurantiacum* L.) and common yarrow (*Achillea millefolium* L.). At WP, Virginia creeper (*Parthenocissus quinquefolia* (L.) Planch.), though a vine rather than a forb, was notable due to its high cover (> 25%). Common encroaching woody species varied by site: at BV, quaking aspen (*Populus tremuloides* Michx.) and white meadowsweet (*Spiraea alba* Du Roi) with 2 to 7% cover by species; at HL, hybrid honeysuckle (*Lonicera x bella* Zabel) and gray dogwood (*Cornus foemina* Mill.) with 2 to 17% cover; at WP, prickly ash (*Zanthoxylum americanum* Mill.) and gray dogwood with 6 to 11% cover. See supplementary table (Table S1) for complete woody plant species list. The Wisconsin State Herbarium's Online Virtual Flora was used for plant taxonomy (Online Virtual Flora of Wisconsin, 2020).

Experimental Design

At each site, 20 x 20 m plots were established in two levels of initial woody plant cover: low cover (5 to 20%) and high cover (20 to 50%). Four treatments were assigned randomly within each woody cover class with three replications, resulting in 12 plots each for low and high initial woody cover (completely randomized design). The four treatments were control (C), graze-only (G), mow and graze (M+G) and herbicide and graze (H+G). The study was initiated in 2016 at all sites, but due to logistical issues, cattle were not applied at WP until 2017. As a result, select treatments were reapplied at WP in 2017.

The control treatment (C) was fenced and no grazing occurred throughout the experiment. The graze-only treatment (G) was only rotationally grazed. The mow and graze treatment (M+G) was mowed to a height of 10 cm once in the winter prior to the beginning of the study using a Bush Hog® (Selma, AL) pulled behind a tractor and rotationally grazed thereafter. The mower cut stems up to 5 cm in diameter and any larger stems were removed near the soil surface with a chainsaw. Due to the delay in implementing grazing at WP, M+G was mowed again (March 2017) prior to initiation of grazing. In the herbicide and graze treatment (H+G), woody plants were treated with herbicide treatment identified as effective once at the beginning of the study (June 2016) and rotationally grazed thereafter. A foliar spray was applied to individual plants as a spray to wet application using a backpack sprayer that delivered a 0.5% solution of 45% triclopyr ester and 16% fluroxypyr ester (Anonymous, 2016a)+ 7 $g \cdot L^{-1}$ of metsulfuron + aminopyralid (Anonymous, 2014). A nonionic surfactant at 0.25% v:v was also included. Volume applied was estimated to be 400 $L \cdot ha^{-1}$ at BV, 600 $L \cdot ha^{-1}$ at HL, and 275 $L \cdot ha^{-1}$ at WP. For aspens at BV, a basal bark treatment utilizing a 30% solution of 60.45% triclopyr (Anonymous, 2016b) mixed with basal bark oil (Anonymous, 2016b) was used instead of the foliar spray. For all other species, herbicide applications targeted woody plant leaves, so interception of spray solution by other plant leaves and stems was limited unless it was beneath the woody plant. At WP, herbicide was not reapplied to H+G in 2017 due to high efficacy of the 2016 treatment. Hybrid honeysuckle was not effectively suppressed by 2016 treatments at HL (< 50% control), necessitating retreatment the following year (June 2017) with a 1.5% aqueous mixture of 34.4% 2,4-D and 16.5% triclopyr (Anonymous, 2010).

Rotational grazing was conducted at each site by private farmers/ranchers who were contracted by WI-DNR, thus grazing methods varied at each site. At BV, 90 animal units of Red angus cattle (cow-calf pairs) were applied to 1 paddock containing treatment plots for grazing periods ranging from 1 to 3 days at a stocking density of 6,000 kg · ha^{-1}. Plots were grazed twice in 2016 (August and September), four times in 2017 (May, July, August and September) and five times in 2018 (each month June through October). At HL, 7 animal units of Scottish Highland cattle (cow-calf pairs in 2016 and steers in 2017) were applied to 3 paddocks containing treatment plots for grazing periods ranging from 1 to 3 weeks at a stocking density of 2,000 kg · ha^{-1}. Plots were grazed once in 2016 (July to August) and once in 2017 (July to September). Due to the needs of the grazier at HL, plots were not grazed in 2018, so data was not collected in spring 2019. At WP, 40 animal units of Holstein cattle (dry heifers) were applied to 1 paddock containing treatment plots for grazing periods ranging from 1 to 2 weeks at a stocking density of 4,000 kg · ha^{-1}. Plots were grazed twice in 2017 (June and August), and twice in 2018 (July and September). In all grazing events, the residual height was 10 cm or higher when cattle were removed from the paddock.

Data collection

Plant community composition was assessed in each plot in June (prior to the first grazing event of the season) using point-intercept transects where points were taken along a line (50 points per plot). Every living plant species touching the point was recorded (Heady *et al.*, 1959). Additionally, in May stem density was measured by counting individual living stems (defined by presence of green leaves) of the woody species at each site. Stems were classified as small (diameter < 2 cm) or large (diameter ≥ 2 cm). Diameter was measured where the stem base met the soil, or just below the soil surface for multi-stemmed species such as honeysuckle.

At BV and HL, plant community composition and stem density data were first collected in spring 2017 after mow and herbicide treatments and one full grazing season had been applied. The same data was collected again in spring 2018 at both sites and at BV in 2019. Due to the delay in implementation at WP, vegetation data was first collected in spring 2018 and again in 2019. As a result, there are 3 site-years of data for BV (2017 to 2019) and 2 site-years for HL (2017 and 2018) and WP (2018 and 2019).

Statistical analyses

Sites and initial woody cover classes were analyzed separately. Total woody plant cover and stem densities of individual species were analyzed, though due to highly variability in stem densities, only the most common woody species could be evaluated. Repeated measures analysis of variance was performed using linear mixed-effects models with an autoregressive (AR1) structure. Plot was a random effect and treatment, year, and treatment by year interaction were included as fixed effects. If the treatment by year interaction was significant, treatments were compared within each year. If treatment was significant without a significant treatment by year interaction, main effects of treatment were evaluated. Significant effects were determined as having a P-value < 0.05 and significantly different means were separated using Fisher's LSD. Data were square root or natural log transformed to meet assumptions of normality and equal variance of the errors whenever necessary. Analyses were performed using R software (version 3.6.2).

Results

High initial woody cover

Treatment main effects are reported across years as the total cover of woody species at each site did not interact with year ($P > 0.19$, Fig. 1). At BV, M+G, H+G and G reduced total woody cover to ≤ 10% ($P < 0.01$) compared to untreated areas (27%). However, at HL only M+G and H+G reduced total woody cover to 12 to 17% ($P = 0.03$) compared to untreated areas (29%) and at WP, only H+G reduced total woody cover to 7% ($P = 0.01$, see Fig. 1) compared to 33% in untreated areas.

Stem density was assessed separately for common species for two stem diameters categories at each location. At BV, stem density of large and small aspens and small *Spiraea* had a treatment by year interaction ($P \leq 0.01$). For both large and small aspens, H+G and M+G reduced stem density 10-fold by year 3 with G treatments effective only on large aspen ($P < 0.04$, 2019, Fig. 2). For *Spiraea*, H+G reduced stem density to near zero in year 1 ($P < 0.01$, 2017), but populations reestablished in subsequent years to similar levels as other treatments (see Fig. 2). At HL, differences in stem density of large and small honeysuckle and small gray dogwood stems were not observed among treatments, nor treatment by year interactions for any species or size (Fig. 3). At WP, differences in stem density of small gray dogwood were also not observed among treatments or treatment by year interactions (Fig. 4). However, for small prickly ash stems H+G reduced stem density four-fold compared to untreated areas ($P < 0.01$, see Fig. 4).

Low initial woody cover

Main effects are reported as the total cover of woody species for each site did not interact with year ($P > 0.6$). At BV and HL, treatments did not differ (BV, $P = 0.91$; HL, $P = 0.12$) but at WP, H+G and M+G reduced total woody cover to 2 to 3% while G increased total woody cover to 23% compared to 11% in untreated areas ($P < 0.01$, Fig. 5).

At BV, small and large aspen stems were too sparse to analyze, but *Spiraea* stem density was similar among treatments ($P = 0.71$, data not shown) with no treatment by year interaction. HL similarly had no differences among treatments for gray dogwood ($P = 0.14$, Fig. 6) and no treatment by year interaction, though areas treated with H+G did trend toward lower stem density. Large honeysuckles did differ with H+G reducing density three-fold and M+G reducing density more than two-fold by year 2 ($P < 0.01$, 2018, see Fig. 6). This response was not observed, however, with the small honeysuckle stem density ($P = 0.46$) nor did treatment interact with year. At WP, dogwood stem density data did not meet assumptions of ANOVA, therefore were not analyzed (Fig. 7). There was no treatment by year interaction and no differences by treatment ($P = 0.07$) for prickly ash, although there was a strong trend toward lower stem density in H+G and M+G treatments (see Fig. 7)

Climate

Precipitation was normal or higher than the 30-year average during all years studied with annual precipitation 10 to 30%, 0 to 20%, 15 to 60% and 50% higher than the 30-year average in 2016, 2017, 2018 and 2019, respectively. Average annual temperatures for each site were similar to the 30-year averages and were within 7% above and 5% below the 30-year average.

Discussion

Successful and sustained reductions in woody cover to preserve, improve, or even expand existing conservation grasslands are necessary to maximize grassland habitat on the landscape. This is important for grassland obligate species like grassland birds, which rely on grasslands and require additional habitat to prevent further population losses and mitigate negative impacts of climate change (With *et al.*, 2008; Zuckerberg *et al.*, 2018). This study demonstrates that rotational grazing in combination with mowing or herbicides can reduce woody cover over 2 to 3 years in conservation grasslands. The response was evident in areas with high initial levels of woody cover (20 to 50%) as woody cover was reduced by H+G across all sites while M+G reduced woody cover at two sites, and G was effective at one site. These results highlight that cattle grazing alone at low stocking density is rarely effective at reducing woody cover, which is similar to the results of others in Wisconsin (Harrington and Kathol, 2009). In low initial woody cover (5 to 20%), however, reductions were only detected at one of the three site (by H+G and M+G). Notably, G resulted in an increase in woody cover in low cover areas at this site, which aligns with studies that link grazing to increases in woody plants (Van Auken, 2000; Briggs *et al.*, 2002). In low initial cover, H+G consistently resulted in the lowest woody cover at each site, though it was not significantly different from C as it was in high cover plots. These results do not support the notion of a woody cover threshold beyond which reductions become increasingly difficult. We expected treatments in low cover to be more effective; however, reductions in cover or stem density were only observed in one instance. It is possible that our low cover areas were generally unresponsive to treatments because they were already above some woody cover threshold, although this would not explain the reductions in high cover. The relatively short timeframe of this study, low and highly variable cover and limited sample size may have influenced our ability to detect reductions in low cover areas as trends in our data suggest management was impactful and biologically significant. The woody cover threshold concept warrants further investigation across a range of climactic and edaphic conditions.

Variable responses observed in our research also highlight the challenges associated with studying woody encroachment. This is a landscape-level phenomenon influenced by biotic and abiotic factors that are heterogeneous across the landscape (e.g. soil type, hydrology, distance to seed sources). We sought to minimize this spatial heterogeneity by focusing treatments in small plots (0.04 ha) within relatively small (< 20 ha) rotationally grazed paddocks. Small plots, however, resulted in high variability of presence and density of less common woody species among plots, and prevented analysis of their response to management. Briggs et al. (2002) encountered a similar issue having a data set that included 15 tree species, but only four with sufficient abundance for detailed analysis. Additionally, in evaluating the effectiveness of our treatments, we cannot rule out the possibility that reductions in H+G and M+G resulted from the effects of herbicides and mowing alone, as we did not include mow and herbicide only treatments. While mowing is widely known to cause resprouting of woody shrubs and when used alone is not effective (Pergams and Norton, 2006; Miller *et al.*, 2015), others have confirmed herbicides are effective for 3 to 8.6 years after treatment (Medlin *et al.*, 2019). However, the response is species, herbicide and context specific as others found herbicide treatments were only effective on shrubs if applied annually over six growing seasons (Farmer *et al.*, 2016). These results were evident in our research as aspen and prickly ash were effectively suppressed by herbicide applications with no resprouting observed, while gray dogwood, honeysuckle and *Spiraea* regrew rapidly. While we postulated that resprouting foliage would be eaten by cattle and provide additional control, this likely did not occur as we rarely observed more than 25% defoliation. Since H+G was the most effective strategy across sites, additional efforts should test herbicide application alone and herbicide application followed by rotational grazing to determine whether grazing provides additional suppression.

While our goal was to test grazing at low stocking density (2,000 to 6,000 kg $\cdot$ ha^{-1}), we believe high cover areas at BV often had higher stocking densities during the hotter parts of the season due to shade-seeking behavior of cattle. Unlike HL and WP, the paddocks containing treatment plots at BV had very little woody cover and therefore very little shade. Fresh dung pat counts performed after each graze event support this conclusion as high initial cover areas consistently had higher counts (4.9 (SE 2.3) pats $\cdot$ 100 m^{-2}) than low cover areas (2.4 (SE 0.8) pats $\cdot$ 100 m^{-2}) at BV. These high cover areas at BV were the only instance of cover and density reductions due to grazing alone, suggesting that high stocking density is key to achieving woody reductions when grazing is the sole management method. Others have suggested the same (Utsumi *et al.*, 2010; Peterson *et al.*, 2013; Petersen *et al.*, 2014), which indicates that targeted grazing at a high stocking density is more effective at suppressing woody plants. This method, however, can run counter to other conservation goals. Land managers should weigh the risks and rewards of low stocking density conservation grazing versus high stocking density targeted grazing when choosing how to best meet conservation goals. In BV high cover areas, H+G and M+G were equally effective as G, indicating they were also reliable methods for reductions at this site. Results in HL and WP high cover areas suggest H+G and M+G are more reliable than grazing alone, which may be partially attributed to more homogenous shade distribution and therefore more homogenous (i.e. lower) stocking density at these sites.

As previously noted, fire was not included in our study because collaborating land managers could not burn these sites and our treatment combinations were of interest as an *alternative* to management with fire. In contexts where prescribed fire is feasible, however, incorporating grazing as an additional management method should be approached with caution. Grazing, by its nature, reduces fine fuels (i.e., forages) and reduces the potential frequency and intensity of fire (Anderies *et al.*, 2002). Since grasslands in the Upper Midwest historically coexisted with fire and it remains an important tool in conservation grassland management, land managers should be cautious when integrating grazing as an additional management method to fire.

Our results also highlight the need to understand and monitor species-specific responses to management methods. Species-specific responses to rotational grazing are common (Fitzgerald *et al.*, 1986; Bailey *et al.*, 1990; Harrington and Kathol, 2009), as are species-specific responses to other woody management techniques, such as fire (Swan, 1970; Briggs *et al.*, 2002). We saw this in several instances, such as in WP and HL high cover areas, where overall woody cover, prickly ash stems and honeysuckle stems were reduced in some treatments, while gray dogwood stem densities did not change. In BV high cover areas, woody cover and aspen stem densities were reduced by the third year, but *Spiraea* stem densities were not. Further exploration of unique responses by select species is presented below.

Gray dogwood

Gray dogwood was present at HL and WP and is a shrub that tends to grow in large clonal patches. Gray dogwood is known for being aggressive and resistant to removal treatments, especially in open habitats (Bowles *et al.*, 1996) and in our study, did not respond to treatments. A similar species, *Cornus drummondii*, also grows aggressively and its continued expansion has been attributed to clonal reproduction, which allows it to avoid competition with grasses and access deep soil water sources (Ratajczak *et al.*, 2011). This species is also resilient to removal of aboveground biomass as stem density increased 600% after application of fire which initially removed all stems (Heisler et al. 2004). Some suggest that the most effective management method for *Cornus* spp. is cutting followed by herbicide application and fire, and timing of management may influence resprouting ability (Converse and Eckardt, 1987). Physical damage by cattle and evidence of browse on gray dogwood was rarely observed. This study does not provide evidence that the conservation grazing alone or combined with other methods were useful in suppression of gray dogwood.

Honeysuckle

Honeysuckle was only common—and therefore only analyzed—at HL. It is an exotic, invasive, multi-stemmed shrub often spread by bird-assisted seed dispersal. *Lonicera* spp. are known to resprout vigorously following disturbance and are highly resilient to removal techniques in open habitats (Luken and Mattimiro, 1991). In our study, large honeysuckle presented varying responses to management. In high initial cover, it was not reduced by treatments, even with an additional herbicide application (H+G) in the second year. In low initial cover, however, large stems showed a reduction by year 2 in M+G and H+G. Similar results were obtained by Love and Anderson (2009) studying *Lonicera marrowii* A. Gray. They found that spring cutting (similar to our mowing treatment) and foliar herbicide (similar to our herbicide treatment) reduced shrub density after one year (any shrub size), although levels of reduction were variable and relatively low (26-68%). They suggested that higher initial densities of honeysuckle lowered the success, which is supported by the lack of response to treatments in our high initial cover (8-36% reduction after two years). Small honeysuckle density, on the other hand, was not reduced by any treatment. This may have been due to new recruitment facilitated by reductions of large honeysuckle cover and resulting increases in resource availability for seedlings. In forests, *Lonicera maackii* (Rupr.) Maxim. seedlings were more likely to establish in removal plots, possibly due to increased light availability (Luken *et al.*, 1997). We observed rings of seedlings present along the edge of large shrubs, likely as a result of seed rain to the soil surface directly from the large shrub (M. Renz, personal observation). Seed rain is a known issue, as Luken and Mattimiro (1991) found high seed densities (1096 ± 47 seeds $\cdot$ m^2) in the soil beneath *L. maackii* in open habitats.

Honeysuckle population dynamics at this site were also likely influenced by the forest directly adjacent to the paddocks. Exotic invasive *Lonicera* spp. are widespread in forests in the Upper Midwest and Wisconsin (Fan *et al.*, 2018) and were the dominant understory plant in the adjacent forest with ongoing dispersal events into the grazed areas (Deering and Vankat, 1999). This highlights that effective management of woody vegetation in a conservation grassland should consider the surrounding landscape. In areas where woody plant dispersal—particularly of exotic invasive species—from adjacent habitats is very likely, frequency and intensity of management needed may remain high over time. Whether conservation grazing is a useful tool in these cases may depend on the stage of invasion, as treatments were more effective in low cover and ‘early detection and rapid response’ is the most effective strategy for reducing invasive plants (Westbrooks, 2004).

Quaking aspen

Quaking aspen is a tree that grows in clonal patches and was only present at BV. Reductions in small aspen densities in high initial cover in all treatments (G, M+G and H+G) were expected as others have shown aspen (*Populus* sp.) is browsed by ungulates (Myking *et al.*, 2011), leaves were within easy reach of cattle (plants < 4.5 m tall), and studies have associated reduced aspen populations with grazing by cattle (Bailey *et al.*, 1990) and wild herbivores (Hessl and Graumlich, 2002; Myking *et al.*, 2011). In addition to browsing, we observed physical damage from cattle rubbing on trees. This was so extensive that it broke the stems and felled the trees (L. Judge and J. Grace, personal observations), thereby reducing large stem densities. Browsing and physical impacts on aspens were consistently apparent, and likely explain the response to treatments.

Management Implications

This study suggests that rotational grazing of cattle at a low stocking density can reduce woody plant cover in conservation grasslands when used in combination with other woody management techniques, and most consistently when used in combination with herbicides. Individual woody species populations, however, responded differently to treatments, and management decisions should account for this. Higher stocking densities of cattle may improve suppression of woody plants, but could also impact other conservation goals. Careful monitoring of impacts and targeted species will be required during management to ensure goals are achieved if integrating conservation grazing to grasslands.

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Figures

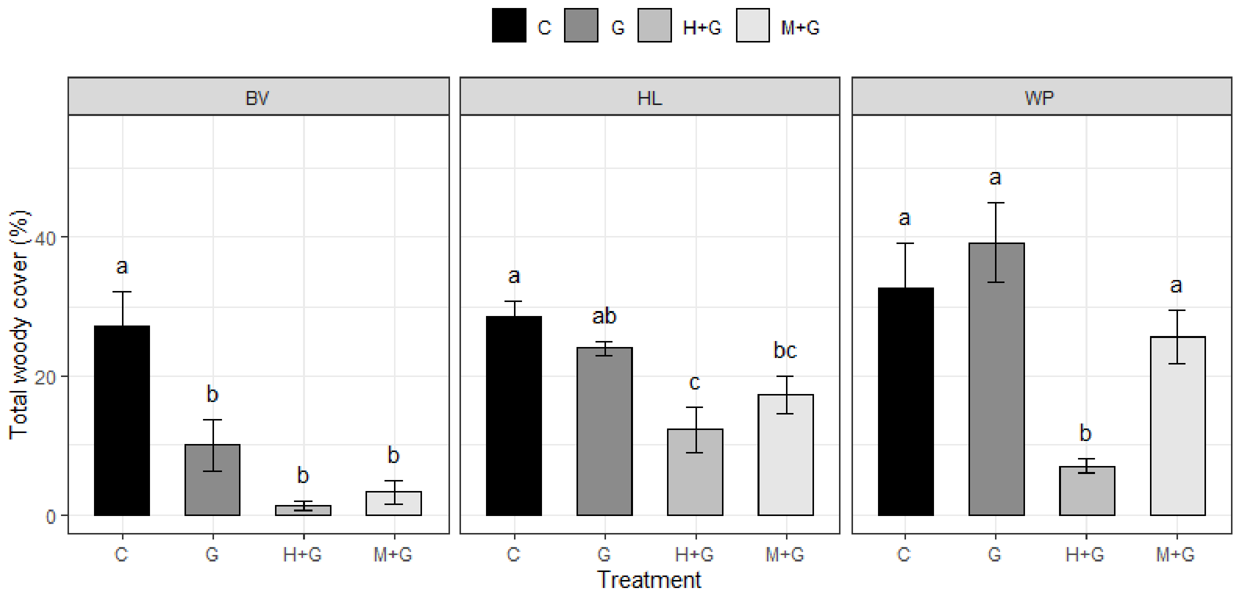

Figure 1. Mean cover of woody species at high initial woody cover (20 to 50%) at Buena Vista Wildlife Area (BV), Hook Lake Wildlife Area (HL) and the Johnson East Tract of Western Prairie Habitat Restoration Area (WP) in Wisconsin measured in June. BV is averaged over 3 years (2017 to 2019) and HL and WP are averaged over 2 years (2017 to 2018 and 2018 to 2019, respectively) because treatment effects did not differ by year. Treatments with different letters differ within that site at $P < 0.05$

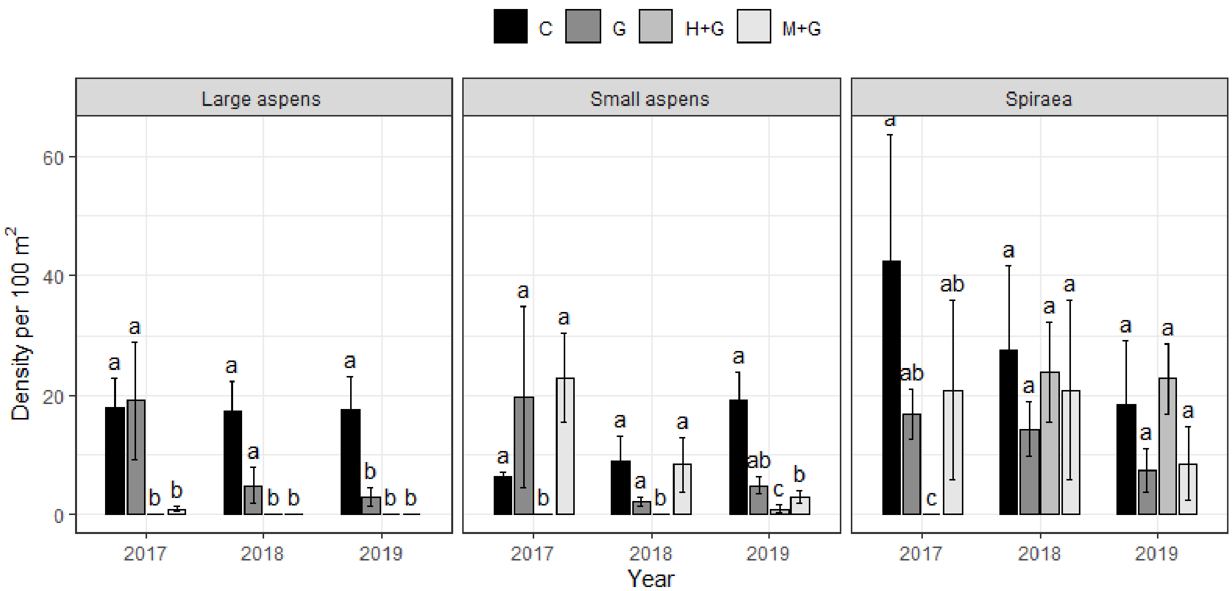

Figure 2. Stem density of common woody species at high initial woody cover (20 to 50%) at Buena Vista Wildlife Area, WI measured in May. Treatments were applied in 2016 with rotational grazing in all plots (excluding control) 2016 to 2019. Treatments with different letters differ within that species and year at $P < 0.05$

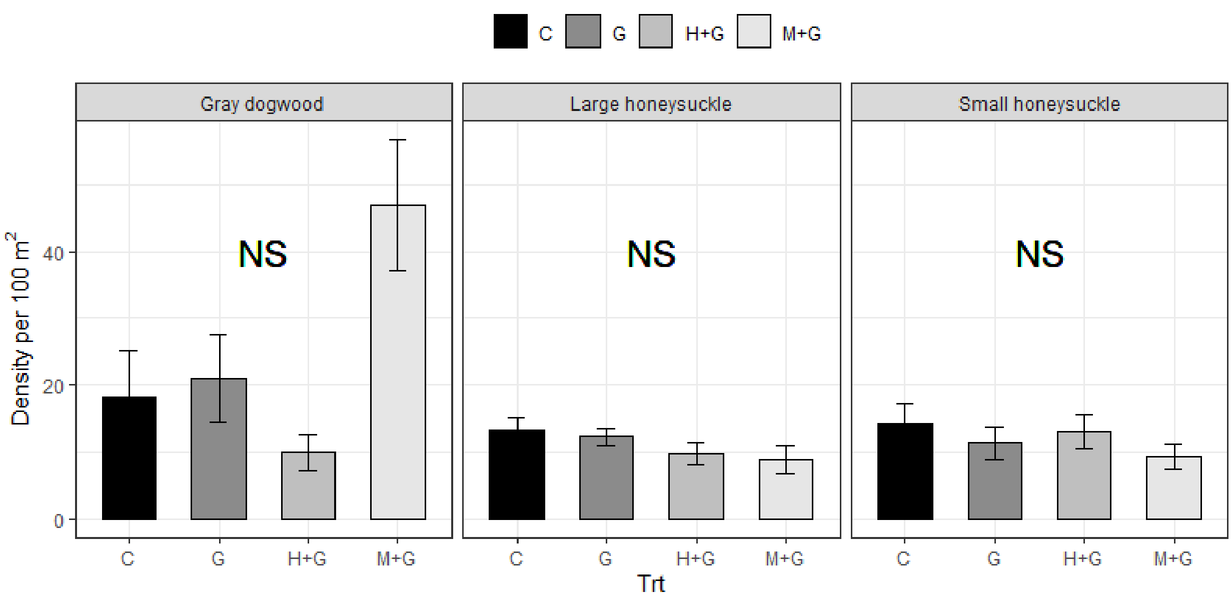

Figure 3. Stem density of common woody species at high initial woody cover (20 to 50%) at Hook Lake Wildlife Area, WI measured in May of 2017 and 2018 (treatment effects did not differ by year). Treatments were applied in 2016 with rotational grazing in all plots (excluding control) 2016 to 2018. An additional herbicide treatment was applied to H+G in spring 2017. NS indicates no difference by treatment

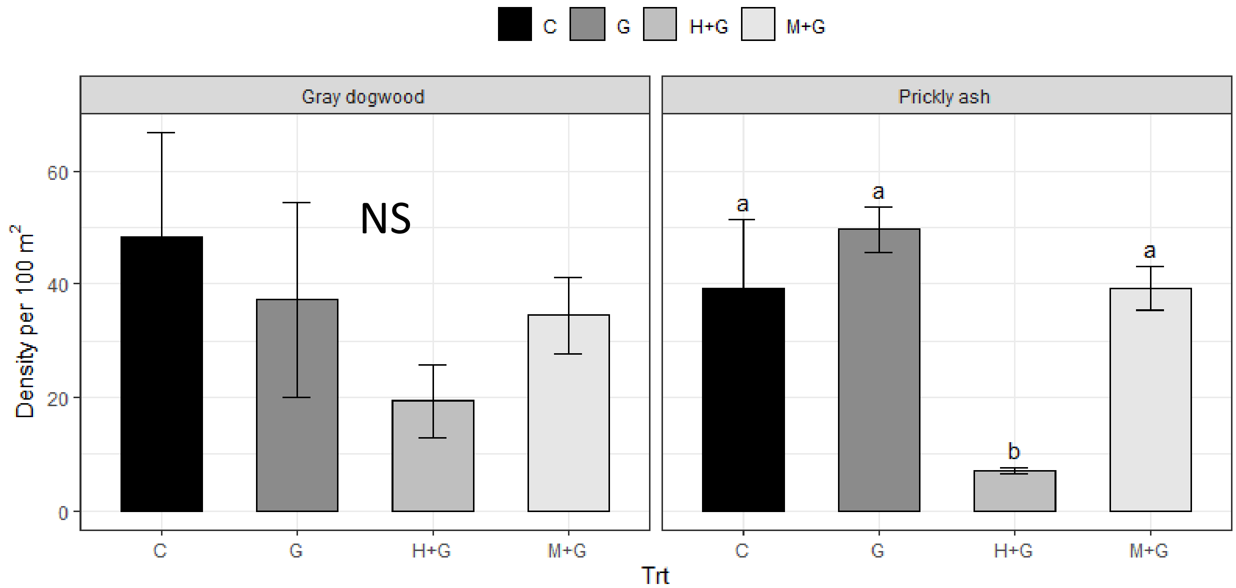

Figure 4. Stem density of common woody species at high initial woody cover (20 to 50%) at the Johnson East Tract of Western Prairie Habitat Restoration Area, WI measured in May of 2018 and 2019 (treatment effects did not differ by year). Rotational grazing occurred in all plots (excluding control) 2017 to 2019. NS indicates no difference by treatment and treatments with different letters differed for that species ($P < 0.05$)

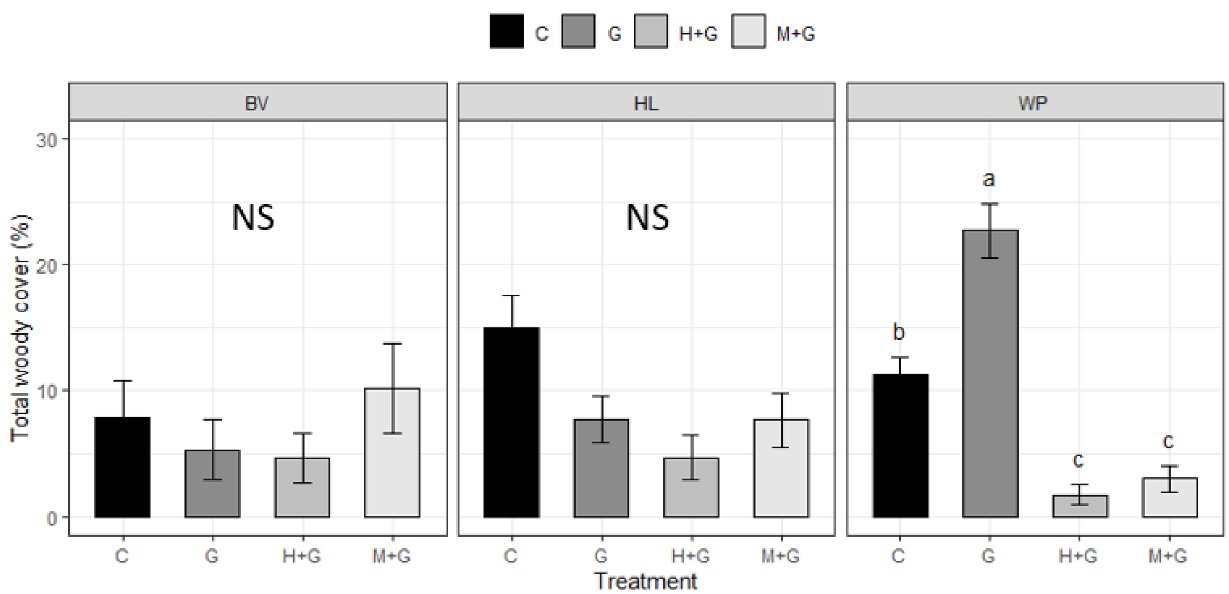

Figure 5. Mean cover of woody species at low initial woody cover (5 to 20%) at Buena Vista Wildlife Area (BV), Hook Lake Wildlife Area (HL) and the Johnson East Tract of Western Prairie Habitat Restoration Area (WP), WI measured in June. BV is averaged over 3 years (2017 to 2019) and HL and WP are averaged over 2 years (2017 to 2018 and 2018 to 2019, respectively) because treatment effects did not differ by year. Treatments with different letters differ within that site at $P < 0.05$ and NS indicates so treatment differences within that site

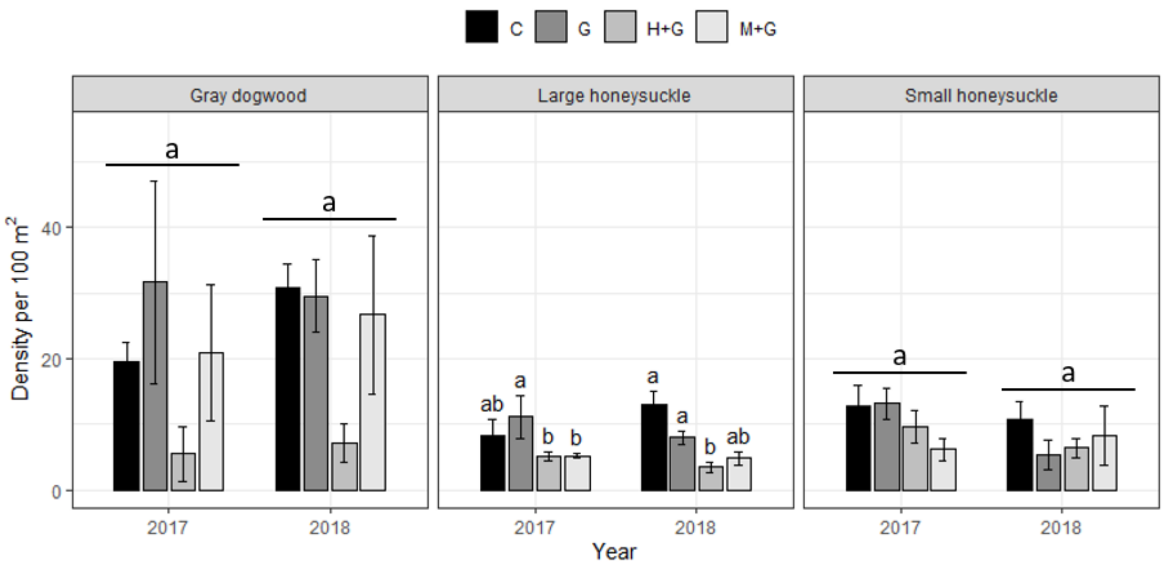

Figure 6. Stem density of common woody species at low initial woody cover (5 to 20%) at Hook Lake Wildlife Area, WI measured in May of 2017 and 2018. Mow (M+G) and herbicide (H+G) treatments were applied in 2016 with rotational grazing in all plots (excluding control) 2016 to 2018. An additional herbicide treatment was applied to H+G in spring 2017. Treatment effects did not differ by year for gray dogwood and small honeysuckle, as indicated by the bar over each year. Treatment effects differed by year for large honeysuckle and different letters indicate a difference within that species and year ($P < 0.05$)

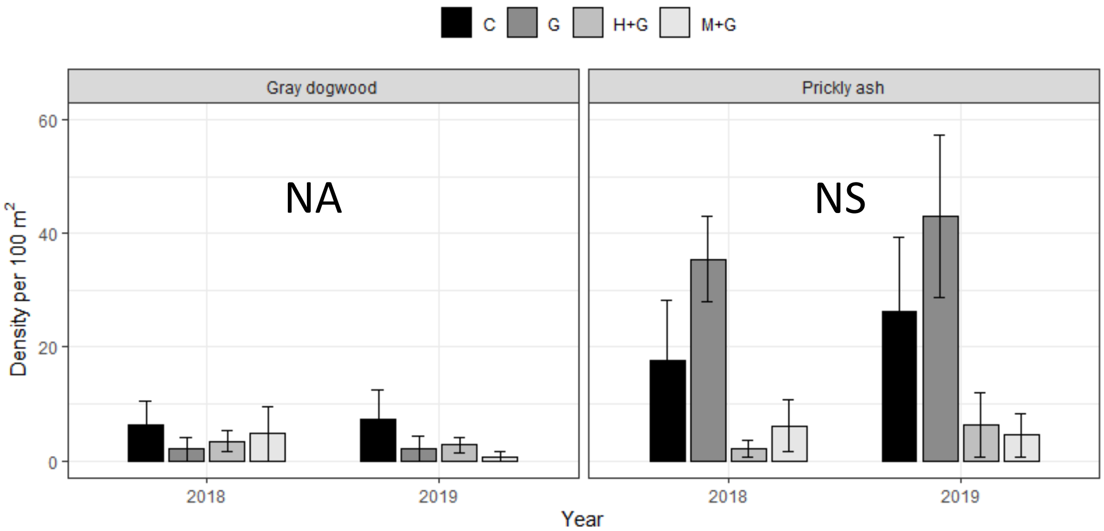

Figure 7. Stem density of common woody species at low initial woody cover (5 to 20%) at the Johnson East Tract of Western Prairie Habitat Restoration Area, WI measured in May of 2018 and 2019. Rotational grazing was applied in all plots (excluding control) 2017 to 2019. NS indicates no difference by treatment within each species and year ($P > 0.05$). NA indicates data did not meet model assumptions and were not analyzed

Supplementary Table

Table S1. Woody plant species at Wisconsin Department of Natural Resources conservation grazing study sites (WA indicates Wildlife Area). Bolded species were most common at that site and evaluated in the stem density analysis

Buena Vista WA	Hook Lake WA	Western Prairie WA
Crataegus sp.	*Amelanchier* sp.	*Acer negundo*
Lonicera x *bella*	*Acer negundo*	***Cornus foemina***
Populus tremuloides	*Acer saccharum*	*Cornus sericea*
Rubus hispidus	*Carya ovata*	*Crataegus* sp.
Rosa carolina	***Cornus foemina***	*Fraxinus pennsylvanica*
Rosa multiflora	*Cornus sericea*	*Lonicera x bella*
Salix alba	*Crataegus* sp.	*Malus* sp.
Salix humilis	*Elaeagnus umbellata*	*Prunus americana*
Spiraea alba	***Lonicera x bella***	*Prunus serotina*
	Malus sp.	*Quercus macrocarpa*
	Pinus sylvestris	*Rhamnus cathartica*
	Prunus serotina	*Rubus allegheniensis*
	Quercus rubra	*Ulmus americana*
	Rhamnus cathartica	*Vibernum lentago*
	Rhus glabra	***Zanthoxylum americanum***
	Rhus typhina	
	Rosa multiflora	
	Rubus allegheniensis	
	Sambucus canadensis	
	Ulmus americana	

Chapter 2: Plant community and grassland bird habitat response to rotational grazing in conservation grasslands in Wisconsin

Abstract

Grasslands are a critically endangered ecosystem and the small amount of grasslands remaining are often dominated by non native plants which provide poor habitat for some species of concern. Grazing, when introduced as a management strategy in conservation grasslands, has potential to affect both the plant community and grassland bird habitat suitability. We assessed plant community and select bird habitat characteristics in 3 temperate, cool-season grass dominated grasslands in Wisconsin that had been rotationally grazed at low to moderate intensity during the 2 or 3 previous seasons. Using a completely randomized design, absolute cover of plant functional groups, diversity metrics, litter cover, litter depth and vegetation height-density were sampled and compared between ungrazed exclusions and grazed areas. Across all sites, non native cool-season grass cover was higher in grazed areas ($P < 0.07$) and at 2 sites *Solidago* spp. cover was lower in grazed areas ($P < 0.06$). Native forb cover (excluding *Solidago* spp.) was higher in grazed areas at only one site ($P = 0.07$) and overall and native plant diversity (Simpson's) and woody plant cover did not differ at any site. While litter depth was lower in grazed areas ($P = 0.02$), litter cover and vegetation height-density did not differ from exclusions. Results indicate that improvements in grassland conservation value with low to moderate intensity rotational grazing were limited and additional management may be required to enhance outcomes for native plants and grassland birds.

Introduction

Grasslands in the United States are considered a critically endangered ecosystem (Noss *et al.*, 1995) and represent critical habitat for grassland obligate wildlife. Remaining conservation grasslands—where conservation goals rather than agricultural production are the primary focus—rarely reflect the plant community of the original grasslands (as they existed prior to European settlement) and continue to change (Alstad *et al.*, 2016). Although these grasslands originated under the influence of periodic drought, frequent fire and mammalian grazers (Anderson, 2006), these natural disturbance processes rarely occur in the same manner in the present day and are often impractical or even impossible to restore (Hobbs and Huenneke, 1992). Consequently, it has been suggested that grassland management and restoration efforts in these highly altered ecosystems focus on achieving conservation goals using a variety of tools and continually refining techniques based on results of management rather than attempting to mimic historic disturbances (Ellis-Felege *et al.*, 2013; Perkins *et al.*, 2019). Conservation goals in these grasslands often include reducing the presence of non native or woody plants, increasing the presence of native plants, improving plant community diversity by reducing dominant species, and providing habitat for grassland bird species and other wildlife (James *et al.*, 2017; Hendrickson *et al.*, 2019).

Grazing with native ungulates or livestock is being used as a management strategy to meet conservation goals. The success of grazing practices, however, is variable and likely dependent on species present and management techniques employed (Chapter 1) and may or may not result in progress toward specific plant community or habitat goals. For instance, there is conflicting data on the benefits of grazing to the native plant community and it is unclear what accounts for these variable outcomes (improvements in Hickman *et al.*, 2004; Gennet *et al.*, 2017; Limb *et al.*, 2018; but see Dix, 1959; Johnson and Cushman, 2007; Seabloom *et al.*, 2015) . This suggests the effects of grazing on grassland plant communities are complex and potentially context-specific.

In addition to improvements to the native plant community, grassland bird species are also a key focus of management (Herkert *et al.*, 1996; Walk and Warner, 2000). Their populations have declined dramatically in recent decades (Rosenberg *et al.*, 2019) and many are Species of Greatest Conservation Need (SGCN) or even threatened in the state of Wisconsin (Wisconsin Department of Natural Resources, 2018). Threatened species include Henslow's sparrow (*Ammodramus henslowi*) and greater prairie-chicken (*Tymanuchus cupido*). Bobolink (*Dolichonyx oryzivorus*) and eastern meadowlark (*Sturnella magna*), and several other species are also SGCN in Wisconsin. Each bird species requires a specific type of habitat structure (encompassing factors like depth of litter layer, amount of bare ground, and the height and density of vegetation) and plant community in terms of the amount of grasses and forbs present (Sample and Mossman, 1997). Consequently, management that affects habitat structure and plant community will affect grassland birds (Bruckerhoff *et al.*, 2020).

Grazing is likely to affect both habitat structure and plant community and has potential to contribute to a landscape-level mosaic of grassland characteristics providing the full range of habitat required by grassland bird species (Sample and Mossman, 1997; Fritcher *et al.*, 2004). If implemented correctly, grazing may benefit grassland bird species by contributing to structural heterogeneity (Derner *et al.*, 2009); however, the need for ungrazed nesting season refuges within grazed areas to prevent the damage to nests caused by grazing animals is critical (Temple *et al.*, 1999; Churchwell *et al.*, 2008; Campomizzi *et al.*, 2019).

Livestock grazing as a conservation grassland management tool is a relatively new practice to the Upper Midwest and this study evaluated its use across Wisconsin. Specifically, we measured the responses of plant community and bird habitat characteristics (litter depth and cover and vegetation height-density) to 2 to 3 seasons of low to moderate-intensity rotational grazing in temperate conservation grasslands dominated by cool-season grasses. We hypothesized that grazing would increase native plant and grassland bird habitat diversity, resulting in improved conservation value of grasslands.

Materials and methods

Site descriptions

Research was conducted on three Wisconsin Department of Natural Resources (WI-DNR) wildlife areas distributed across Wisconsin. These were Buena Vista Wildlife Area (BV, 44°21'47"N, 89°35'05"W), Hook Lake Wildlife Area (HL, 42°56'23"N, 89°19'11"W) and the Johnson East Tract of the Western Prairie Habitat Restoration Area (WP, 45°12'31.8"N, 92°25'14.4"W). Soil types were mucky to mucky loamy sand at BV, fine-silty to fine-loamy at HL, and loam to silt loam at WP. All sites had a history of agricultural use but were managed as conservation grasslands for at least 15 years prior to the inception of the study. The climate of Wisconsin is temperate, with cold winters and hot summers. Across sites, the 30-year average of annual precipitation ranges from 802 to 900 mm and average of annual temperature ranges from 6.4 to 8.2°C. The herbaceous component of the plant community was dominated by non native cool-season grasses (cover >85%) across all sites, with Kentucky bluegrass (*Poa pratensis* L.) the predominant species. The most prevalent forb across all sites was Canada goldenrod

(*Solidago canadensis* L.) with cover > 14%. See Table 1 for the full plant community by site and Table 2 for pre-treatment cover of functional groups of interest.

Experimental design

At each site, 20 x 20 m plots were established and treatments randomly assigned with three replications, totaling 12 plots in a completely randomized design. Plots were selected to be early in the invasion process of woody plants and had woody cover between 5 and 20% prior to initiation of the study. Treatments evaluated included a control treatment (C, n = 3) that was fenced to exclude grazing and 3 treatments that were rotationally grazed. A graze only treatment (G), in addition to graze and one-time mow and graze and one-time foliar herbicide (targeting woody plants), was also applied (described in Chapter 1). These initial mow and initial herbicide treatments were applied at the time of plot establishment and were rotationally-grazed each season thereafter. A nonmetric multi-dimensional scaling (NMDS) analysis of 2019 (2018 for HL) plant communities found neither the initial mow, then grazed plots nor the initial herbicide, then grazed plots differed from the graze-only plots so all plots with grazing management were combined at each site and are represented by G hereafter (n = 9). The only exception was initially mowed, then grazed plots at WP, so these were not included in G (n = 6) for subsequent analysis. The study was initiated in 2016 at all sites, but due to logistical issues, cattle were not applied at WP until 2017.

Rotational grazing was conducted at each site by private farmers/ranchers who were contracted by WI-DNR, thus grazing methods varied at each site. At BV, 90 animal units of Red angus cattle (cow-calf pairs) were applied to one paddock containing treatment plots for grazing periods ranging from 1 to 3 days at a stocking density of 6,000 kg · ha^{-1}. Plots were grazed twice in 2016 (August and September), four times in 2017 (May, July, August and September) and five times in 2018 (each month June through October). At HL, 7 animal units of Scottish Highland cattle (cow-calf pairs in 2016 and steers in 2017) were applied to three paddocks containing treatment plots for grazing periods ranging from 1 to 3 weeks at a stocking density of 2,000 kg · ha^{-1}. Plots were grazed once in 2016 (July to August) and once in 2017 (July to September). Plots were not grazed in 2018 at HL, so plant community data was collected in spring 2018 and not collected in spring 2019. At WP, 40 animal units of Holstein cattle (dry heifers) were applied to one paddock containing treatment plots for grazing periods ranging from 1 to 2 weeks at a stocking density of 4,000 kg · ha^{-1}. Plots were grazed twice in 2017 (June and August), and twice in 2018 (July and September). In all grazing events, the height of residual vegetation was 10 cm or higher when cattle were removed from the paddock.

Data collection

Plant community composition was assessed in each plot in late May or early June (prior to the first grazing event of the season) using point-intercept transects where points were taken along a line (50 points per plot). Every living plant species touching the point was recorded (Heady *et al.*, 1959). Plant community composition data was collected in 2018 at HL and 2019 at BV and WP. Due to the 1 year delay in implementation at WP, 2019 results reflect 2 prior seasons of grazing, whereas at BV results reflect 3 prior seasons of grazing. At HL, data collected in spring 2018 reflect 2 prior seasons of grazing. At BV and WP in 2019, ground cover was characterized as litter (defined as prostrate dead vegetation) or bare ground (encompassing mineral soil, moss, cow pie, or rock) at each point to help characterize grassland bird habitat.

Additional grassland bird habitat measures were collected at BV and WP in May 2019, which included vegetation height-density (a measure of visual obstruction) and litter depth. These measurements were taken in the first and third weeks of May, after vegetation began to grow and prior to the first grazing event of the season. Due to resource constraints, only half the plots were measured at each site, resulting in n = 3 for grazed (G) and ungrazed (C) plots. Within each plot, a Robel pole (Robel *et al.*, 1970) was placed randomly once within each quarter of the plot (stratified design to ensure representation of the entire plot). At each placement, the lowest visible band (decimeter increments) from a distance of 4 m and height of 1 m was recorded in each of the 4 cardinal directions. The distance from the ground to the top of the litter layer (litter depth) was also measured at each of these viewing points, resulting in 16 height-density and litter depth measures within each plot.

Data analyses

Sites were analyzed separately and plot was the experimental unit. Cover values were calculated as the absolute cover of each species or functional group (number of points where encountered per plot divided by number of points measured). As multiple species were counted at each point, cover can exceed 100%. The dominant grass species and forb genus across all sites—*Poa pratensis* and *Solidago* spp., respectively—were analyzed individually and subsequent functional group analyses excluded them. Richness was determined as the total number of species or native species present per plot. Diversity and evenness were also calculated, as richness alone does not adequately reflect exotic plant dominance (Seabloom *et al.*, 2013). Simpson's index was chosen for diversity and evenness because it performs better than other indices with small samples sizes and is best suited to detect changes in dominance

(Magurran, 2004), which was a primary concern on these sites. Simpson's (1 – D) was calculated where D = $\sum(P_i)^2$ and P_i is the proportional cover of each species based on total cover. Evenness was calculated from Simpson's reciprocal index by (1/D)/S where S is species richness. Comparisons of cover, diversity and evenness within grazed areas and grazing exclusions were made using Welch's T-test or the Mann-Whitney U-test when assumptions of normality or homogeneity of variance were not met, as determined by Shapiro-Wilk test for normality and Levene's test.

Repeated measures analysis of variance was performed on vegetation height-density and litter depth data using linear mixed-effects models with an autoregressive (AR1) structure. Plot was a random effect and treatment, sample timing, and treatment by sample timing interaction were included as fixed effects. Square root transformations were performed to meet model assumptions and no significant treatment by sample timing interactions were found. P-values reported are results of the overall F-test. Significant effects were determined as having a P-value < 0.10 due to small sample size and all analyses were performed using R software (version 3.6.2).

Results

Climate

Precipitation was normal or higher than the 30-year average during all years studied with annual precipitation 10 to 30%, 0 to 20%, 15 to 60% and 50% higher than the 30-year average in 2016, 2017, 2018 and 2019, respectively. Average annual temperatures for each site were similar to the 30-year averages and were within 7% above and 5% below the 30-year average.

Plant community

Limited change in plant community composition was observed among common species and other cover classes at each site. *P. pratensis* increased with grazing at BV (Fig.1; $P = 0.06$) while other graminoid species (non native cool-season grass species comprise > 99.5% of this category) increased at HL and WP (Fig. 2; $P < 0.05$). In contrast *Solidago* spp. cover decreased in grazed areas at HL and WP (Fig. 3; $P < 0.05$). Other native forbs (excluding *Solidago* spp.) increased cover in grazed areas at HL ($P = 0.07$; Table 3) but non native forb and woody plant cover did not differ at any site (Table 3). Limited differences in richness, evenness and Simpson's diversity for the total and native plant community were observed as total community evenness was higher in BV grazing exclusions ($P = 0.08$; Table 4) and no other differences were found.

Bird habitat

Litter depth was reduced > 56% by grazing at both BV and WP grazed areas ($P < 0.03$) and did not differ by sample timing (Fig. 4). Litter cover at both sites was > 93 ± 2% across all treatments and did not differ at any site. Vegetation height-density did not differ between grazed areas and exclusions, though did differ by sample timing at WP (Fig. 5; $P < 0.01$).

Discussion

Plant community

Introducing rotational grazing into Wisconsin's conservation grasslands decreased *Solidago* spp. cover at 2 sites (discussed below) and increased cover of the already dominant non native cool-season grasses at all sites. Similarly, Maier (2012) observed increases in cool-season grasses with rotational grazing in Wisconsin and some suggest that grazing facilitates its invasion (Dix, 1959; Murphy and Grant, 2005). Grazing did not affect total or native plant community diversity, richness or evenness, which is likely attributable to the high dominance of the non native cool-season grasses, which makes the community resistant to change (Ellis-Felege *et al.*, 2013). Our results differed from other research that found that low-intensity grazing in temperate grasslands was associated with increased plant richness and diversity, but moderate to heavy grazing decreased diversity (Wang and Tang, 2019). Though our stocking densities were relatively low, longer grazing periods at WP and HL may have prevented increases in plant diversity. The relatively short timeframe of this study (2 to 3 years) could also have played a role in the lack of change in diversity as others have observed changes in diversity in longer-term studies (at least one decade) (Towne *et al.*, 2005; Limb *et al.*, 2018; Lyseng *et al.*, 2018), although this is not always the case (Allred *et al.*, 2012).

Solidago spp. cover was reduced in grazed areas at two sites and trended lower than exclusions at the third. These species were not desired at their current cover as they outcompete other desirable forbs in conservation grasslands (Banta *et al.*, 2008), so reductions are considered beneficial. We observed reductions due to rotational grazing over 2 to 3 seasons, but some have observed increases (Towne *et al.*, 2005) while other observed decreases (Hartnett *et al.*, 1996) due to continuous grazing, suggesting that rotational grazing may result in more reliable reductions. Reductions in our study were likely due to physical impacts (i.e. trampling), biophysical changes due to grazing (e.g. changes in nutrient distribution or cycling), or enhanced competitiveness of other species present as cattle did not graze *Solidago* spp. (L. Judge, personal observation). Future studies could further investigate the response of *Solidago* spp. to rotational grazing.

Reductions in *Solidago* spp. cover were not accompanied by increases in non native forbs. Increases in non native forbs can be a concern on grazing sites because grazing can cause microsites for invasion (Hobbs and Huenneke, 1992; Middleton, 2002), allowing for easy establishment of undesirable species. Conversely, microsites can also allow for establishment of desirable native forbs. Increases in native, non *Solidago* spp. forbs were only observed on 1 site (HL), where mean cover of this functional group was 12% higher in grazed areas (37%) compared to exclusions (25%). Results suggest that cover of native forbs was high enough prior to initiation of grazing to allow for increased cover. On the other 2 sites where increases were not observed, cover in grazed areas and grazing exclusions was ≤ 6%. Lack of differences at these two sites may have resulted from insufficient local native species pools to facilitate increases (Foster *et al.*, 2011). In these cases, grazing should be accompanied with the addition of native seeds as grazing can facilitate native plant emergence (Martin and Wilsey, 2006). Land managers should consider this strategy where increasing native forbs is a priority.

Woody plant cover did not differ between grazed areas and exclusions at any site. Reductions in woody cover generally increase the conservation value of grasslands; however, grazing alone (when not combined with other management techniques) rarely provides woody reductions (Harrington and Kathol, 2009; Lyseng *et al.*, 2018; Chapter 1).

Grassland bird habitat

Measurements of bird habitat characteristics in May 2019 revealed differing responses of each characteristic. Data was collected before grazing began in 2019, so BV reflects 3 prior seasons of rotational grazing and WP reflects 2 prior seasons of grazing. Commonly, researchers measure the in-season effects of grazing by classifying 'ungrazed' areas as those that have been grazed in previous seasons but not yet grazed during the current season (e.g. Gennet *et al.*, 2017; Vold *et al.*, 2019). In our study, ungrazed exclusions were never grazed and our measures of grazed areas were designed to measure the between-season effect of grazing (i.e. the legacy effect of previous seasons' grazing). This allowed us to evaluate whether an ungrazed refuge (ungrazed until August 1st) within the fenced area would provide habitat distinct from the grassland area outside the fence borders. We focused on this aspect because trampling and other livestock-caused disturbance of nests is a known issue (Paine *et al.*, 1996; Nack and Ribic, 2005) and best management for nest success includes a nesting refuge or rest-rotation grazing that leaves an area ungrazed during the nesting season (Temple *et al.*, 1999; Campomizzi *et al.*, 2019). Additionally, many factors ultimately determine habitat suitability, such as the composition of the surrounding landscape and patch size, (Ribic and Sample, 2001; Shahan *et al.*, 2017) and our study focused only on within-field habitat characteristics.

Vegetation height-density differed by sample timing (higher in the third week of May than the first week) at WP, possibly due to abundant precipitation (50% higher than 30 year average) combined with high cover of *P. pratensis* (Government of Alberta, 2017). Vegetation height-density did not differ, however, between grazed areas and grazing exclusions. At both sites litter depth was lower in grazed areas than in exclusions. This was expected and similar to the findings of others (Naetht *et al.*, 1991; Gennet *et al.*, 2017). There was no effect on litter cover as it was relatively high (≥ 94%) in both grazed areas and grazing exclusions. As only 1 of the 3 bird habitat characteristics differed due to rotational grazing in prior seasons, results suggest that grazing only marginally increases grassland bird habitat diversity in a current-year ungrazed refuge and will only differ from areas outside the fenced pasture in having lower litter depth. Although vegetation characteristics are important throughout the season (not just in May), we do not believe the trends we observed would change later in the season. Future studies could test this assumption.

The low litter depths (1.3 to 1.6 cm) provided by grazing within the range of low vegetation height-density (0.8 to 5 cm) and high litter cover (≥ 94%) and modest changes in plant community found in grazed areas on our sites are limited in their significance for grassland bird species, which are all Species of Greatest Conservation Need (SGCN) in Wisconsin (Wisconsin DNR, 2018). Grasshopper sparrows (*Ammodramus savannarum*) may find suitable habitat in grazed areas, as they respond positively to decreasing vegetation height-density ((Madden *et al.*, 2000; Byers *et al.*, 2017). Peak abundances, however, have been found at 3 cm litter depth and 10% bare ground (Vold *et al.*, 2019) and nests have been associated with a mean litter depth of 3.6 cm (Hubbard *et al.*, 2006), meaning litter depth in ungrazed areas (outside fence borders) may be equally or more appropriate for this species. Furthermore, grasshopper sparrow nest density has been found to increase with increasing native forbs (Byers *et al.*, 2017) and grazing only increased native forb cover on 1 of our sites, while increasing non native grass cover on all sites. Vesper sparrows (*Pooecetes gramineus*) prefer litter < 3 cm deep and at < 50% cover (Sample, 1989) and abundances have been found to increase with increasing bare ground (up to 75%) (Vold *et al.*, 2019). Based on the high litter cover in grazed areas on our sites, vesper sparrows are not likely to benefit from rotational grazing (via increased amounts of preferred habitat). Western meadowlark (*Sturnella neglecta*) has been associated with increasing forb cover (around 50% appears optimal) (Madden *et al.*, 2000; Vold *et al.*, 2019) and abundances increase with litter depth up to 5 cm (Madden *et al.*, 2000), meaning the grass-dominated, low litter depth grazed areas are likely not preferred by this species. Bobolink (*Dolichonyx oryzivorus*) is considered a habitat generalist that can occupy a range of litter depth and other habitat characteristics (Sample, 1989), meaning it may benefit from grazing management, although nest success was found to be highest with deep litter and high litter cover (Byers *et al.*, 2017). Finally, eastern meadowlark (*Sturnella magn*a) nest occurrence and nest success were associated with deep litter and high litter cover (Sample, 1989; Byers *et al.*, 2017). Litter depth of 8.3 cm was associated with nest sites in Kansas (Hubbard *et al.*, 2006), therefore grazed areas likely do not benefit reproductive success of this species.

Additional species listed as 'threatened' in Wisconsin (Wisconsin DNR, 2018) exhibit a range of probable responses to rotational grazing. Henslow's sparrow (*Ammodramus henslowii*) is considered a tallgrass species (Sample and Mossman, 1997) and has higher abundances and higher nest success on sites with deeper litter (Swengel and Swengel, 2001; Byers *et al.*, 2017) indicating it will likely not benefit from grazing. The upland sandpiper (*Bartramia longicauda*) also requires tall, dense vegetation for successful nesting, but needs areas that have been recently grazed, hayed or burned for brood-rearing habitat (Derner *et al.*, 2009). Grazing may benefit this species during brood-rearing, although it is generally found in higher abundance in areas with high amounts of bare ground (Fuhlendorf *et al.*, 2006), which our grazed areas did not offer. Like the upland sandpiper, the greater prairie-chicken (*Tympanuchus cupido*) requires a full range of vegetation characteristics for different life stages (i.e. nesting and brood-rearing) which can be provided by grazing and burning (Hardy, 2018). Benefits to prairie chickens are limited, however, because of their need for large areas of grassland (> 100 ha) and low populations (e.g. the Buena Vista Wildlife Area is home to the only remaining population of the greater prairie-chicken in Wisconsin).

Grazing management at the low to moderate intensity applied on our study sites in Wisconsin's temperate cool-season grass dominated grasslands appears to potentially provide a marginal benefit to some grassland birds species of concern. To improve outcomes for a greater number of species, it may be beneficial to vary grazing management by factors like rotation length, residual height and stocking rate. This may provide more heterogenous habitat than the habitat found on our grazing sites, although Sliwinski et al. (2020) found that different grazing management systems did not result in greater habitat diversity or avian species richness. They suggest that management with prescribed fire and extreme stocking densities (both very low and very high) will be needed to benefit the greatest number of SGCN bird species. Patch burn grazing may provide a viable option, as it has been shown to increase habitat heterogeneity (Churchwell *et al.*, 2008; Coppedge *et al.*, 2008; Fuhlendorf *et al.*, 2008; Hovick *et al.*, 2014) although some have found management with fire to be detrimental to certain species (Swengel and Swengel, 2001).

Management Implications

Promoting both native plant and avian community populations and diversity within those communities using low to moderate intensity rotational grazing over 2 to 3 seasons is challenging. Increases in non native cool-season grasses were observed and results suggest additional efforts are needed on sites dominated by non native species to achieve increases in native forb cover. Changes in grassland bird habitat were observed, although characteristics of nesting refuges (ungrazed until August 1st) within grazed areas are likely to differ only modestly from areas that are ungrazed and potential benefits are limited to a small number of species. Land managers may need to choose between managing for plant biodiversity and bird habitat if using grazing as the sole management method.

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Figures

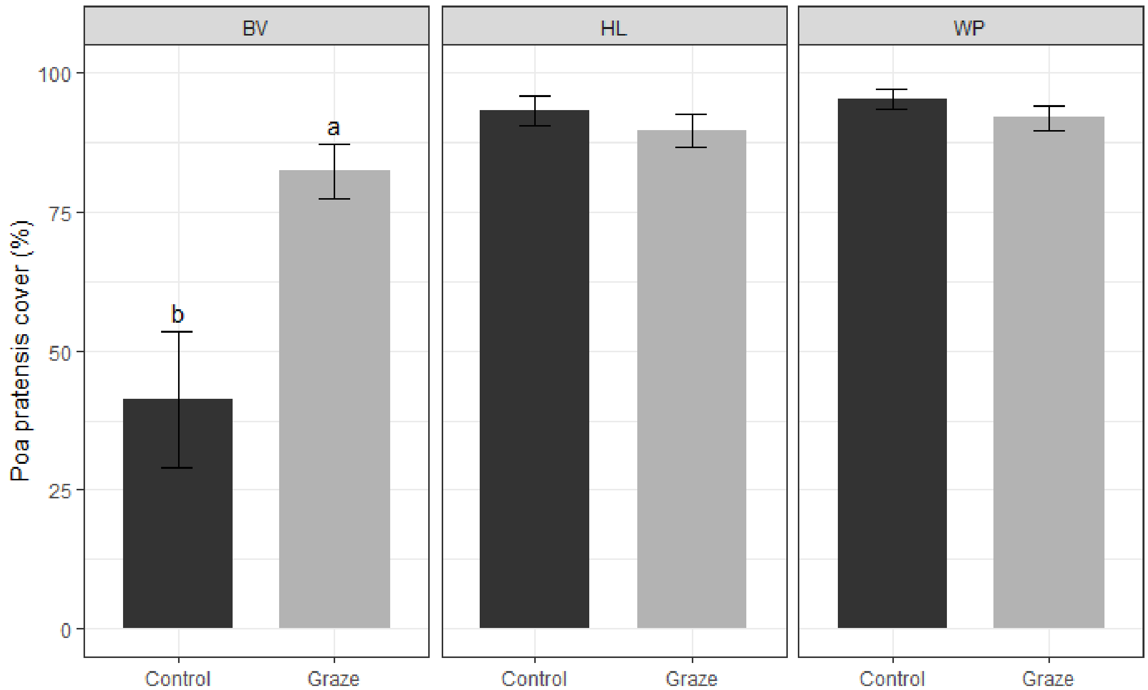

Figure 1. Absolute cover of *Poa pratensis* at Buena Vista Wildlife Area (BV), Hook Lake Wildlife Area (HL) and the Johnson East Tract of Western Prairie Habitat Restoration Area (WP) in Wisconsin measured in late May/early June, 2019 (BV and WP) and late May, 2018 (HL). Grazed areas were rotationally grazed during the 3 prior growing seasons at BV and 2 prior growing seasons at WP and HL. Treatments with different letters differ within that site at P < 0.10

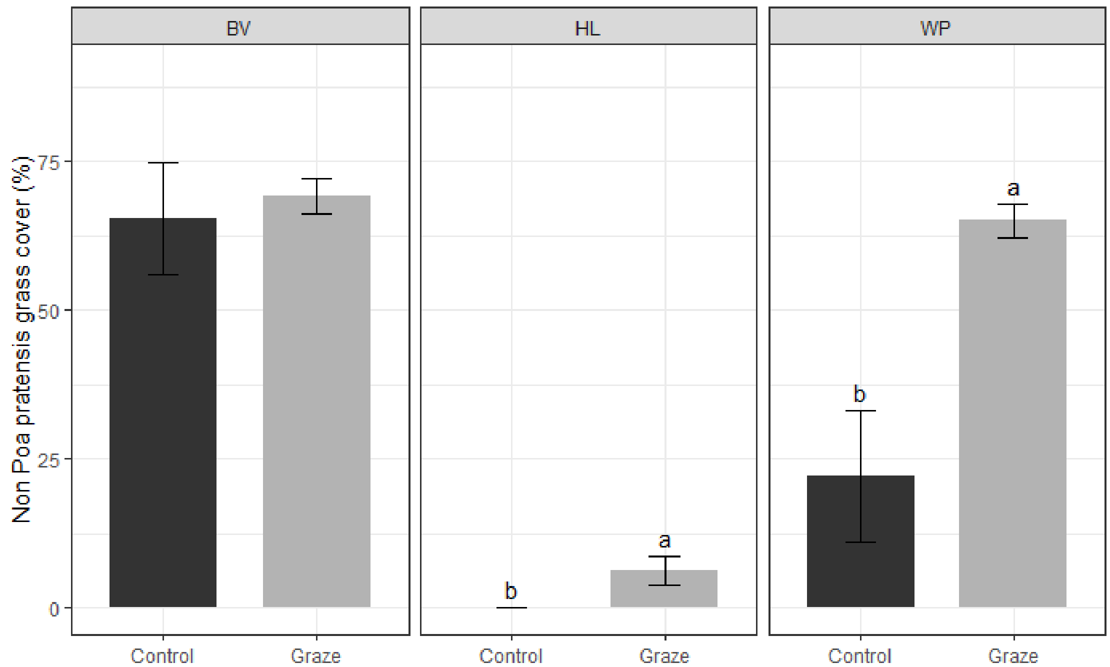

Figure 2. Absolute cover of non native cool-season grasses exclusive of *Poa pratensis* at Buena Vista Wildlife Area (BV), Hook Lake Wildlife Area (HL) and the Johnson East Tract of Western Prairie Habitat Restoration Area (WP) in Wisconsin measured in late May/early June, 2019 (BV and WP) and late May, 2018 (HL). Grazed areas were rotationally grazed during the 3 prior growing seasons at BV and 2 prior growing seasons at WP and HL. Treatments with different letters differ within that site at P < 0.10

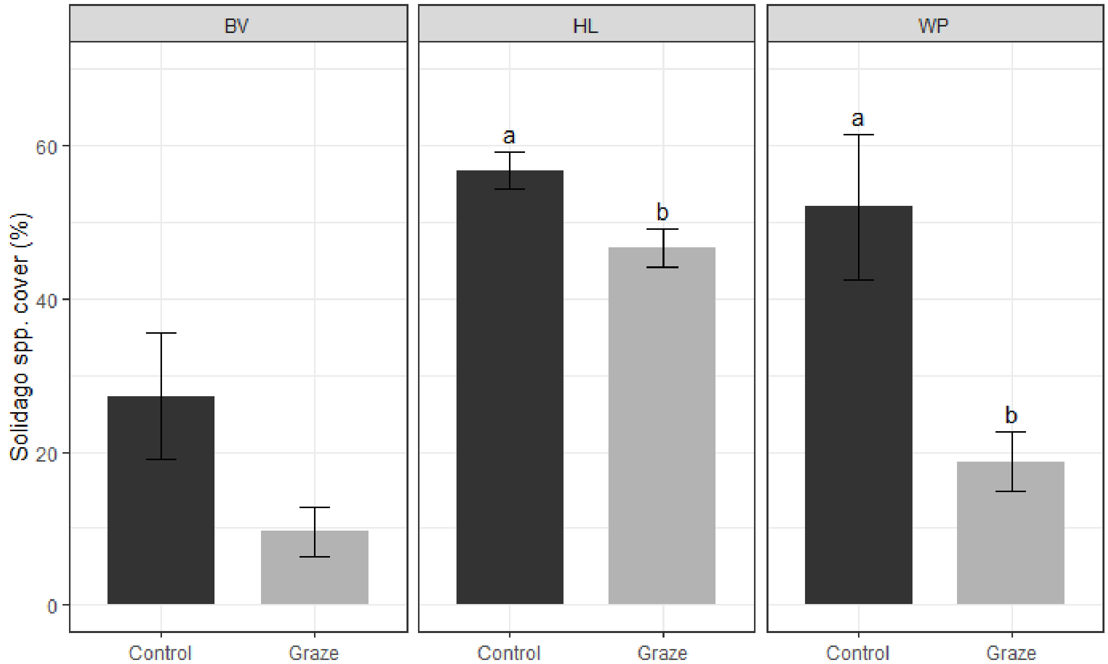

Figure 3. Absolute cover of *Solidago* spp. at Buena Vista Wildlife Area (BV), Hook Lake Wildlife Area (HL) and the Johnson East Tract of Western Prairie Habitat Restoration Area (WP) in Wisconsin measured in late May/early June, 2019 (BV and WP) and late May, 2018 (HL). Grazed areas were rotationally grazed during the 3 prior growing seasons at BV and 2 prior growing seasons at WP and HL. Treatments with different letters differ within that site at $P < 0.10$

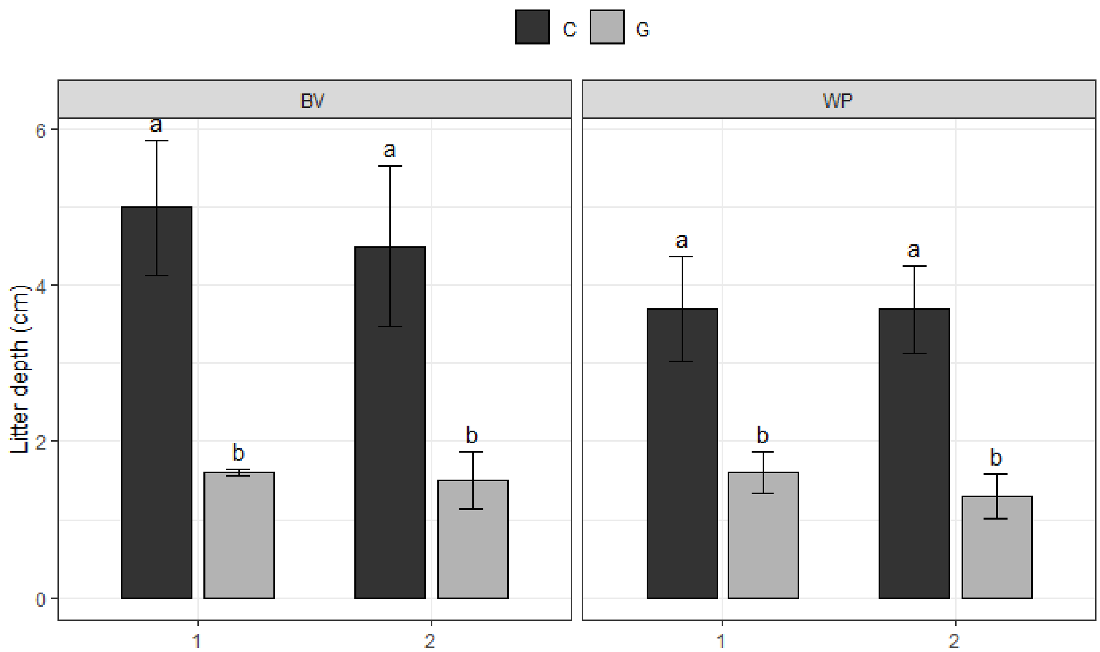

Figure 4. Litter depth at Buena Vista Wildlife Area (BV) and the Johnson East Tract of Western Prairie Habitat Restoration Area (WP) in Wisconsin measured in the first week of May (indicated by 1) and third week of May (indicated by 2) 2019. Grazed areas (G) were rotationally grazed during the 3 prior growing seasons at BV and 2 prior growing seasons at WP and control (C) excluded grazing. Treatments with different letters differ within that site at $P < 0.10$

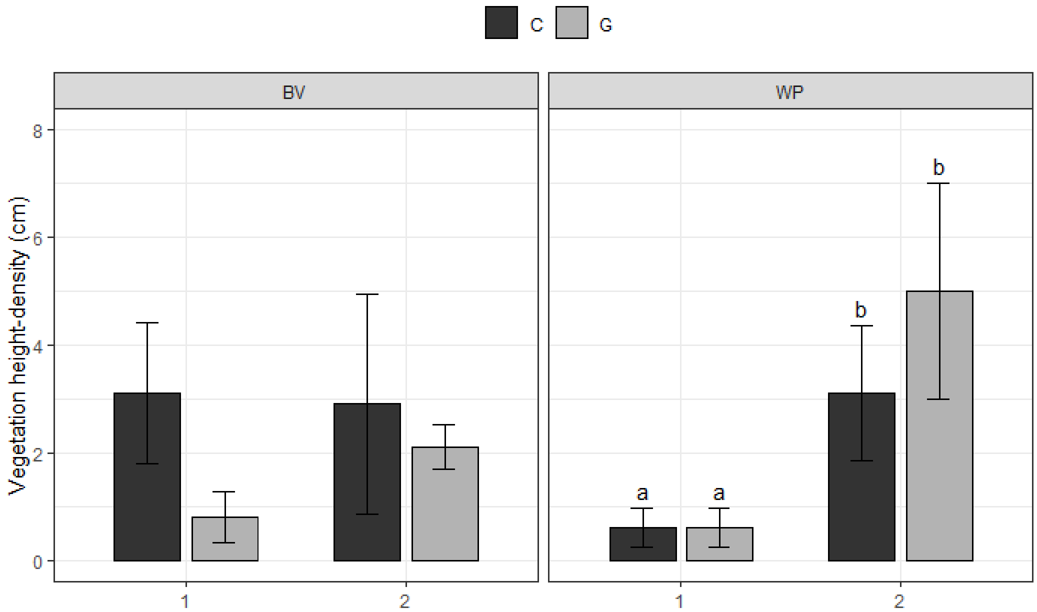

Figure 5. Vegetation height-density at Buena Vista Wildlife Area (BV) and the Johnson East Tract of Western Prairie Habitat Restoration Area (WP) in Wisconsin measured in the first week of May (indicated by 1) and third week of May (indicated by 2) 2019. Grazed areas (G) were rotationally grazed during the 3 prior growing seasons at BV and 2 prior growing seasons at WP and control (C) excluded grazing. Bars with different letters differ within that site at $P < 0.10$

Tables

Table 1. Plant species present at Wisconsin Department of Natural Resources grazing study sites (WA indicates Wildlife Area) measured in late May/early June, 2019 (Buena Vista and Western Prairie) and late May, 2018 (Hook Lake). Species are listed from highest to lowest cover within each functional group and values were averaged over all plots. Native species are bolded and absolute cover indicated in parentheses

Functional group	Site		
	Buena Vista WA	Hook Lake WA	Western Prairie WA
Graminoid	*Poa pratensis* (72)	*Poa pratensis* (92)	*Poa pratensis* (93)
	Bromus inermis (42)	*Bromus inermis* (3)	*Elymus repens* (32)
	Elymus repens (29)	*Elymus repens* (2)	*Phleum pratense* (16)
	Phalaris arundinacea (1)	***Carex* sp.** (0.3)	*Bromus inermis* (3)
	***Carex* sp.** (0.3)	***Andropogon gerardii*** (0.2)	*Phalaris arundinacea* (0.7)
Forbs & broadleaves	***Solidago canadensis*** (14)	***Solidago canadensis*** (40)	***Solidago canadensis*** (30)
	Lotus corniculatus (13)	***Achillea millefolium*** (30)	*Cirsium arvense* (6)
	Carduus nutans (1)	*Hieracium aurantiacum* (22)	*Pastinaca sativa* (2)
	Linaria vulgaris (0.8)	***Solidago rigida*** (12)	***Symphyotrichum lanceolatum*** (1)
	Taraxicum officinale (0.8)	*Taraxicum officinale* (6)	*Trifolium repens* (1)
	Potentilla recta (0.8)	*Trifolium repens* (5)	*Barbarea vulgaris* (1)
	Hieracium aurantiacum (0.5)	*Trifolium pratense* (4)	***Asclepias syriaca*** (0.9)
	Trifolium repens (0.3)	*Daucus carota* (2)	*Cirsium vulgare* (0.7)
	Potentilla simplex (0.3)	***Monarda fistulosa*** (2)	*Berteroa incana* (0.7)
	Trifolium pratense (0.2)	*Cerastium fontanum* (2)	***Lactuca canadensis*** (0.7)
	Solidago gigantea (0.2)	***Geum* sp.** (1)	*Potentilla recta* (0.7)
	Daucus carota (0.2)	***Ratibida pinnata*** (1)	*Silene latifolia* (0.7)
	Erysimum cherianthoides (0.2)	*Glechoma hederacea* (0.8)	*Taraxicum officinale* (0.7)
	Fragaria virginia (0.2)	*Medicago sativa* (0.7)	*Cerastium fontanum* (0.4)
	Stachys palustris (0.2)	***Potentilla simplex*** (0.5)	***Calystegia sepium*** (0.4)
	Urtica dioica (0.2)	*Vicia* sp. (0.3)	*Lotus corniculatus* (0.4)
	Antennaria neglecta (0.2)	***Athyrium felix-femina*** (0.3)	***Symphyotrichum oolentangiense*** (0.4)
	Cerastium fontanum (0.2)	***Agrimonia gryposepela*** (0.2)	***Ambrosia artemisifolia*** (0.2)
	Penstemon digitalis (0.2)	***Anemone virginiana*** (0.2)	***Erigeron annuus*** (0.2)
	-	***Cirsium discolor*** (0.2)	***Geum aleppicum*** (0.2)
	-	***Geranium maculatum*** (0.2)	*Melilotus* sp. (0.2)
	-	*Pastinaca sativa* (0.2)	***Potentilla simplex*** (0.2)
	-	***Polygonatum biflorum*** (0.2)	***Rudbeckia hirta*** (0.2)

	-	***Symphyotrichum* sp**. (0.2)	***Toxicodendron radicans*** (0.2)
	-	-	***Urtica dioica*** (0.2)
Woody & vines	***Spiraea alba*** (2)	*Lonicera x bella* (7)	***Parthenocissus quinquefolia*** (26)
	***Salix* spp.** (0.5)	***Cornus foemina*** (2)	***Zanthoxylum americanum*** (10)
	Rubus hispidus (0.5)	***Rubus allegheniensis*** (1)	***Vitis riparia*** (2)
	Rosa Carolina (0.2)	***Prunus americana*** (0.2)	***Acer negundo*** (2)
	-	*Elaeagnus umbellata* (0.2)	***Vibernum lentago*** (0.7)
	-	***Crataegus*** sp. (0.2)	***Cornus foemina*** (0.2)
	-	*Pinus sylvestris* (0.2)	-
	-	***Quercus rubra*** (0.2)	-
	-	***Vitis riparia*** (0.2)	-

Table 2. Absolute cover of functional groups at three Wisconsin Department of Natural Resources grazing study sites (WA indicates Wildlife Area) measured in June 2016 prior to initiation of grazing treatments

Functional group	% Cover (SE)		
	Buena Vista WA	Hook Lake WA	Western Prairie WA
Poa pratensis	62 (8)	94 (3)	99 (0)
Non native cool-season grasses (excluding *P. pratensis*)	72 (7)	2 (1)	27 (3)
Solidago spp.	40 (9)	59 (4)	41 (4)
Native forbs (excluding *Solidago* spp.)	2 (1)	19 (3)	3 (1)
Non native forbs	14 (4)	32 (9)	4 (2)
Woody plants & vines	7 (3)	12 (2)	60 (6)

Table 3. Absolute cover of functional groups at Buena Vista Wildlife Area (BV), Hook Lake Wildlife Area (HL) and the Johnson East Tract of Western Prairie Habitat Restoration Area (WP) in Wisconsin measured in late May/early June, 2019 (BV and WP) and late May, 2018 (HL). Grazed areas were rotationally grazed during the 3 prior growing seasons at BV and 2 prior growing seasons at WP and HL. Welch's T test performed except where noted by *, where Mann-Whitney U test performed. Differences considered significant when P < 0.10 and are bolded

Functional group		Cover (%)				
		Grazed		Control		
	Site	Mean	SE	Mean	SE	p-value
Native forbs (excluding *Solidago* spp.)	BV	0	0	5	4	0.85*
	WP	5	1	6	2	0.72
	HL	37	4	25	4	**0.07**
Non native forbs	BV	18	5	13	11	0.71
	WP	16	4	9	5	0.36
	HL	36	5	33	11	0.80
Woody plants & vines	BV	4	2	2	1	0.92*
	WP	32	7	43	11	0.46
	HL	8	1	17	4	0.13

Table 4. Plant community richness and diversity and evenness indices for Buena Vista Wildlife Area (BV), Hook Lake Wildlife Area (HL) and the Johnson East Tract of Western Prairie Habitat Restoration Area (WP) in Wisconsin measured in late May/early June, 2019 (BV and WP) and late May, 2018 (HL). Grazed areas were rotationally grazed during the 3 prior growing seasons at BV and 2 prior growing seasons at WP and HL. Welch's T test performed except where noted by *, where Mann-Whitney U test performed. Differences considered significant when P < 0.10 and are bolded. Blanks indicate where native species presence was inadequate for calculating index values

			Grazed		Control		
Response	Species included	Site	Mean	SE	Mean	SE	p-value
Richness	Total	BV	8.8	0.9	7.7	1.8	0.61
		WP	13	1.5	12	2.1	0.72
		HL	13.8	0.6	12.7	1.5	0.54
	Native	BV	2.2	0.6	2.7	0.9	0.69
		WP	6	0.4	6.7	1.3	0.7
		HL	6	0.5	7	1	0.43
Evenness	Total	BV	0.41	0.02	0.53	0.04	**0.08**
		WP	0.37	0.02	0.33	0.01	0.13
		HL	0.34	0.03	0.35	0.01	0.66
	Native	BV	-	-	-	-	-
		WP	0.83	0.1	0.62	0.08	0.26*
		HL	0.79	0.08	0.65	0.08	0.28*
Diversity	Total	BV	0.70	0.02	0.73	0.05	0.72
		WP	0.78	0.02	0.73	0.04	0.41
		HL	0.77	0.02	0.77	0.03	0.99
	Native	BV	0.40	0.12	0.32	0.16	0.70
		WP	0.64	0.06	0.58	0.09	0.62
		HL	0.61	0.04	0.69	0.05	0.28

Chapter 3: Weighing trade-offs on grazing farms: forage quality and yield versus grassland bird habitat refuge

Well-managed pastures on private livestock farms can provide high-quality habitat for declining grassland birds, especially if undisturbed areas within the pasture acreage are set aside during the nesting season. Nearly all of Wisconsin's original grasslands—commonly called prairies—have been lost over the past century and in tandem with this loss, grassland bird populations have declined. Some of these species are categorized as threatened or endangered. Today, many of these birds nest on or near the ground in pastures and hayfields, and previous research in Wisconsin has shown that leaving some portion of this land unharvested during the nesting season—by designating a 'nesting refuge'—can increase reproductive success. This practice, however, is not common due to uncertainty surrounding the extent of losses in forage yield and quality and questions on how to manage these areas following the refuge period.

Laura Judge, Laura Paine, Alicia Dixon and Mark Renz, researchers with the UW-Madison Department of Agronomy and the Agroecology master's degree program, quantified the loss in forage yield and quality under different management scenarios in a nesting refuge established in southern Wisconsin cool-season pasture. Establishing nesting refuges with ungrazed and unharvested forages resulted in losses that varied with different management practices, but only in the year that the nesting refuges were in place.

This information may encourage more farmers to establish nesting refuges and contribute to halting and reversing declines in grassland bird populations. While approximately 100,000 acres of grassland are managed by government agencies and conservation groups for the benefit of wildlife, improving reproductive success—otherwise known as nest success—on the *millions* of acres of pastures and hayfields in Wisconsin provides the biggest opportunity for improvement. Grassland bird nests are in use and vulnerable to trampling or disturbance by grazing cattle from May 1st through August 1st. To avoid predation, they need the protective cover provided by dense plant growth. Hay cutting and grazing, under typical management, remove this cover during the nesting season. Additionally, mowing and grazing can damage nests, leading to nest failure.

While practices such as leaving tall pasture residuals and allowing long intervals between grazing events *may* improve nest success, only nesting refuges are *known* to improve nest success. Ultimately, the results of this study could provide insight on the appropriate level of financial compensation for farmers who establish nesting refuges in pastures and hayfields.

Quantifying the yield loss

With support from the USDA North Central Sustainable Agriculture Research and Education program and the USDA Dairy Forage Research Center, the researchers partnered with Paine Family Farm, a grass-based beef farm in Columbus, to quantify the loss of forage yield and quality when setting aside land for a nesting refuge. The farm seasonally grazed approximately 22 beef cows, three yearlings and 14 calves on cool-season pasture dominated by Kentucky bluegrass, timothy and orchardgrass.

In addition to looking at forage losses on the nesting refuge land in that year, the researchers also tested whether unharvested forages in a nesting refuge might suppress pasture growth the following season. They used two management strategies to remove built-up plant material from nesting refuge treatments: hay harvest and burning. Plots were established (50 x

50 ft) and treatments randomly assigned within three replicated blocks. Treatments are outlined in Table 1.

Table 1. Four treatments replicated three times over two years in a cool-season grass pasture.

	Control	Nesting Refuge Treatments		
Treatment	**G**	**N**	**N + H**	**N + B**
2018	Graze	Nesting refuge	Nesting refuge, then hay Aug. 1	Nesting refuge
2019	Graze	Graze	Graze	Spring burn, then graze

Nesting refuge treatments were established in 2018. Two of these treatments were unharvested the entire year (N, N + B), while hay was harvested from the third on August 1st (N + H). The control plots were rotationally grazed (G) with grazing events in June, August and October. In 2019, one of the unharvested nesting treatments (N + B) was burned at the beginning of April. All plots were returned to normal grazing rotations in 2019, with grazing events in July, August and September. Paddocks that included the treatment plots were grazed when the farmer determined readiness by visual evaluation, and cattle were moved out of the paddock when forage heights reached four to eight inches. This resulted in grazing intervals as short as four hours and long as 24 hours for each of the three paddocks containing the replicated blocks.

Available forage was measured before each grazing event, and prior to hay harvest in N + H, by clipping plants within a ¼-meter squared quadrat to a height of four inches. Samples were dried and weighed, then ground and analyzed using Near Infrared Reflectance Spectroscopy to determine forage quality. Relative forage quality (RFQ)—an index that combines various measures of forage quality—was calculated for G and N + H in 2018, and all treatments in 2019 (Figure 1). Percent crude protein—an important component of forage quality—was also measured (Figure 2), as well as season-long forage yield in tons of dry matter per acre (Figure 3).

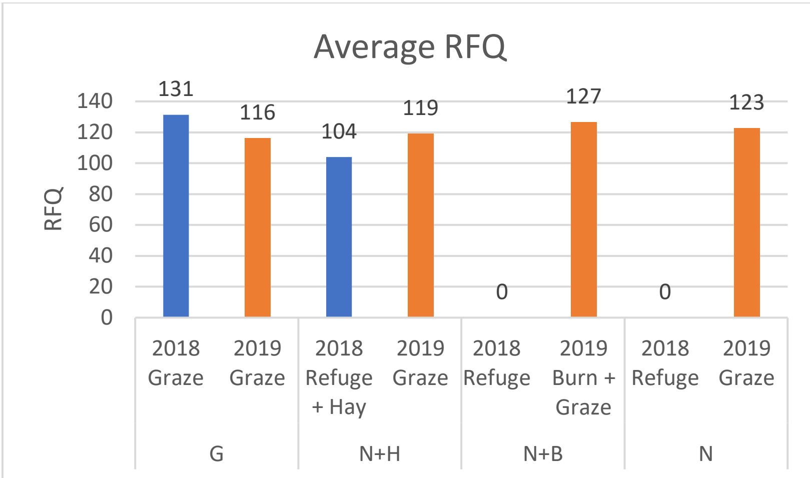

Figure 1. Average relative forage quality (RFQ) for 2018 and 2019 grazing seasons for each treatment

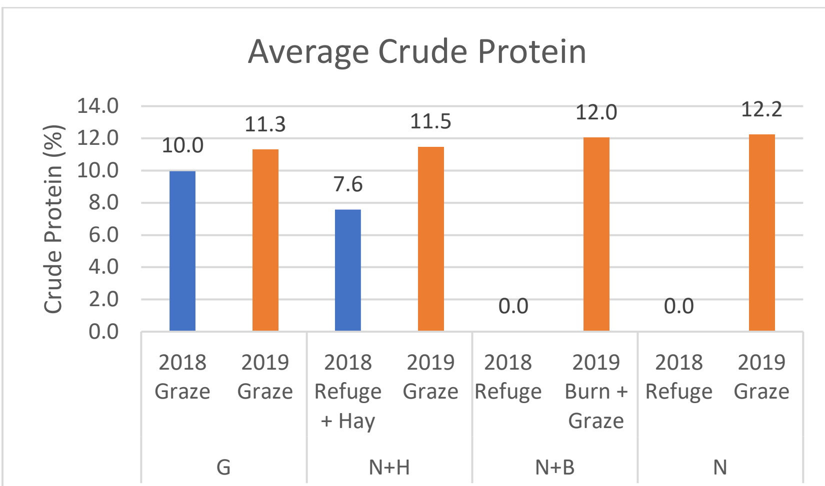

Figure 2. Average crude protein (%) for 2018 and 2019 grazing seasons for each treatment

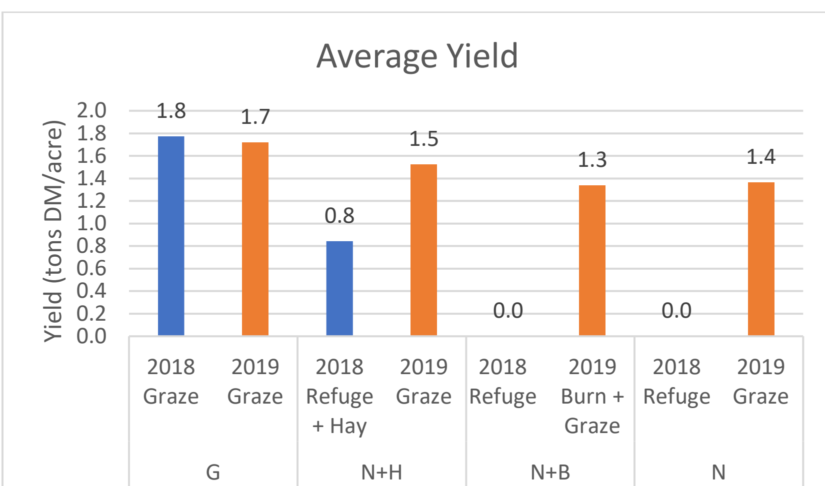

Figure 3. Average season-long yield (tons of dry matter/acre) for 2018 and 2019 grazing seasons for each treatment

In 2018, establishing an unharvested nesting refuge followed by an August 1st hay harvest resulted in a 55% reduction in forage yield compared to normal grazing. Additionally, the harvested hay had an RFQ 21% lower and crude protein level 24% lower than an average grazing event in that year. The monetary loss associated with these reductions can be roughly characterized using hay grades and prices. University of Wisconsin Extension (Hay Market Report, May 25, 2020) categorizes the RFQ of the grazed treatment as Grade 1 hay valued at \$159/ton (large square bale). The hay harvested following the refuge treatment was in the Grade 2 category valued at \$132/ton (large square bale). Considered together, results indicate a moderate loss in income due to reduced forage quality and a major loss in income due to forage yield (55% loss) on land in the nesting refuge.

In 2019, when all plots were returned to normal grazing management, minimal variability was observed. Plots that were in nesting refuge in 2018 and spring burned in 2019 had the highest average RFQ at 127 and normal grazing (both years) plots had the lowest RFQ at 116. Similarly, plots in nesting refuge in 2018 had the highest crude protein at 12.2% and normal grazing (both years) plots had the lowest at 11.3%. While statistical tests could not be performed on these measures, the range of values does not indicate large variability or reductions in forage quality compared to normal grazing and may even suggest forage quality trended higher in the refuge treatments. Additionally, statistical tests of forage yield indicated no differences between treatments. Plots in nesting refuge in 2018 and spring burned in 2019 had the lowest yield at 1.3 tons of dry matter/acre and normal grazing (both years) plots had the highest yield at 1.7 tons of dry matter/acre.

These results suggest that there are no losses in forage quantity or quality when pasture area is returned to normal grazing the season following a nesting refuge. While researchers did *visually* observe differences between treatments in early spring, such as higher quality and lower quantity of forages in spring-burned plots compared to unharvested nesting refuges (Picture 1), plots were visually similar when grazing and forage measurements began in early July. Therefore, grazing earlier in the season may yield different results than those shown here. Additionally, both growing seasons (April through the end of August) studied were wetter than normal (based on the 30-year average of precipitation) with 2018 precipitation 37% higher and 2019 13% higher than normal.

Picture 1. Taken April 24, 2019. Both plots were unharvested in 2018. The plot on the right was burned one month prior (N + B).

What do the findings mean for farmers?

This study only captures the effects of these management scenarios on one farm, and additional experiments should be performed across the state. Nevertheless, the 2018 results highlight that the loss to a farmer when setting aside land for a nesting refuge can be significant.

The 2019 results indicate that, in this case, there was no legacy effect of the previous season's management on forage yield and quality when the land was returned to normal grazing after a nesting refuge season. This suggests that the loss incurred by the farmer only happens *on the unharvested land in a designated nesting refuge in that year*. That same land returns to normal productivity the following year. Further studies should be done to verify this result in different years and environments.

What does this study mean for birds?

Previous research on grassland birds suggests different species prefer different habitat characteristics (for example, high or low vegetation height and density, or shallow or deep litter). In this study, the researchers sought to determine which species might respond to managing pastures with nesting refuges. While actual use of the nesting refuges by bird species was not measured in the study, inferences about the suitability of the habitat were made from measurements of vegetation height and density and litter depth and cover. Measurements taken in the spring in 2018 indicate that a nesting season refuge of an appropriate size could provide habitat for species that prefer medium vegetation height, density and litter layer, such as eastern meadowlark and bobolink. These results are in line with other research on grassland birds, which shows that certain species prefer pastures and hayfields as habitat.

What does this study mean for decision-makers?

These results highlight the opportunity to help slow or reverse the decline in numbers of these grassland bird species though altering pasture and hayfield management. While direct and cost-share payments to farmers for conservation practices already exist through government agencies and conservation organizations, none fully compensate farmers for direct losses associated with a nesting refuge. These results could be used as a starting point for determining appropriate compensation levels for farmers setting aside land for a nesting refuge. These payments could be calculated based on the farm's average income per acre or using current hay prices and on-farm measurements of forage yield and quality. Providing appropriate compensation through conservation funding is likely a key component of improving grassland bird nest success in pastures and hayfields in Wisconsin.

Laura Judge says, "Most grassland bird species are currently charting a course toward extinction. This study suggests a new path forward, where conservation funding can help improve nest success in pastures and hayfields while supporting the farmers who manage them." Further validation of these results through additional research will help promote the conservation of grassland birds.