VEGETATION PATTERN AND RESPONSE IN THE CONTEXT OF HETEROGENEOUS PASTURE MANAGEMENT IN SOUTHEASTERN NEBRASKA

By

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Abstract:

Prior to European settlement, the interaction of fire and grazing was a dominant disturbance and driver of vegetation structure and diversity in the Great Plains. Periodic wildfires and anthropogenic fires created burned areas on which grazing animals would congregate and consume highly palatable forage regrowth. In recent years, rangeland management practices have sought to reintroduce this interaction into confined pasture settings. Several studies have shown benefits for native plants and wildlife. Most of such studies have been conducted on conservation lands and experimental research stations. This study was conducted on both public conservation land and private working rangeland in southeastern Nebraska. Depending on the scale and functional level examined, vegetation can either be influenced by or can have potential confounding influences on rangelands managed with fire and grazing.

At the pasture level, vegetation structure was found to be more heterogeneous in pastures managed with patch burning compared to pastures managed in a more traditional manner. Significant differences between measures of plant diversity between treatments were not observed. Plant community composition was found to vary across the landscape due largely to ownership and regional edaphic characteristics. Examining specific butterfly host plant species, no significant differences were observed between treatments nor between differences in time since fire. At the landscape scale, woodland vegetation was found to have an influence on the abundance of grassland birds and a potentially confounding influence on heterogeneous pasture management.

Incorporating fire and grazing onto private working rangelands can create vegetation heterogeneity, which has shown benefits for wildlife. Plant genera that have evolved with fire and grazing in the Great Plains should be adapted to, and potentially benefit from this management. Further, depending on vegetation that exists in the greater landscape, such as woodland vegetation, goals and objectives can be confounded.

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CHAPTER I

FIRE AND GRAZING IN WORKING RANGELANDS: VEGETATION STRUCTURE, DIVERSITY, AND COMPOSITION.

Abstract

Private ownership accounts for much of the rangeland in the Great Plains. Rangeland management practices that are sensitive to landowner goals and demonstrate benefits for biological diversity may encourage adoption of conservation practices on private working rangelands. The recoupling of fire and grazing interaction to create pasture heterogeneity is beneficial for many wildlife species. Our study examines the influence of fire and grazing treatments on vegetation structure, diversity, and composition on both private working rangelands and publicly managed rangelands in southeastern Nebraska. Treatments consisted of patch burn grazed pastures and traditionally managed pastures; both were moderately stocked. Vegetation structural measurements included vegetation height and visual obstruction. We compared diversity measures between treatments and ownership using species richness, Shannon diversity index, and floristic quality index. A detrended correspondence analysis (DCA) was used to examine patterns of plant community composition in study pastures. At the treatment level, we found no difference in vegetation height and visual obstruction between treatments and ownership. At the patch level, we found higher mean standard deviation (heterogeneity) of vegetation height and visual obstruction within patch burn treatment

pastures than in traditional treatment pastures. We did not observe a difference in heterogeneity of patches with pasture ownership. Diversity measures were not significantly different between treatments. We did find higher richness and Shannon diversity in publicly owned pastures compared to private pastures. Our DCA suggested that plant species composition was correlated to pasture ownership and soil properties. These results suggest that pasture vegetation heterogeneity can be established in private working rangelands while still maintaining cattle production. While we found differences in diversity with pasture ownership, it is difficult to determine whether this difference is biologically significant.

Introduction

Private ownership accounts for 212 of the total 316 million hectares of grazing land that exists in the United States (Kimble et al. 2000, Lubowski et al. 2006). Maintaining biodiversity on private working lands, in addition to protected lands, is an important goal of conservation efforts (Polasky et al. 2005). To achieve this goal, developing strategies that are supportive of landowner goals and can sustain biological diversity on rangelands is imperative. Traditional rangeland management has long encouraged the homogenization of vegetation structure throughout pastures by promoting uniform livestock grazing distribution (Fuhlendorf and Engle 2001, Bailey 2004). This management approach is built on a utilitarian conservation paradigm of sustaining livestock production but conflicts with historic disturbance patterns that existed in the Great Plains before European settlement (Fuhlendorf et al. 2012). Prior to European settlement, anthropogenic and natural fires resulted in a mosaic of recently burned areas on which grazing animals would congregate and selectively graze regrowing vegetation (Anderson 2006, Fuhlendorf et al. 2009).

A recent management approach, known as patch burning, attempts to restore historical disturbance regimes of fire and grazing on rangelands. While clearly the scale and extreme range of disturbance events cannot be restored on a fragmented and privately owned landscape, patch burning attempts to more closely imitate the mosaic of rangeland vegetation structure that would have existed in the historic landscape of the Great Plains (Fuhlendorf and Engle 2001). Through patch burning, grazing animals consume the vegetation regrowth that occurs in recently burned patches and avoid the denser, less palatable, and dead vegetation in unburned patches (Allred et al. 2011). This resulting vegetation heterogeneity has shown beneficial results for many wildlife species that are dependent on heterogeneous rangeland vegetation structure (Coppedge et al. 2008, Fuhlendorf et al. 2010, Doxon et al. 2011).

In addition to structural heterogeneity, the selective pressures from fire and grazing animal interactions are important drivers of plant diversity (Collins et al. 1998). Rangeland management that promotes homogeneity should in turn promote a reduction in plant diversity (Whittaker and Levin 1977), though it is difficult to understand the importance of these predictions on landowner goals. Further, the temporal scale required to monitor changes in plant species diversity can be extensive and incorporate many natural and anthropogenic confounding factors. While plant species diversity can be important for some wildlife species, such as insects (Knops et al. 1999), evidence suggests that heterogeneity is an important driver of overall biodiversity (Fuhelndorf et al. 2006).

Research conducted on private lands can have confounding influences due to landowner management, goals, or social constructs that can compromise research designs (Hilty and Merenlender 2003). In part because of this, many fire and grazing studies in rangelands have been conducted on conservation lands or experimental research stations, as opposed to private working rangelands (Collins et al. 1998, Fuhlendorf et al 2006). This has led to a gap between the results of empirical studies and rancher perceptions of management (Teague et al. 2013). If landowners are encouraged to adopt conservation efforts that maintain biodiversity and landowner goals, empirical research conducted on private working rangelands can help serve as a proof of concept (Miller et al. 2012).

This study, located in southeastern Nebraska, was conducted on both private working rangelands and publicly owned lands, which provides a unique opportunity for comparison. Since many patch burn studies have been conducted on conservation lands and research stations, examining privately owned pastures that may have land-use legacy differences is of interest (McGranahan et al. 2013a, Debinski et al. 2011). We compared patch burn treatments with traditional treatments wherein the entire pasture is burned using prescribed fire. We evaluated plant community composition, vegetation structure, and three measures of plant diversity to determine if the short term application of patch burning could be applied to private working lands and result in changes in plant community composition, structure, and diversity. The three metrics chosen, richness, Shannon diversity index, and floristic quality index, were chosen because of their range of subjectivity for rare and conservative plant species. Richness gives equal weight to each plant species present and provides a comparison across pastures between total number species. Shannon's diversity index incorporates evenness of plant species, in addition to richness, to provide a metric of diversity. The floristic quality index assigns a rank, or coefficient, to each species based on its perceived conservation value as determined by local botanical experts, which is then summed and divided by the square root of the total number of species. This metric is highly subjective but attempts to determine the quality of the plant community. We postulate that through the use of patch burning, heterogeneity can be established in private working pastures. Further, we expect that diversity measures will not significantly differ between patch burn treatment and traditional treatment pastures. Finally, we hypothesize that landscape determinants will influence plant community composition and explore how such determinants may influence management goals and objectives.

Methods

Study Site

Fifteen study sites were located and analyzed from 2009-2011 in southeast Nebraska in the counties of Gage, Jefferson, Pawnee, and Johnson (Fig. 1). Pastures were typical for the region and ranged in size from a maximum of 67.6 ha to a minimum of 28.3 ha with an average of 41.3 ha. Eight pasture units were treated with patch burn management, where a different 1/3 of the pasture was burned each year of the study. The remaining seven pastures were burned in their entirety the first year of the study and considered traditional management pastures. All prescribed burns were conducted in the spring season for all pastures. At the completion of the study in 2011, all pastures had been burned in their entirety with the only difference associated with the spatio-temporal pattern of the fires. Together, seven of the traditional treatments, and seven of the patch-burn treatment pastures were considered to be paired, based on proximity, leaving a single, unpaired patch-burn treatment pasture. Eight pastures were privately owned while seven were owned by the Nebraska Game and Parks Commission. Dominant grass species include *Andropogon gerardii* Vitman, *Schizachyrium scoparium* (Michx.) Nash, *Sorghastrum nutans* (L.) Nash., *Bromus inermis* Leyss., and *Poa pratensis* L. Pastures were comparably stocked across both treatments primarily with cow/calf cattle herds. In 2009, one patch burn treatment and one traditional treatment pasture were stocked with yearlings and a mixed herd. Stocking rates varied slightly between pastures (2.22 ± 0.08 AUM/ha) and were determined using NRCS recommended stocking rates. In 2009, cattle turn-in dates were between April 1st and April 15th. In 2010 and 2011, cattle turn-in dates were between May 1st and May 15th. Cattle take-out dates for all years were between October 1st and October 15th. Annual mean temperatures for the region in 2009, 2010, and 2011 were 9.7°C, 10.5°C, and 10.7°C (www.ncdc.noaa.gov). Annual precipitation totals for the region in 2009, 2010, and 2011 were 665 mm, 833 mm, and 709 mm (www.ncdc.noaa.gov). Soils in the study sites ranged between silt loam, loam, and clay loam.

Structural measurements

Vegetation visual obstruction was measured using a Robel pole, which provides an estimation of vegetation density (Robel et al. 1970). Vegetation height was measured at the same time as visual obstruction measurements by recording the highest point at which vegetation crossed between the observer and the Robel pole. Visual obstruction was determined by recording the lowest point visible on the Robel pole. Observations were made 2 m from the pole, and 1 m above the ground surface. The pole was marked from 0 to 150 centimeters at 1 cm increments. Measurements were made at each of the four cardinal directions at 10 m intervals along a 100 m transect, for a total of 40 observations per transect. A total of 12 transects occurred in each pasture with four transects occurring in each of the three resulting treatment patches. Vegetation height and visual obstruction were analyzed by comparing pasture means using Welch's two-sample t-test. Pasture vegetation heterogeneity was also examined by obtaining the standard deviation of patch burn treatment and traditional treatment patch means for vegetation height and vegetation visual obstruction measurements. Transects within the traditional treatment pastures were grouped into three hypothetical patches, with four transects per patch, for comparison. Pasture vegetation heterogeneity was also separated by year for analysis to examine between year differences. Analysis of variance, followed by Tukey's HSD, was used to compare patch burn treatment and traditional treatment pastures by year. Heterogeneity of vegetation height and vegetation visual obstruction were also compared between publicly and privately owned pastures. We also examined heterogeneity of vegetation height and visual obstruction for publicly and privately owned patch burn treatment pastures. Analyses were conducted using R statistical software version 3.0.3 (R Development Core Team 2014).

Plant Species Composition

Plant species composition surveys were conducted on 13 of the pastures (seven patch burn and six traditional treatments) each of the three years except for four pastures that were sampled only twice. Surveys were conducted between June 12th and August 11th in all three years. Twelve, 100 m transects were used per pasture (Fig. 2), except one patch burn treatment pasture and one traditional treatment pasture, where only six transects were used due to avoidance of areas that showed evidence of historically tilled soils. Transects were established in 2009 using aerial imagery and were placed evenly across each study site to reside in suitable vegetation. Patch burn treatments contained four transects per treatment patch, except for one pasture which contained two transects per patch. Each transect was selected using ESRI's ArcGIS (ESRI 2013), and three waypoints, start, midpoint, and end, for each transect were uploaded to GPS units that were used in the field to determine transect locations. A 1 m^2 PVC frame was used to estimate percent coverage for each species found within the frame. The frame was placed along each transect at 10 m intervals for a total of 10 samples per transect (Fig. 2). Coverage estimates for each species were estimated at 5% intervals from 0% to 100%, with 5% being the smallest possible value for a species present within the frame. Plant species that could not be identified were collected and given a unique temporary identification name, pressed, and later identified at the University of Nebraska – Lincoln's Bessey Herbarium. Plants that could not be identified to species level were identified to genus or family instead.

Species richness was compared between patch burn treatment and traditional treatment pastures. Diversity was measured using the Shannon diversity index (H') where H' = -$\sum i\ pi \log pi$ and compared between patch burn treatment and traditional treatment pastures (Magurran and McGill 2011). Floristic quality index using coefficient of conservatism values, as developed by Gerry Steinauer of the Nebraska Game and Parks Commission, where FQI = $\sum C\ /\sqrt{n}$, was compared between patch burn treatment and traditional treatment pastures (Swink and Wilhelm 1994). Welch's two-sample t-test was used to compare mean richness, Shannon diversity index, and floristic quality index, between the patch burn treatment and tradition treatment pastures, as well as between public and private pasture ownership. Analyses were conducted using R statistical software version 3.0.3 (R Development Core Team 2014).

A detrended correspondence analysis (DCA) was conducted on the plant community data using CANOCO 5 (ter Braak and Šmilauer 2012) to identify potential gradients and patterns in the plant species composition. DCA was chosen for its ability to remove potential arch effects and compression of the ordination axes towards the middle (McCune 2002). Rare plant species were down weighted, and a log transformation was used for the DCA analysis.

Results

When internal pasture heterogeneity was analyzed at the patch level, mean standard deviation of vegetation height was significantly greater for patch burn treatment pastures compared to traditional treatment pastures for each of the three years of the study (Fig. 3a). In other words, the vegetation height had greater variation between patches in the patch burn treatment pastures compared to traditional treatment pastures, which were more homogenous. The mean standard deviation of vegetation visual obstruction was significantly greater in both 2009 and 2010 inside patch burn treatment pastures compared to traditional treatment pastures (Fig. 3b). However, no significant difference was found in 2011 (Fig. 3b). The mean standard deviation of vegetation height was not significantly different between public and private pastures for any of the years in the study (Fig. 4a). Also, the mean standard deviation of the vegetation visual obstruction was not significantly different between public and private pastures (Fig. 4b). When comparing between private and public only in patch burn treatment pastures across all three years, we did not see a significant difference in either vegetation height or vegetation visual obstruction (Fig. 5).

Vegetation height, averaged across the entire pasture and across all three years of the study, did not significantly differ between patch burn treatment (40.60 cm ± 2.30 SE) and traditional treatment (39.07 cm ± 2.72 SE) pastures ($p = 0.68$, $t = -0.43$, $df = 12.27$). Also at the treatment level, vegetation visual obstruction did not significantly differ between patch burn treatment (8.63 cm ± 1.06 SE) and traditional treatment (8.60 cm ± 1.30 SE) pastures ($p = 0.98$, $t = -0.02$, $df = 12.02$).

Species diversity, regardless of diversity metric, was not found to be significantly influenced by treatment. A total of 242 plant species or groups were identified across all study sites with 196 of those being native species, 40 being exotic species, and 6 genus groups that include both native or exotic species. Richness, the least subjective diversity measure which gives equal weight to native and exotic species, did not significantly differ between patch burn treatment pastures and traditional treatment pastures (Table 1). Shannon diversity index, which incorporates both richness and evenness, also did not differ between treatments (Table 1). Floristic quality index, which is assigns weights to species based on perceived conservation rank in the ecosystem, also did not differ between treatments (Table 1). However, richness and Shannon diversity index were significantly greater in pastures publicly owned by the Nebraska

Game & Parks Commission compared to privately owned pastures (Table 1). Floristic quality index did not differ significantly between public and private pastures (Table 1).

Plant composition showed two distinct trends across the study area based on ownership and regional edaphic characteristics. The first two axes of the DCA explained 36.12% of the total variation in the compositional data (Fig. 6). The eigenvalue of axis one was 0.198 and the eigenvalue of axis two was 0.090. The gradient lengths of these two axes were 1.76 and 1.25, respectively. The first DCA axis separates site scores along a gradient of private or public ownership. Commonly occurring plants at the extremes of the first DCA axis include *Bouteloua curtipendula* (Michx.) Torr. for private pastures with a species score of -0.28 and *Solidago canadensis* L. for public pastures with a species score of 2.95. The second DCA axis separates sites by biologically unique landscapes (BUL), which are a function of soil texture with sandy soils occurring in Sandstone Prairies BUL and loam soils occurring in the Southeast Prairies BUL (Schneider et al. 2011). Species scores along the second axis indicate that *Oligoneuron rigidum* (L.) Small (-1.3) was highly associated with the Sandstone Prairies BUL, while *Bromus inermis* Leyss. (1.78) was highly associated with the Southeast Prairies BUL.

Discussion

The inclusion of conservation practices on private working rangelands has been cited as a goal to maintain biodiversity in the Great Plains (Polasky et al. 2005). Market forces and social constructs may impede the adoption of sound rangeland management practices (Krueter et al. 2006). To help overcome these obstacles, examples of management strategies applied to working lands that support both long-term productivity and biological diversity are needed (Miller et al. 2012). In this study, we found that vegetation structural heterogeneity, which is important for many wildlife species, can be established in private working pastures through patch burning. We also found no significant differences in plant species diversity between patch burn treatment and traditional treatment pastures, and we attribute determinants in plant community composition across the landscape to a function of land-use legacy and soil differences.

Mean vegetation visual obstruction and height across patch burn treatment and traditional treatment pastures was not found to be significantly different, at the treatment level, which was expected since cattle stocked at similar rates should remove similar amounts of forage from each pasture. However, we did see higher mean standard deviation, or variation, in patch burn treatment pastures compared to traditional treatment pastures. This is due to focal grazing on the most recently burned patch, which contains vegetation regrowth that has higher palatability and higher crude protein than unburned patches (Allred et al. 2011). This caused the unburned patches to be avoided and thus allowed for greater structural heterogeneity. In contrast, the traditional treatment pastures were burned in their entirety the first year of the study, and thus some portions of the pasture were likely not utilized by cattle. When pasture ownership was examined we did not see a significant difference in mean standard deviation, suggesting that heterogeneity was established in both public and private pastures. As further evidence, when only treatment pastures were compared between public and private pastures we also found no significant difference in mean standard deviation. This suggests that heterogeneity can be established in private working rangelands with similar results as those seen on public and research lands. While the use of patch burn management has been shown to create vegetation structural heterogeneity, it should be noted that stocking rate is still a key driving factor of rangeland vegetation structure (Holechek et al. 2004, McGranahan et al. 2013b).

When comparing diversity measures we did not find significant differences between patch burn treatment and traditional treatment pastures. However, we did find significantly lower richness and Shannon diversity index in private pastures compared to public pastures. Potential reasons for this might be differences in land-use history, such as a history of heavy stocking or greater herbicide use in private pastures compared to more conservation oriented public lands. Since private working rangelands are largely managed for commodity production it stands to reason that attempts to maximize profit may occur to the detriment of plant species diversity. For instance, several grazing models predict a reduction in plant diversity as disturbance increases due to high cattle stocking rates (Houston 1979, Milchunas et al. 1988). While the difference in plant diversity measures is statistically significant for pasture ownership, it is unclear whether these differences have a true biological significance on these rangelands. Further, while we see a difference in diversity metrics between pasture ownership, it is difficult to expand on effects of fire and grazing on plant species diversity in these study sites due to the relatively short period of study.

Examining broader landscape plant composition patterns, the first axis of the DCA appears to separate site scores based on their ownership with private pastures having a low site score and public pastures having a high site score. The second DCA axis appears to separate site scores based on physical location. These sites appear to be separated physically into biologically unique landscapes (Schneider et al. 2011). Sites with a high site score on both axis 1 and axis 2 are associated with the Southeast Prairies BUL, which are characterized by loam soils. Sites with a low site score on axis 1 and axis 2 are associated with the Sandstone Prairies BUL, which are characterized by sandy soils. Local edaphic factors appear to be an important driver of plant species composition across the studied native rangelands in southeastern Nebraska. This can potentially influence management goals and objectives by potentially overriding the effects of applied treatments.

Implications

The results of this study provide evidence that vegetation structural heterogeneity can be established in private working rangelands. Even with a delay between the application of prescribed fire and cattle turn-in dates in 2010 and 2011, structural variation within the patch burn treatment pastures was still greater than in traditional treatments . This delay could have potentially caused the regrowth in the recently burned patch to become taller and less palatable. The results might have been discontinuous use of the recently burned patch and less structural heterogeneity in the patch burn treatment pastures. However, this did not appear to be the case in our study.

Given the short duration of this study, there was little evidence of changes in plant diversity due to management. In turn, we cannot draw a valid conclusion about fire and grazing management techniques influence on plant species diversity. A difference in plant diversity measures was observed between privately and publicly managed pastures, but whether these differences have biologic importance is unknown. However, patterns in the landscape plant community composition suggest that ownership and edaphic factors influence the local plant community composition. This is important when determining goals and objectives when implementing a fire and grazing prescription for the benefits of natural resources. It may be that issues arise due to the landscape or local factors, such as soils in this case, that limit or determine final outcomes.

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Table 1. Mean ± SE plant community diversity measures between traditional and patch burn treatment pastures located in southeastern Nebraska.

Diversity Measure	Traditional Treatment	Patch Burn Treatment	df	*t*	*p*
Richness	88 ± 4.7	87 ± 1.9	6.6	0.29	0.78
Shannon diversity index	3.4 ± 0.08	3.3 ± 0.04	7.9	0.56	0.59
Floristic quality index	28.8 ± 1.2	28.8 ± 1.4	11.0	0.05	0.96

Table 2. Mean ± SE plant community diversity measures between privately and publicly owned pastures located in southeastern Nebraska.

Diversity Measure	Private	Public	df	*t*	*p*
Richness	82 ± 3.0	92 ± 2.4	9.9	-2.46	0.03
Shannon diversity index	3.3 ± 0.03	3.5 ± 0.06	8.7	-2.81	0.02
Floristic quality index	27.5 ± 1.4	29.9 ± 1.1	9.8	-1.37	0.20

Figures

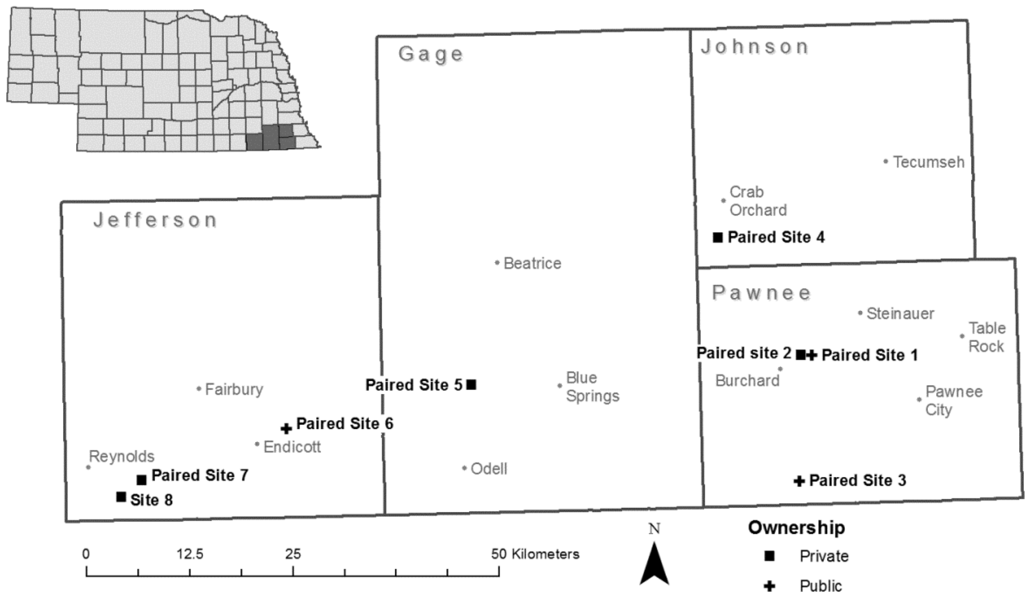


Figure 1. Study sites located in southeast Nebraska. Privately owned pastured are represented by black squares. Publicly owned pastures are represented by black crosses.

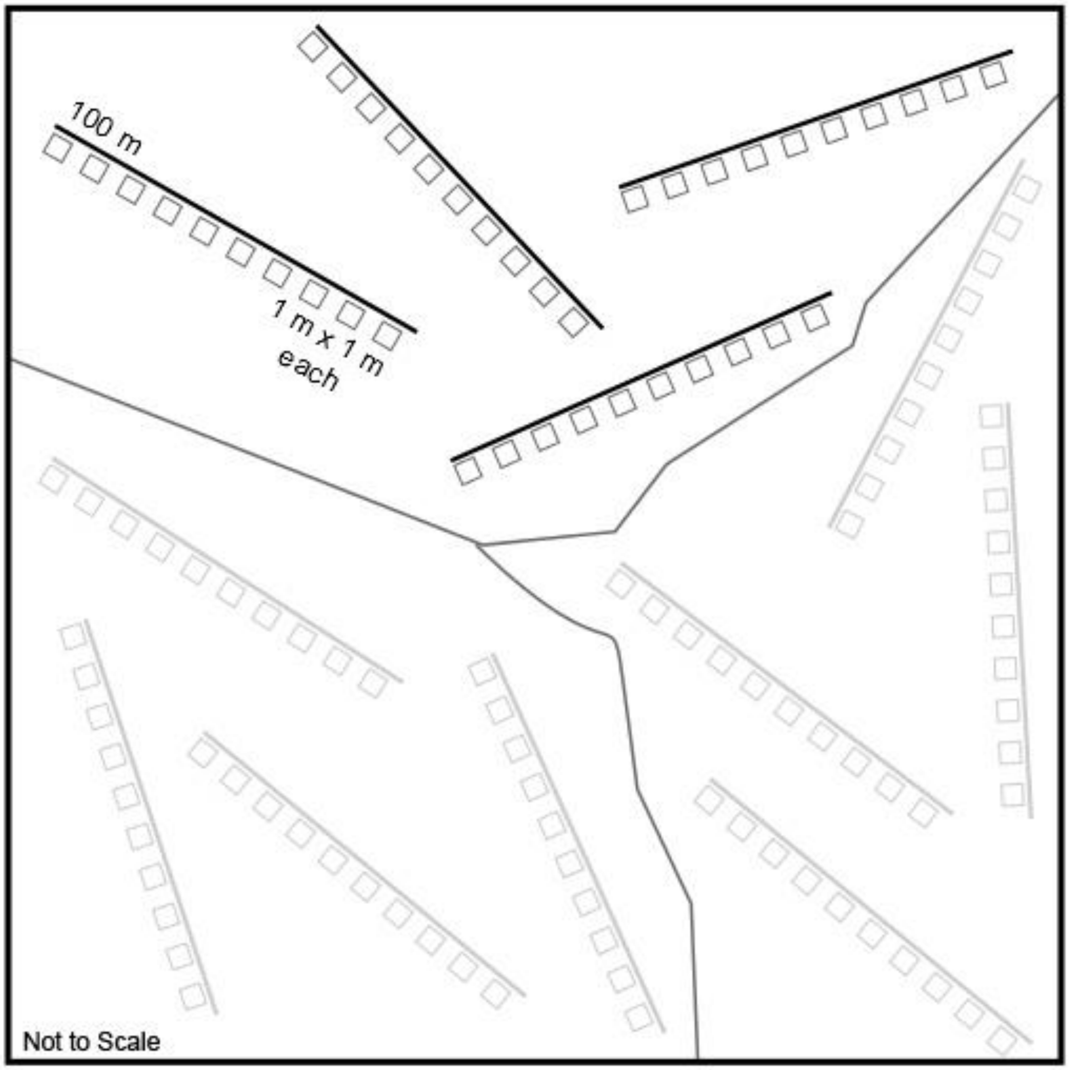


Figure 2. Example of transects with plots used to estimate plant community composition arranged in a hypothetical patch burn treatment pasture.

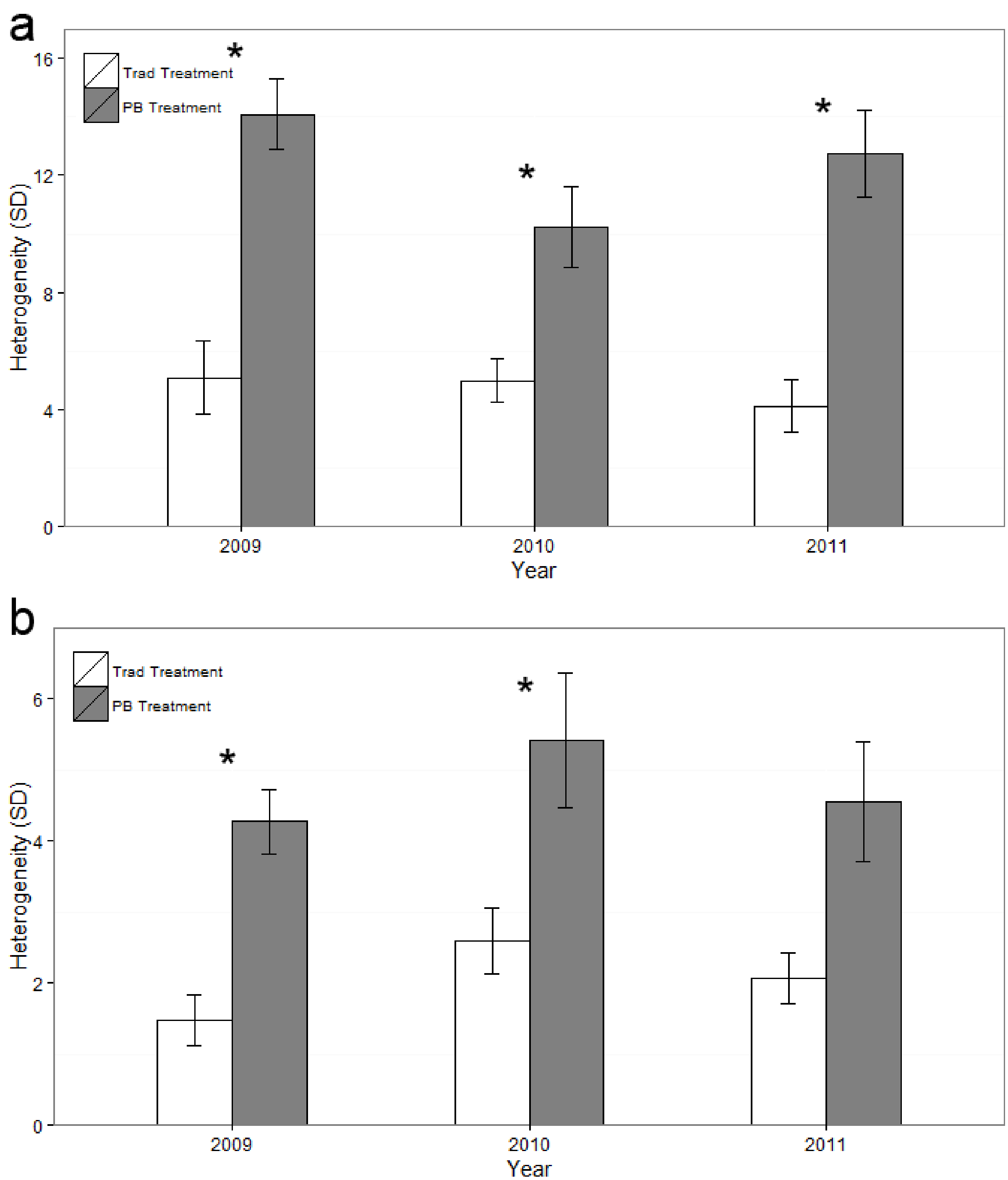


Figure 3. Mean ± SE of a) standard deviation of vegetation height between treatment and year and b) standard deviation of vegetation visual obstruction between treatment and year. Dark grey bars represent patch burn treatments, and white bars represent traditional treatment pastures. Asterisks indicate significance differences between treatments within year (*p* < 0.05).

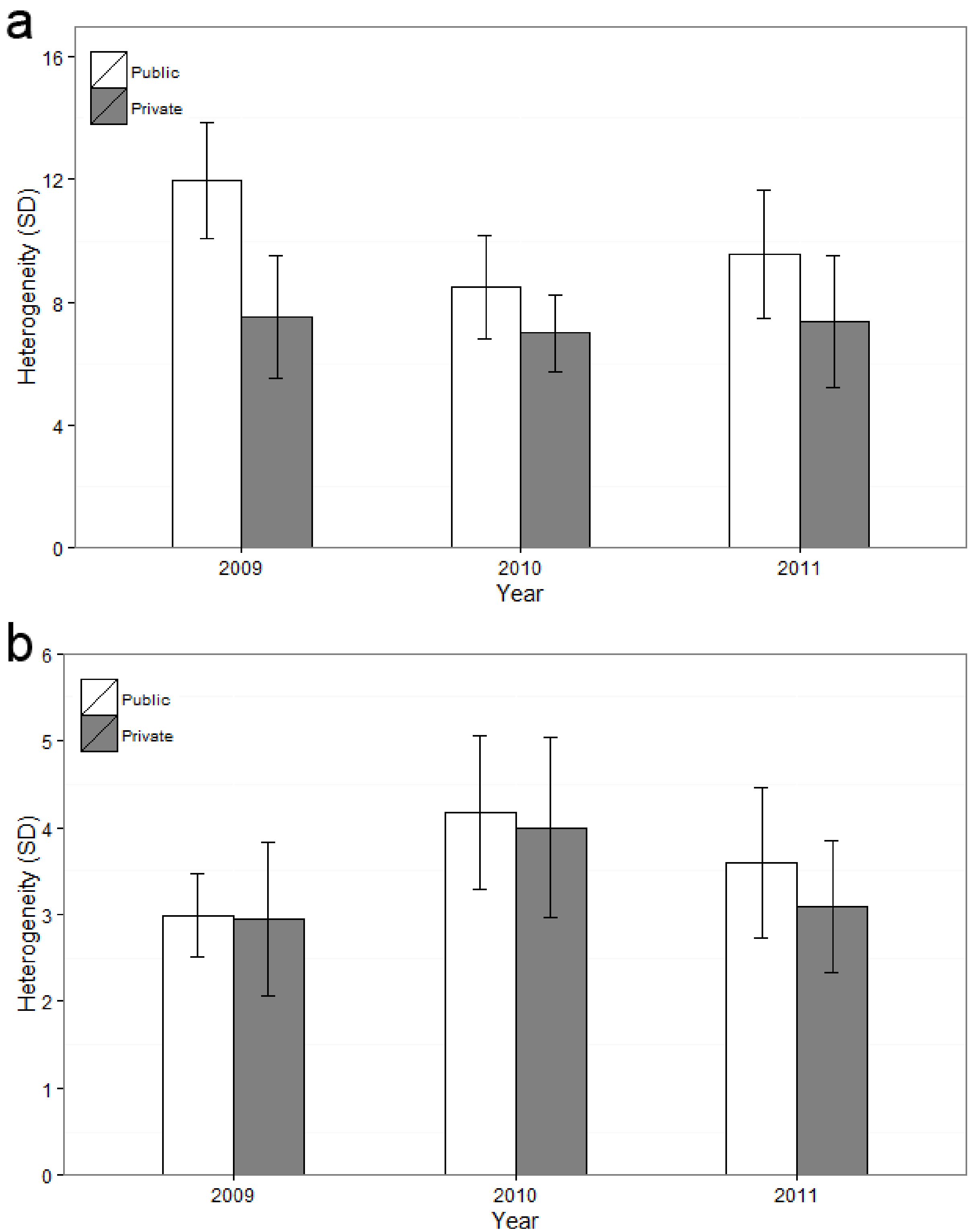
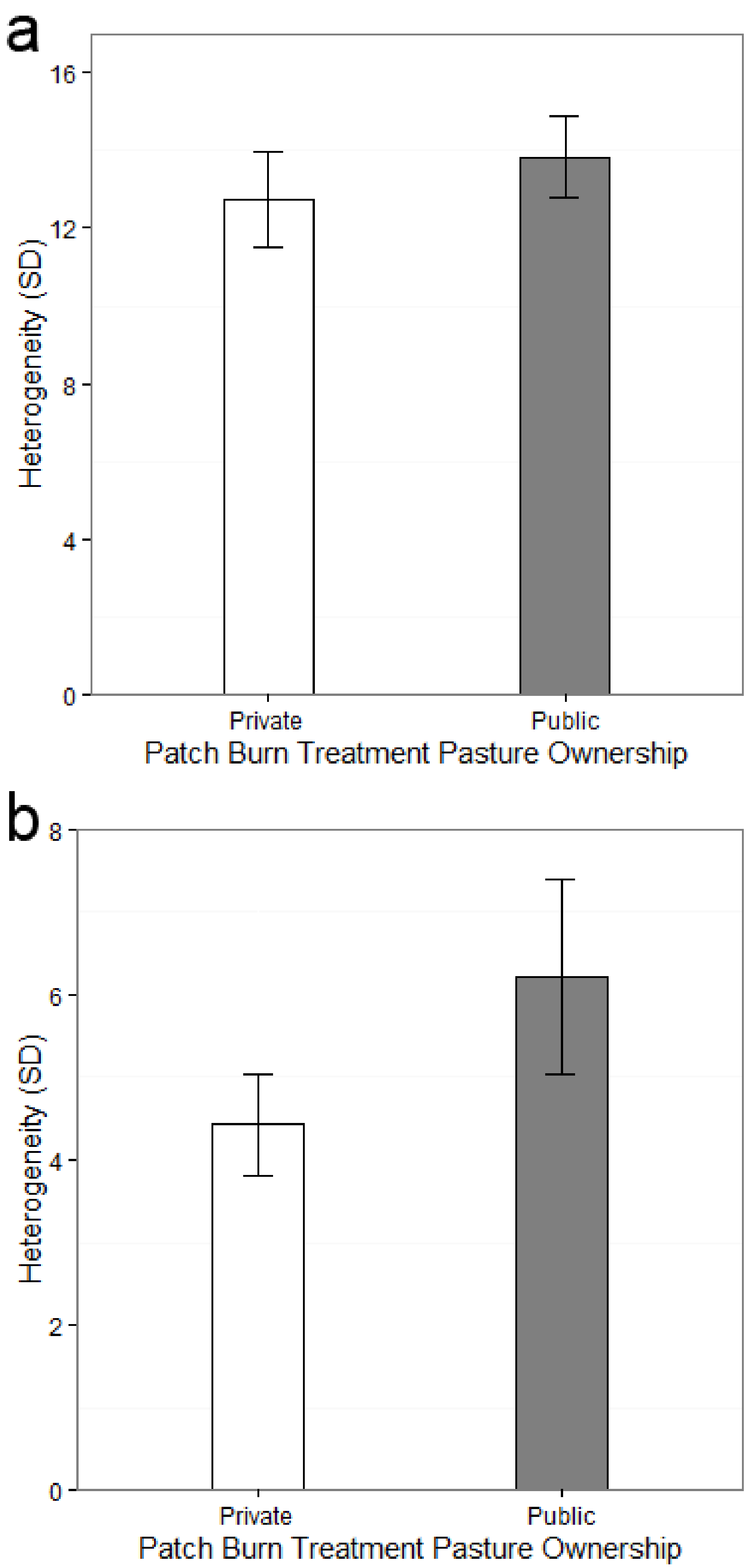


Figure 4. Mean ± SE of a) standard deviation of vegetation height separated by pasture ownership and year and b) standard deviation of vegetation visual obstruction separated by

pasture ownership and year. Dark grey bars represent privately owned pastures, and white bars represent publicly owned pastures. No within year differences in ownership were found to be significant.


a
16
12
8
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Heterogeneity (SD)
Private
Public
Patch Burn Treatment Pasture Ownership
b
8
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Heterogeneity (SD)
Private
Public
Patch Burn Treatment Pasture Ownership

Figure 5. Mean ± SE of a) standard deviation of vegetation height of patch burn treatment pastures separated by pasture ownership and b) standard deviation of vegetation visual obstruction separated by pasture ownership. Dark grey bars represent privately owned pastures, and white bars represent publicly owned pastures. No differences between pasture ownership were found to be significant.

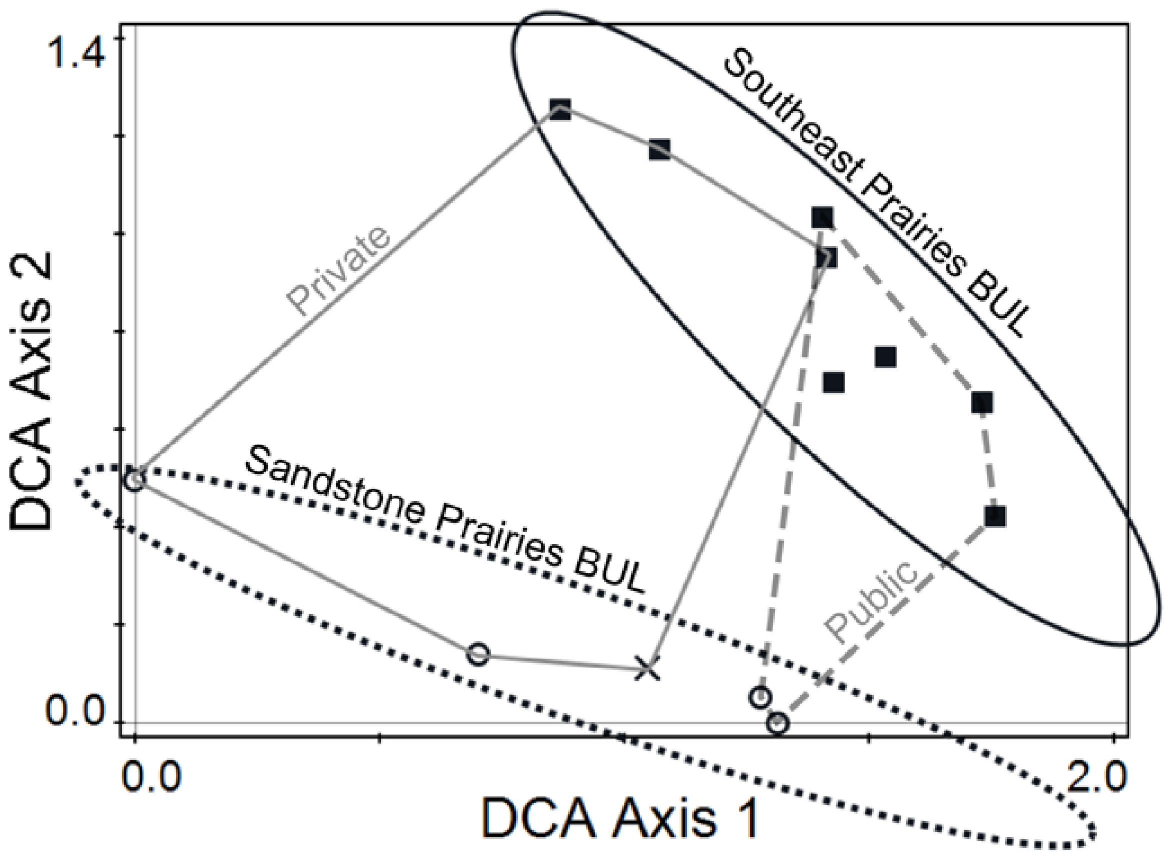

Figure 6. Detrended correspondence analysis of study sites based on plant community composition. Black envelopes (solid and dotted) represent Biologically Unique Landscapes. Grey envelopes (solid and dashed) represent pasture ownership. Solid squares represent study sites located in the Southeast Prairies Biologically Unique Landscape. Open circles represent study

sites located in the Sandstone Prairies Biologically Unique Landscape. Letter “X” represents a study site that was not physically located in either of the two Biologically Unique Landscapes.

CHAPTER II

SELECT BUTTERFLY HOST PLANT RESPONSE TO FIRE AND CATTLE GRAZING

Abstract

The interaction of fire and grazing animals has been a dominant disturbance in grasslands of the Great Plains for thousands of years. Many native plant and animal species are adapted and often require periodic disturbance to maintain viable populations. Violets (*Viola spp.)* and milkweeds (*Asclepias spp.)* are the host plants of the regal fritillary (*Speyeria idalia*) and monarch (*Danaus plexippus*) butterfly, respectively. Studies suggest these two butterfly species' populations are in decline, and loss of host plants has been suggested as an influential factor. We examine the response of these host plants to two different fire-grazing treatments, as well as time since fire, in southeastern Nebraska pastures. We did not observe a significant difference between treatments on the occurrence of these two host plants. We also did not observe a significant difference with time since fire on these two host plants. Our results are inconclusive and suggest a longer study period may be needed to observe significant differences.

Introduction

Fire and grazing are important disturbances in the Great Plains, and drivers of landscape biological diversity and heterogeneity (Fuhlendorf and Engle 2001, Anderson 2006). Rangelands in the Great Plains have undergone many changes since European settlement. These changes include the homogenization of vegetation structure, decline of fire as a disturbance, and habitat fragmentation (Coppedge et al. 2001a, Fuhlendorf and Engle 2001, Bailey 2004). Together these changes have negatively impacted wildlife populations (Herkert 1994, Siemann 1998, Coppedge et al. 2001b). Patch burn grazing is a rangeland management technique with the objective of creating spatio-temporal vegetation heterogeneity (Fuhlendorf and Engle 2004). Vegetation heterogeneity is generated through the use of patch burning and the attraction of grazing animals to regrowing vegetation on recently burned patches, known as pyric-herbivory (Fuhlendorf et al. 2009, Allred et al. 2011). Several studies have been conducted on individual plant species response to fire and grazing interaction in the Great Plains (Viton and Hartnett 1992, Helzer and Steuter 2005, Winter et al. 2011). However, only a few studies have examined the effects of fire and grazing management on *Viola spp.* and *Asclepias spp.*, which could be important to the conservation of butterflies (Debinski and Kelly 1998, Baum and Sharber 2012).

Viola spp., the host plant for larval regal fritillary butterflies (*Speyeria idalia* Drury), are perennial forb species found throughout the Great Plains. In several Midwestern states the regal fritillary is considered a species of concern, and sensitive to fire (Powell et al. 2007, Moranz et al. 2014, NatureServe 2014). *Viola spp.* growth and reproductive capacity may be affected by spring fires that are used to remove litter and improve forage for grazing livestock (Lovell et al. 1983). Several studies on *Viola pedatifida* G. Don, a common violet found in native prairies, have found no conclusive evidence that fire alone affects violet populations (White 1983, Tester 1996). In southern Wisconsin, *Viola pedatifida* was found to decrease in leaf area and flower production when subjected to late spring burns (mid-May) (Lovell et al. 1983). In contrast, this same study showed an increased number of leaves and fruits per flowering individual following both fall and early spring burns (late March to early April)(Lovell et al. 1983). Finally, a study from western Iowa found *Viola pedatifida* abundances to be highest in pastures that had experienced both fire and grazing (Debinski and Kelly 1998). These studies suggest that, depending on burn timing, fire may or may not impact *Viola spp.* populations, which could affect regal fritillary populations.

Asclepias spp. are host plants for larval monarch butterflies (*Danaus plexippus* L.), and are also perennial forbs found in the Great Plains. Monarch population declines have been observed in their wintering grounds of Mexico for the past decade (Brower et al. 2012). Many potential reasons are given for decline, but loss of milkweed habitat is cited as an important factor (Brower et al. 2012). The loss of milkweed habitat has been largely attributed to the loss of native prairie and the increased use of herbicides, and has been found to be correlated with monarch population decline (Pleasants and Oberhauser 2013). Due to the dependence of monarchs on milkweed host plants, the effects of managing native rangelands with fire and grazing on populations of *Asclepias spp.* is an important consideration. Evidence suggests that fire is an important disturbance for milkweed populations (Johnson and Knapp 1995, Baum and Sharber 2012).

Since regal fritillary and monarch larvae are dependent on violets and milkweeds, respectively, the impact of fire and grazing pressure on these host plant genera are of notable interest (Shepherd and Debinski 2005, Baum and Sharber 2012). *Viola spp.* and regal fritillaries as well as *Asclepias spp*. and monarchs have evolved in the pyric-herbivory driven Great Plains, and both should be adapted to a landscape mosaic induced by fire and grazing pressures. Our objective was to determine the influence of fire and grazing management on violets and milkweeds of southeastern Nebraska. Our study pastures were paired and each received a different treatment of fire and grazing. One treatment was considered traditional for the area and received an entire pasture burn the initial year of the study. Patch burn treatment pastures received a 1/3 pasture burn each year of the study. Since these host plant species are native to this region and likely adapted to fire and grazing, we expected little difference at the treatment level. Our treatments, however, might show a difference with time since fire, since the difference between treatment pastures was a spatio-temporal application of fire.

Methods

Study Sites

Fifteen study sites were located from in southeast Nebraska in the counties of Gage, Jefferson, Pawnee, and Johnson in 2009-2011. Pastures were typical for the region and ranged in size from a maximum of 67.6 ha to a minimum of 28.3 ha with an average of 41.3 ha. Eight pasture units were treated with patch burn management, where a different 1/3 of the pasture was burned each year of the study. The remaining seven pastures were burned in their entirety the first year of the study and considered traditional management pastures. All prescribed burns were conducted in the spring season for all pastures. At the completion of the study in 2011, all pastures had been burned in their entirety with the only difference associated with the spatio-temporal pattern of the fires. Together, seven of the traditional treatments, and seven of the patch-burn treatment pastures were considered to be paired, based on proximity, leaving a single, unpaired patch-burn treatment pasture. Eight pastures were privately owned while seven were owned by the Nebraska Game and Parks Commission. Dominant grass species include *Andropogon gerardii* Vitman, *Schizachyrium scoparium* (Michx.) Nash, *Sorghastrum nutans* (L.) Nash., *Bromus inermis* Leyss., and *Poa pratensis* L. Pastures were comparably stocked across both treatments primarily with cow/calf cattle herds. In 2009, one patch burn treatment and one traditional treatment pasture were stocked with yearlings and a mixed herd. Stocking rates varied slightly between pastures (2.22 ± 0.08 AUM/ha) and were determined using NRCS recommended stocking rates. In 2009, cattle turn-in dates were between April 1st and April 15th. In 2010 and 2011, cattle turn-in dates were between May 1st and May 15th. Cattle take-out dates for all years were between October 1st and October 15th. Annual mean temperatures for the region in 2009, 2010, and 2011 were 9.7°C, 10.5°C, and 10.7°C (www.ncdc.noaa.gov). Annual precipitation totals for the region in 2009, 2010, and 2011 were 665 mm, 833 mm, and 709 mm (www.ncdc.noaa.gov). Soils in the study sites ranged between silt loam, loam, and clay loam.

Plant Surveys

Plant species composition surveys were conducted on 13 treatment pastures (6 traditional and 7 patch burn) all three years except for two treatments that were sampled only twice. One sampled patch burn treatment pasture was excluded from butterfly host plant analysis due to milkweed occurrence being an extreme outlier compared to other pastures. Part of the reason this pasture's milkweed occurrence was extreme is due to only six transects being used and those occurred mostly within in a single burned patch (see appendix paired site 5). Surveys were conducted between June 12th and August 11th in all three years. Twelve, 100 m transects were used per pasture, except one traditional treatment pasture, where only six transects were used to avoid areas that showed evidence of historically tilled soils. Transects were established in 2009 using aerial imagery and were placed evenly across each study site to reside in suitable vegetation. Patch burn treatments contained four transects per treatment patch, except for one pasture which contains two. Each transect was established using ESRI's ArcGIS (ESRI 2013). Three waypoints, start, midpoint, and end, for each transect were uploaded to GPS units that were used in the field to determine transect locations. A 1 m^2 PVC frame was used to estimate percent coverage for each species found within the frame. The frame was placed along each transect at 10 m intervals for a total of 10 samples per transect. Coverage estimates for each species were estimated at 5% intervals from 0% to 100%, with 5% being the smallest possible value for a species present within the frame. Coverage of *Viola spp.* and *Asclepias spp.* rarely reached above 5% and for analysis were treated as presence or absence data for analysis. Two species of *Viola spp*. occurred in the same plot on two occasions across all three years and were treated as a single presence. Also, two species of *Asclepias spp.* occurred in the same plot on two occasions across all three years and were also treated as a single presence.

Analysis

Violet and milkweed occurrence in plots was averaged between treatment pastures for comparison at the treatment level. Welch's two-sample t-test was used to compare mean occurrence in plots between treatments. Since time since fire was different at the patch level, host plant occurrence in plots was averaged across patches for time since fire comparisons. Since our study was conducted on both private and public pastures, we also used Welch's two-sample t-test to examine differences between ownership on these two genera. To determine if time since fire had an influence on mean occurrence in plots, ANOVA followed by Tukey's honestly significant difference was used. We also examined genera occurrence in plots between years, since the traditional treatment pastures received a full pasture burn the initial year of the study. All analyses were conducted using R statistical software version 3.0.3 (R Development Core Team 2014).

Results

In total, 1148 occurrences of *Viola spp.* were found in 3960 total plots across all three years. The most common violet species observed was *Viola pedatifida* with 1096 plot occurrences. The only other violet species that occurred was *Viola sororia* Willd. which occurred in 52 plots. 189 *Asclepias spp.* occurrences were found in 3960 total plots across all three years. The most common milkweed species was *Asclepias verticillata* L. with 136 plot occurrences. Other milkweed species observed include *Asclepias viridis* Walter (n=22), *Asclepias viridiflora* Raf. (n=8), *Asclepias stenophylla* A. Gray (n=6), *Asclepias sullivantii* Engelm. Ex A. Gray (n=1), *Asclepias syriaca* L. (n=10), and *Asclepias tuberosa* L. (n=6).

We found no significant difference between mean occurrence of *Viola spp.* in plots of patch burn treatment and traditional treatment pastures (t = -0.93, df = 10.16, p = 0.37). We also did not find a significant difference between mean occurrence of *Asclepias spp.* in plots of patch burn treatment and traditional treatment pastures (t = 0.42, df = 9.93, p = 068). No significant difference was found in *Viola spp.* presence in plots between privately and publicly owned pastures (t = -1.51, df = 8.98, p = 0.17). Also, we did not find a significant difference of *Asclepias spp.* presence in plots between publicly and privately owned pastures (t = 0.24, df = 6.82, p = 0.82).

When examining *Viola spp.* presence in plots by time since fire, we observed no significant difference with time since fire in either treatment pastures at α = 0.05 level (Fig. 1a). We also compared percent occurrence in plots by year, since the traditional treatment pastures were burned in their entirety the first year of the study. We also did not find a significant difference between years of treatments for presence of *Viola spp.* presence in plots at α = 0.05 level (Fig 1b). Examining *Asclepias spp.*, we did not find a significant difference in presence in plots between both time since fire between patch burn and traditional treatment pastures at α = 0.05 level (Fig. 2a). We also did not find a significant difference in *Asclepias spp.* presence in plots across years in patch burn and traditional treatment pastures at α = 0.05 level (Fig. b).

Discussion

The interaction of fire and grazing has been shown to influence plant species in the Great Plains (Viton and Hartnett 1992, Helzer and Steuter 2005, Winter et al. 2013). However, we did not observe significant differences in plot presence in either *Viola spp.* or *Asclepias spp.* at the treatment level in our study. We did observe differences with changing time since fire and across years, although these differences were not significant.

There are several potential reasons our study did not detect differences for either examined genera. One possible explanation is that the population of violets and milkweeds have a longer response time to the disturbances of fire and grazing than the three-years of our study. Another possibility is that these two genera are well adapted to fire and grazing, since both have existed in the Great Plains for thousands of years. Also, land use legacy could potentially affect these populations due to management such as heavy grazing or herbicide use. However, we did not observe differences in plot presence between public and private ownership. Finally, local topo-edaphic characteristics can potentially constrain populations of violet and milkweed species by limiting potential locations for development such as unsuitable soil conditions or extreme slopes.

Our study results are similar to other studies on *Viola spp.* wherein fire was not found to influence the occurrence of violets (White 1983, Tester 1996). Other studies have shown potential benefits of fire to violet populations (Lovell et al. 1983). Further, it has been suggested that the absence of fire can lead to an increase in woodland vegetation, which may cause a reduction in the presence of *Viola spp.* (Krasewski and Waller 2008).

We found higher mean occurrence of *Asclepias spp.* in plots located in recently burned patches, but this result was not statistically significant. It’s possible that there really is a higher occurrence of milkweeds in the most recently burned areas, but that we did not collect enough data to pass the rigors of hypothesis testing. Another study located in the Flint Hills of Kansas did find significantly higher abundance of *Asclepias syriaca* in locations that received frequent fire (Johnson and Knapp 1995). Fire has also been suggested as an important disturbance for other species of milkweeds in tallgrass prairie (Kettle et al. 2000, Baum and Sharber 2012). This suggests that fire and grazing can benefit *Asclepias spp.* populations.

Since Regal Fritillary and Monarch populations are dependent on these genera as host plants, management implications on violets and milkweeds are of importance. We cannot accept the null hypothesis that fire and grazing have no influence on violet and milkweed populations. It may be due to the short duration of this study that no significant differences were observed. However, we can infer that lack of fire may lead to an increase in woodland vegetation, which could hinder populations of these genera (Krasewski and Waller 2008). Native violet and milkweed species have existed with the disturbance regimes of fire and grazing that existed in the historic Great Plains, and as a result should be tolerant to prescribed fires and moderate amounts of grazing. However, more evidence is needed to draw such conclusions.

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Figures

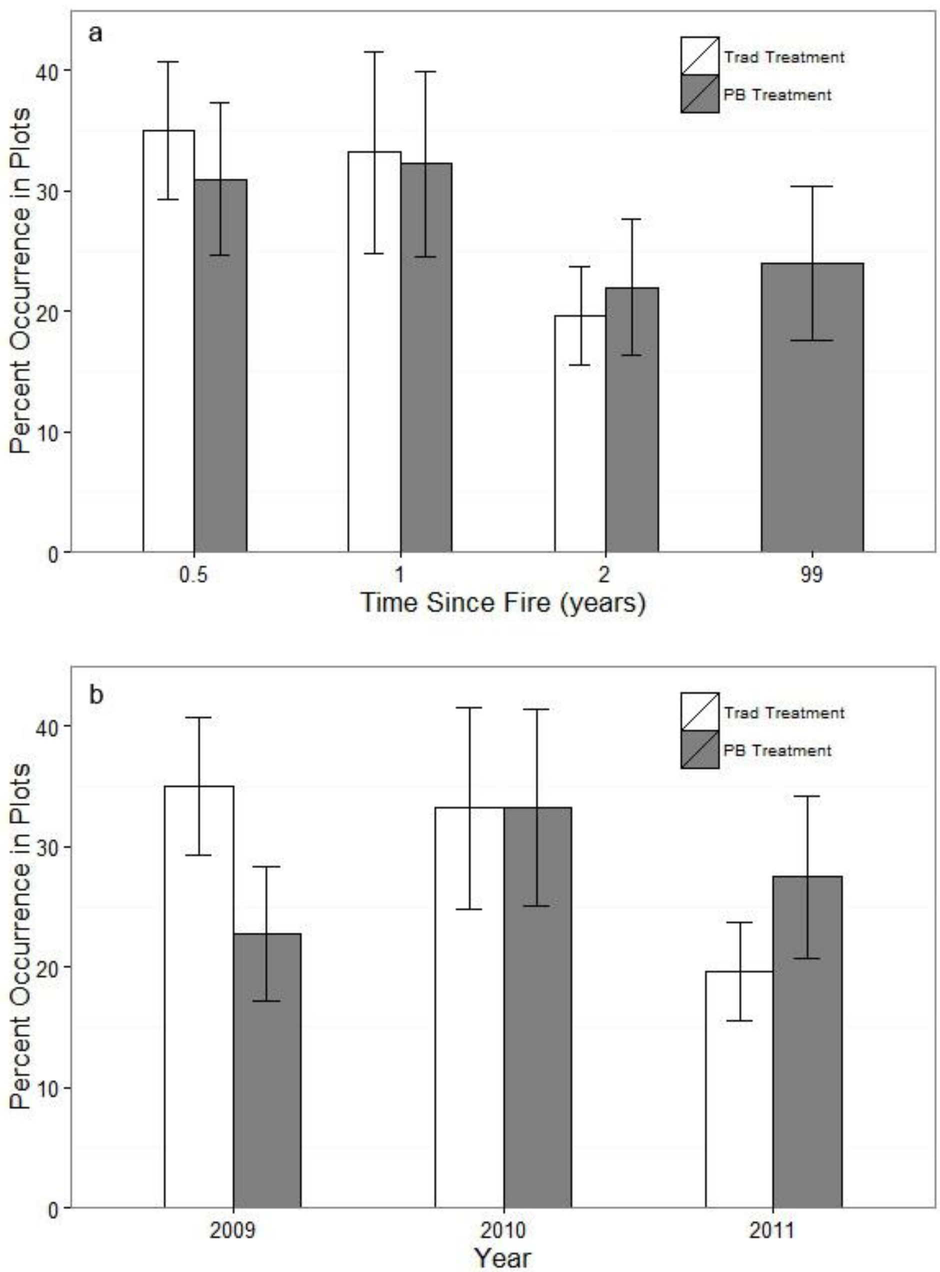

Figure 1. Differences in mean percent occurrence of *Viola spp.* in sample plots. a) Mean occurrence by time since fire and separated by treatment. b) Mean occurrence by year and separated by treatment. Grey bars represent patch burn treatment pastures. White bars represent traditional treatment pastures. Time since fire year 99 represents unburned portions of patch burn treatment pastures, which occurred in 2009 and 2010 only. Values are mean ± SE. No significant differences were observed.

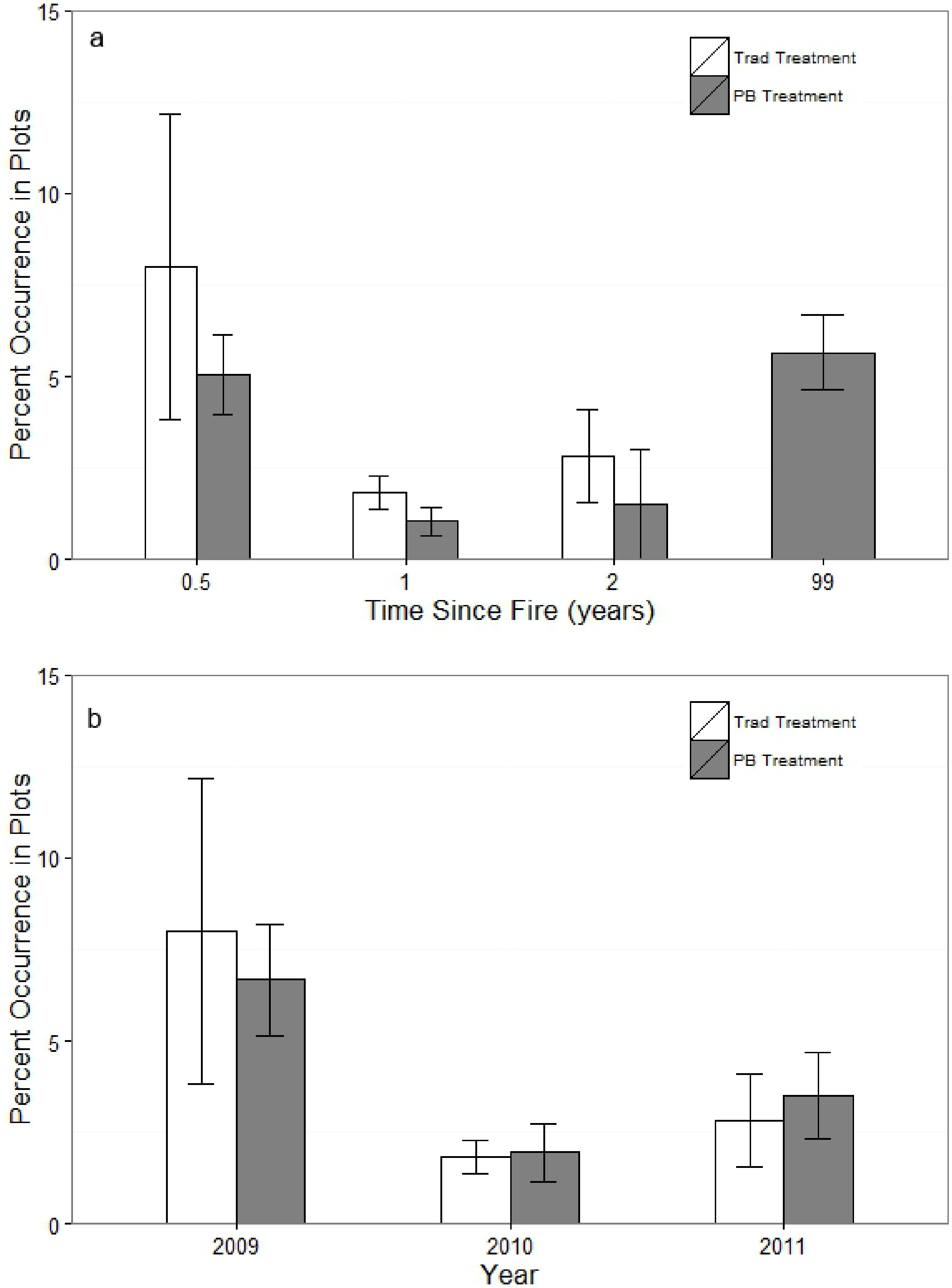

Figure 2. Differences in mean percent occurrence of *Asclepias spp.* in sample plots. a) Mean occurrence by time since fire and separated by treatment. b) Mean occurrence by year and separated by treatment. White bars represent traditional treatment pastures. Time since fire

year 99 represents unburned portions of patch burn treatment pastures, which occurred in 2009 and 2010 only. Values are mean ± SE. No significant differences were observed.

CHAPTER III

RANGELAND MANAGEMENT AND LANDSCAPE WOODLAND VEGETATION: INFLUENCES ON OBLIGATE GRASSLAND BIRDS

Abstract

Woodland vegetation has been increasing in rangelands around the world. In the Great Plains of the United States, the increase in woodland vegetation, in conjunction with the conversion of grasslands to row crop agriculture, has been blamed for the decline in obligate grassland bird species. We examine a rangeland management technique in southeastern Nebraska pastures has been previously shown to benefit grassland birds. We further examine the influence of woodland landscape vegetation at multiple scales and different woodland patch configurations on Grasshopper Sparrow, Dickcissel, and Eastern Meadowlark abundances, and how this woodland vegetation may confound management objectives. We compared average abundance of these three grassland birds to different management treatments. We examined mean distance to nearest woodland edge, percent total woodland and woodland perimeter to area ratio at three distances, 200, 400, and 800 meters, and determined influence of these variables on obligate grassland birds. We did not observe a significant treatment difference in mean abundance for any of the grassland birds. We found Grasshopper Sparrow abundance to be most influenced by woodland perimeter to area ratio (degree of edge) within the 200 m buffer. Dickcissel abundance was associated with mean distance to nearest woodland edge.

Finally, Eastern Meadowlark abundance responded most to percent total woodland within the 800 m buffer. Each of these grassland birds responded to different scale and configuration of woodland landscape vegetation, which suggests that conservation efforts should focus on maintaining large areas of open grassland with minimal woodland vegetation.

Introduction

An increase in woodland vegetation, such as trees and shrubs, has been occurring in grasslands around the world (Briggs et al. 2005). The encroachment of woodland vegetation can cause habitat fragmentation and changes in suitable habitat in North American grasslands that can affect wildlife populations (Horncastle et al. 2005, Matlack et al. 2008, Alford et al. 2012). Many reasons for the increase in woodland vegetation have been suggested (Archer et al. 1995). However, the decline of historic fire regimes is often highlighted as one of the most significant reasons (Briske et al. 2006). Fire is known to hinder woody plant encroachment (Briggs et al. 2002), and the use of fire and grazing to promote temporal and spatial heterogeneity has been shown to create beneficial habitat structure for many obligate grassland birds (Fuhlendorf 2006, Coppedge et al. 2008, Powell 2008). However, landscapes fragmented by woodland vegetation at broader scales than individual pastures may override the potential benefits of heterogeneous pasture management on obligate grassland birds (Hovick et al. 2012). Because of this, examining landscape woodland vegetation influence on obligate grassland birds can help prioritize and refine strategies for grassland bird conservation.

In the Great Plains, obligate grassland birds have seen dramatic population declines (Heckert 1995). Since the mid-1960s, the obligate grassland bird guild has declined more than any other breeding bird guild in the Midwest (Heckert 1995). A large part of this decline has been attributed to the increase in woodland vegetation, which impacts grassland birds both directly through displacement and indirectly through habitat alteration that promotes predation or nest parasitism (Winter and Faaborg 1999, Coppedge et al. 2001). Supporting the relationship that the decline of grassland birds is in part due to woody plant encroachment, densities of some grassland birds have been found to increase as non-linear woodland features decreased in patch size, and as distance to woodland vegetation increased (Ribic and Sample 2001). Examining the impact of landscape vegetation on obligate grassland birds requires multi-scale analysis, since vegetation patterns can have many spheres of influence existing at several spatial and temporal scales (Wiens et al. 1987, Fuhlendorf et al. 2002, Winter et al. 2006). Grassland bird densities are influenced by habitat patch size, landscape, and regional and local vegetation structure (Chapman et al. 2004, Winter et al. 2006). Other studies have shown that grassland bird nest success is reduced as nest distance to forest edge increases due to increased nest predation and nest parasitism (Gates and Gysel 1978, Winter et al. 2000). While several studies have been conducted looking at total percent woodland cover on grassland birds, studies looking at the effect of woodland edge on grassland birds have also supported the hypothesis that as the perimeter to area ratio (PAR) of woodland edge decreases, grassland birds increase in richness and density (Helzer and Jelinski 1999, Robels 2010).

The obligate grassland birds, Grasshopper sparrow (*Ammodramus savannarum*), Dickcissel (*Spiza americana*), and Eastern Meadowlark (*Sturnella magna*), have all been identified as being area sensitive (Ribic et al. 2009). Higher abundances of these three grassland birds were found in areas with high percentage of grassland and low percentage of forests in southwestern Wisconsin (Murray et al. 2008). However, these three species are generalists in terms of grassland specific structural requirements (Bakker et al. 2002, Fuhlendorf et al. 2006). This makes these three species ideal candidates for studying the influence of woodland vegetation patterns on area sensitive obligate grassland bird species. Further, obligate grassland birds can potentially serve as indicators for changes in woodland landscape vegetation, and may have population responses similar to other grassland dependent wildlife (Swengel and Swengel 1999, Browder et al. 2002).

Our objective is to determine whether the benefits of heterogeneous pasture management for obligate grassland birds are constrained by woodland landscape vegetation at different scales and contexts. We predict treatment effect will have little influence on grassland bird abundances in our study. We expect Grasshopper Sparrow, Dickcissel, and Eastern Meadowlark abundances to change with total woodland vegetation and with woodland perimeter to area ratio. We also hypothesize that these obligate grassland birds will exhibit increased abundances as mean distance from nearest woodland edge increases.

Methods

Study Sites

Fifteen study sites were located in southeast Nebraska in the counties of Gage, Jefferson, Pawnee, and Johnson in 2009-2011. Pastures were typical for the region and ranged in size from a maximum of 67.6 ha to a minimum of 28.3 ha with an average of 41.3 ha. Eight pasture units were treated with patch burn management, where a different 1/3 of the pasture was burned each year of the study. The remaining seven pastures were burned in their entirety the first year of the study and considered traditional management pastures. All prescribed burns were conducted in the spring season for all pastures. At the completion of the study in 2011, all pastures had been burned in their entirety with the only difference associated with the spatio-temporal pattern of the fires. Together, seven of the traditional treatments, and seven of the patch-burn treatment pastures were considered to be paired, based on proximity, leaving a single, unpaired patch-burn treatment pasture. Eight pastures were privately owned while seven were owned by the Nebraska Game and Parks Commission. Dominant grass species include

Andropogon gerardii Vitman, *Schizachyrium scoparium* (Michx.) Nash, *Sorghastrum nutans* (L.) Nash., *Bromus inermis* Leyss., and *Poa pratensis* L. Pastures were comparably stocked across both treatments primarily with cow/calf cattle herds. In 2009, one patch burn treatment and one traditional treatment pasture were stocked with yearlings and a mixed herd. Stocking rates varied slightly between pastures (2.22 ± 0.08 AUM/ha) and were determined using NRCS recommended stocking rates. In 2009, cattle turn-in dates were between April 1st and April 15th. In 2010 and 2011, cattle turn-in dates were between May 1st and May 15th. Cattle take-out dates for all years were between October 1st and October 15th. Annual mean temperatures for the region in 2009, 2010, and 2011 were 9.7°C, 10.5°C, and 10.7°C (www.ncdc.noaa.gov). Annual precipitation totals for the region in 2009, 2010, and 2011 were 665 mm, 833 mm, and 709 mm (www.ncdc.noaa.gov). Soils in the study sites ranged between silt loam, loam, and clay loam.

Bird Sampling

Fourteen pastures were used to conduct grassland bird surveys, seven patch burn treatment and seven traditional treatment pastures. Bird surveys were conducted between June 15 and July 1 in 2010 and 2011, and each pasture was surveyed two times per year for a combined total of four surveys per pasture. The sampling protocol was conducted with winds less than 25km/hr, no rain, and cloud cover less than 75%. Surveys were conducted using a single point count location per pasture, where birds were identified by sight or sound within a 400 m radius, during a 10 minute sample period. This distance was chosen to allow for a separate analysis examining Northern Bobwhite quail. Birds in flight, and birds flying through the survey area, were not counted unless they landed in or flushed from within the survey area. Individual bird locations were marked on data sheets that contained an aerial image of the site and were recorded at 25 m distance intervals. These individual bird locations were entered into a GIS, analyzed, and compared with land classification data. A comprehensive avian community analysis can be found in Winter et al. (2013).

Land classification was developed from aerial photography obtained through the Farm Service Administration's National Agriculture Imagery Program (NAIP). The images were downloaded from the United States' Department of Agriculture's Geospatial Data Gateway. The images were from July 27, 2010 and have a spatial resolution of 1 m x 1 m. These images were loaded into ESRI's ArcGIS version 10.1, cropped to a more manageable size, and merged into a mosaic image. Five land cover classification types were identified and delineated: grassland, woodland, cropland, water, and developed. The grassland classification included pastures, CRP, grass right-of-ways, and hay meadows. Woodland classification included areas of contiguous woodland vegetation as well as individual woody plants large enough to visualize from the imagery, typically greater than 9 m^2. Cropland classification included all row crop fields. The water classification included streams, man-made lakes, and ponds. Developed classification included all gravel and paved surfaces. A supervised classification was performed using ArcGIS to separate pixels into their respective land cover classification. The classified raster image was then converted into a classified polygon layer using ArcGIS. Classified polygons were visually checked for errors and misclassified polygon areas were corrected by manually reassigning the proper class number to the given polygon or modifying the polygon to accurately represent the classification type. Finally, the classified polygon layer was reconverted to raster data for analysis. An accuracy assessment was conducted using ArcGIS to randomly choose 250 points within the classified image and then compare the classified image with the NAIP image (Congalton 1991). An error matrix was created to compare observed versus expected pixel classifications. From this error matrix, Cohen's kappa statistic was calculated to estimate overall accuracy; resulting in a kappa of 0.945 (Congalton 1991). Within the Eastern Meadowlark's 800 m landscape buffer, the land cover across all pastures was found to be 70.76% grass cover, 13.04% agricultural cover, 11.46% wooded cover, 4.12% water cover, and 0.62% developed cover.

Analysis

Only a subset of the total birds identified were used for analysis; these being birds within the grassland bird guild such as Grasshopper Sparrow, Dickcissel, and Eastern Meadowlark, which are all obligate grassland species. Three variables of landscape composition were analyzed for each bird species: percent woodland area, woodland perimeter to area ratio (PAR), and distance to nearest woodland edge. Woodland area and woodland PAR were analyzed within 200, 400, and 800 meter radii buffers. Buffers were generated for each species using all individuals of that species within each pasture (Fig. 1). Buffers were chosen using individuals rather than the original point count to capture the landscape metrics based on the location of individual birds themselves. The radii distances were chosen to be consistent with established studies, though with different methodology (Ribic and Sample 2001, Cunningham and Johnson 2006), as well as examine landscape features within and beyond the pasture boundaries. Statistics from the classified raster image were extracted for each species' buffer distances using Geospatial Modeling Environment version 0.7.2.1 and Fragstats version 4.1. Woodland PAR was analyzed by measuring the total length of edge and dividing by the total woodland area using the 200, 400, and 800 meter radii from each individual within each bird species. Finally, distance to nearest woodland edge was analyzed by measuring the distance between each recorded individual to the nearest area of woodland classification in ArcGIS.

Multiple regression using model selection was used to determine which variables best explain the variation in the data with the greatest parsimony. Regression was used to model each species' mean abundance to percent woodland, woodland PAR, distance to nearest woodland edge, and subsequent interactions for each pasture. Average abundance of Eastern Meadowlarks was log transformed to meet the assumptions of normality. To parse out scales of vegetation influence, percent woodland and woodland PAR covariates were restricted to each buffer extent, 200, 400, and 800 meters, for each model. Since our pasture sample size was less than 40, second-order Akaike information criterion (AIC_c) was used to select a best model from candidate models for each bird species (Burnham and Anderson 2002). Top models with Δ AIC_c < 3 containing multiple explanatory variables were analyzed for collinearity between covariates by calculating the variance inflation factor (VIF) and determining the correlation. Models found to have high collinearity were excluded from results. Models determined to include pretending variables were also excluded from each bird species top models (Anderson 2008). We also examined parameter estimates from the AIC_c determined best model. Relative variable importance for each species was determined from all models having an Akaike weight greater than 0.01 (Burnham and Anderson 2002). All analyses were conducted using R statistical software version 3.0.3 (R Development Core Team 2014).

Results

Our results suggest that each examined grassland bird species responded differently to woodland landscape patterns and scales. Mean abundance of Grasshopper Sparrows (n = 156) was most influenced by woodland PAR within the 200 m buffer area (r^2 = 0.21)(Table 1, Fig. 2). All other models had a ΔAIC_c of greater than 2.6. The second best model, with a ΔAIC_c of 2.6, contained only the variable woodland PAR within the 800 m buffer. Parameter estimates of this top model indicate that woodland PAR within the 200 m buffer had a negative influence on mean Grasshopper Sparrow abundances with a coefficient of -1.69 (Table 2). Based on all models with an Akaike weight of >0.01, woodland PAR within the 200 m buffer is suggested as the most influential variable with a weight of 0.51, followed by percent woodland within the 200 m buffer with a weight of 0.24 (Table 3).

The AIC_c best model for mean Dickcissel (n = 193) abundance included the variable of mean Dickcissel distance to nearest woodland edge with an (r^2 0.32)(Table 1, Fig. 3). The AIC_c second best model for mean Dickcissel abundance included total percent woodland within the 400 m buffer with a ΔAIC_c of 2.1. Other models had a ΔAIC_c of greater than 2.8. The parameter estimates for the top model show that mean distance to nearest woodland edge had a positive influence on mean Dickcissel abundance with a coefficient of 0.01 (Table 2). Mean distance to nearest woodland edge was also found to be the most important variable with a weight of 0.57, followed by total woodland within the 400 m buffer with a weight of 0.25 (Table 3).

Mean abundance of Eastern Meadowlarks (n = 98) was significantly influenced by total percent woodland within the 800 m buffer area according to the AIC_c best model with an (r^2 = 0.37)(Table 1, Fig. 4). The second best model, with a ΔAIC_c of 1.8, suggested that total percent woodland within the 400 m buffer as a probable model. Other models had a ΔAIC_c of greater than 2.5. The best model showed a negative relationship between mean abundance and percent woodland within the 800 m buffer with a coefficient of -0.05 (Table 2). The most influential variable for mean abundance of Eastern Meadowlarks was total percent woodland within the 800 m buffer with a weight of 0.46, and the second most important variable was mean Eastern Meadowlark distance to nearest woodland edge with a weight of 0.31 (Table 3).

We did not observe a significant difference in mean bird abundances between patch burn treatment and traditional treatment pastures for Grasshopper Sparrow (t = 0.98, df = 11.74, p = 0.34), Dickcissel (t = -0.70, df = 9.57, p = 0.50), or Eastern Meadowlark (t = -0.14, df = 11.86, p = 0.89). We also did not see a difference in mean abundance between public and private pasture ownership for Grasshopper Sparrow (t = 0.58, df = 11.15, p = 0.58) and Eastern

Meadowlark (t = -0.55, df = 11.39, p = 0.59). However, we did see a significantly higher mean abundance of Dickcissels in publicly owned pastures compared to privately owned pastures (t= -2.30, df = 11.99, p = 0.04).

Discussion

The increase of woodland vegetation in the Great Plains has the potential to affect objectives and outcomes of rangeland management practices that have shown benefits for wildlife species. Our results suggest that woodland vegetation has important constraining effects on the importance of local habitat conditions for obligate grassland birds. Further, it suggests that different grassland bird species respond differently to woodland vegetation landscape patterns and scales. We did not observe a treatment effect across management styles.

It is common to describe grassland birds as area sensitive and several studies have highlighted their dependence on landscape pattern, but few have compared local and landscape level patterns. High woodland perimeter to area ratio was related to lower abundances of Grasshopper Sparrows in central Nebraska (Helzer and Jelinski 1999). Also, in southwestern Wisconsin similar results were found wherein lower Eastern Meadowlark abundances were associated with areas of high proportion woodland (Murray et al. 2008). Several studies examining distance to nearest woodland edge have been conducted on grassland birds that include Dickcissels (Winter and Faaborg 1999, Bakker et al. 2002, Coppedge et al. 2008). However, in our study, distance to nearest woodland edge was strongly related to mean Dickcissel abundance.

Patch burn grazing, a pasture management technique that incorporates prescribed patch burns, has shown benefits for obligate grassland birds through the establishment of vegetation structural heterogeneity (Fuhlendorf 2006, Churchwell et al. 2008, Coppedge et al.

2008). Several reasons could explain our study's inability to detect differences in treatment effect. First, our sampling methodology may not be rigorous enough to evaluate differences in grassland bird abundances between treatments. Second, since these obligate grassland birds are typically generalists in terms of vegetation structure, they may not respond to the differences in pasture management. Rather, fire and grazing applied in pastures to create a shifting mosaic of vegetation structure may provide more benefit to obligate grassland bird species that specialize in extremes of grassland vegetation structure. Evidence suggests that this second reason is certainly plausible (Fuhlendorf et al. 2006, Jacobs et al. 2012). Lastly, the influence of woodland vegetation in this study may constrain the differences in pasture treatments. Patch heterogeneity that exists within the patch burn treatments may have unfavorable landscape woodland vegetation for these grassland birds.

Since a treatment effect was not observed, it is likely that core habitat and fragmentation are more influential factors on abundances of these three species (Heckert 1994, Winter and Faaborg 1999, Renfrew and Ribic 2007). Landscape context can also influence the abundance of obligate grassland birds by influencing habitat fragmentation (Coppedge et al. 2001, Fletcher and Koford 2002). Our results suggest that the scale and configuration of woodland landscape vegetation can also influence the abundances of Grasshopper Sparrows, Dickcissels, and Eastern Meadowlarks. However, each bird was associated with a different woodland vegetation variable, which suggests that these grassland birds respond to landscape configurations differently. Grasshopper Sparrows had higher mean abundances when patches of woodland vegetation had lower amounts of edge compare to woodland area at a relatively close extent. This suggests woodland features such as tree rows might significantly fragment Grasshopper Sparrow habitat. Dickcissels responded greatest to mean distance to nearest woodland edge, which could potentially be a single tree within a large core of grassland habitat.

Further, many studies do not distinguish between types of woodland edge (Johnson 2001, Winter et al. 2006). In reality woodland edge can be anywhere within a gradient from the perimeter of a single tree to the edge of a dense stand of woodland (Porensky and Young 2013). Eastern Meadowlarks responded to total woodland vegetation at the largest scale examined, suggesting they select for core grassland habitat with less woodland vegetation in the greater landscape.

The implication of these results can confound management techniques that have been shown to be beneficial to grassland bird species. Management for obligate grassland birds in small prairies or pastures may be hindered by the vegetation or land use of the greater landscape. In contrast, priority areas for grassland bird conservation should take place in locations that have large areas of grassland dominated landscapes. Scale must be considered when developing management strategies for wildlife that may have many spheres of influence. Each of the generalist, obligate grassland bird we examined responded differently to scale and configuration of woodland landscape vegetation. These grassland birds preferred grassland landscapes that have few trees and large amounts of grassland.

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Tables

Table 1. Models of mean abundance of obligate grassland birds to landscape woodland vegetation in southeastern Nebraska, USA. Models shown are for ΔAICc < 3.

Model	*K*	ΔAICc
Grasshopper Sparrow		
Perimeter to area ratio within 200m	1	0.0
Perimeter to area ratio within 800m	1	2.6
Perimeter to area ratio within 400m	1	2.8
Percent total woodland within 200m	1	2.8
Percent total woodland within 400m	1	2.8
Dickcissel		
Distance to nearest woodland edge	1	0.0
Percent total woodland within 400m	1	2.1
Percent total woodland within 200m	1	2.1
Percent total woodland within 400m + Perimeter to area ratio within 400m	2	2.8
Eastern Meadowlark		
Percent total woodland within 800m	1	0.0
Percent total woodland within 400m	1	1.8
Percent total woodland within 800m + Distance to nearest woodland edge	2	2.5
Perimeter to area ratio within 800m	1	2.7

Table 2. Coefficient values of AICc selected best models.

Variables	Coefficient	Standard Error	t-value	p-value
Grasshopper Sparrow				
Intercept	3.07	0.23	13.35	<0.001
Perimeter to area ratio within 200m	-1.69	0.94	-1.80	0.10
Dickcissel				

Intercept	2.60	0.40	6.48	<0.001
Distance to nearest woodland edge	0.01	0.004	2.40	0.03
Eastern Meadowlark				
Intercept	1.04	0.26	3.99	0.002
Percent total woodland within 800m	-0.05	0.02	-2.66	0.02

Table 3. Relative variable importance based on Akaike weights for obligate grassland birds.

Variable	w_i (AICc)
Grasshopper Sparrow	
Perimeter to area ratio within 200m	0.51
Percent total woodland within 200m	0.24
Distance to nearest woodland edge	0.13
Dickcissel	
Distance to nearest woodland edge	0.57
Percent woodland within 400m	0.25
Percent woodland within 200m	0.17
Eastern Meadowlark	
Percent woodland within 800m	0.46
Distance to nearest woodland edge	0.31
Percent woodland within 400m	0.22

Figures

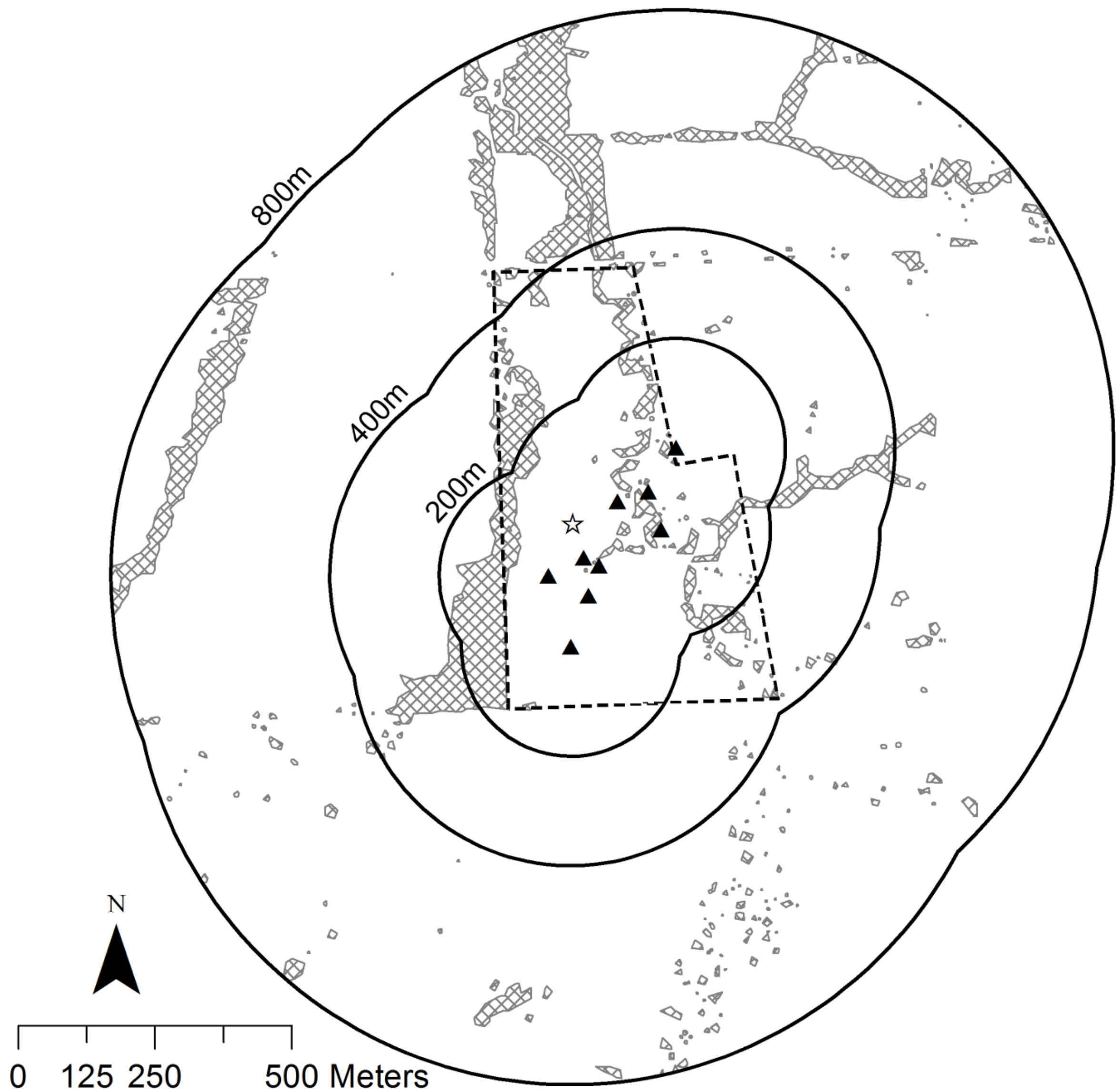


Figure 1. Example of buffer distances generated from individual Dickcissel locations. Solid triangles represent individual Dickcissels. Crosshatching represents woodland features. Hollow star represents point count location. Dashed line represents treatment pasture boundary.

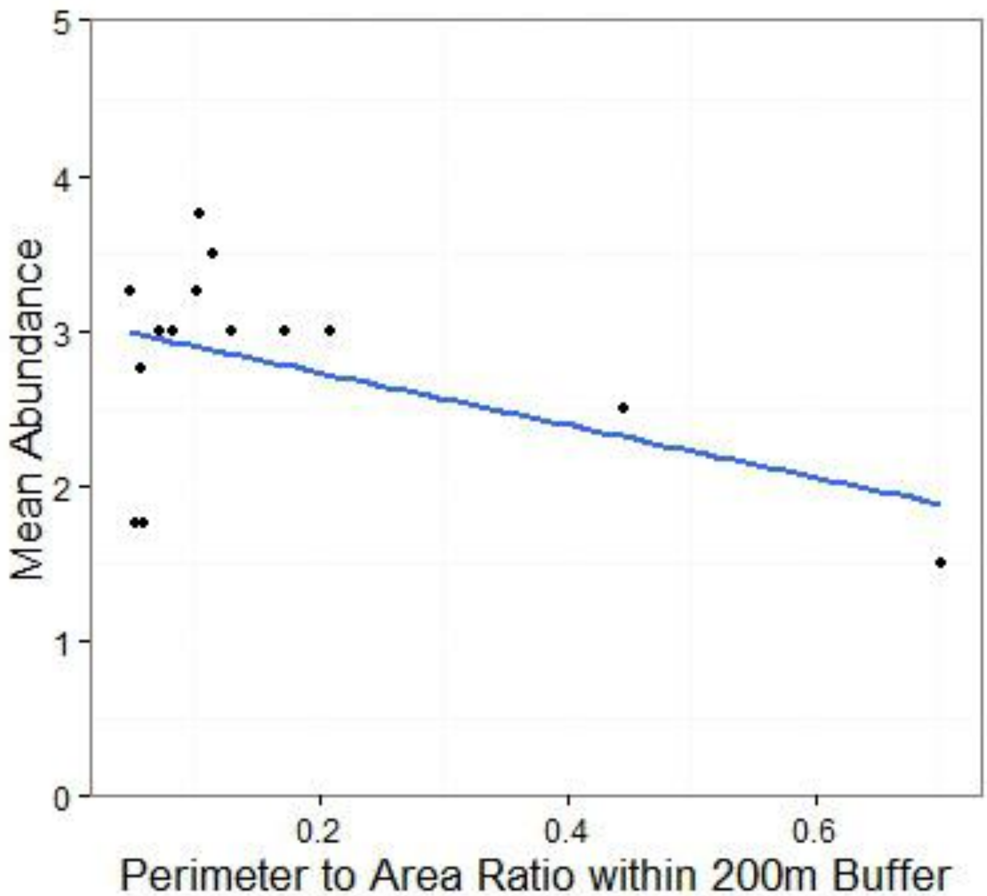


Figure 2. Grasshopper Sparrow mean abundance to perimeter to area ratio within 200m buffer based on AICc best model (r^2 = 0.21).

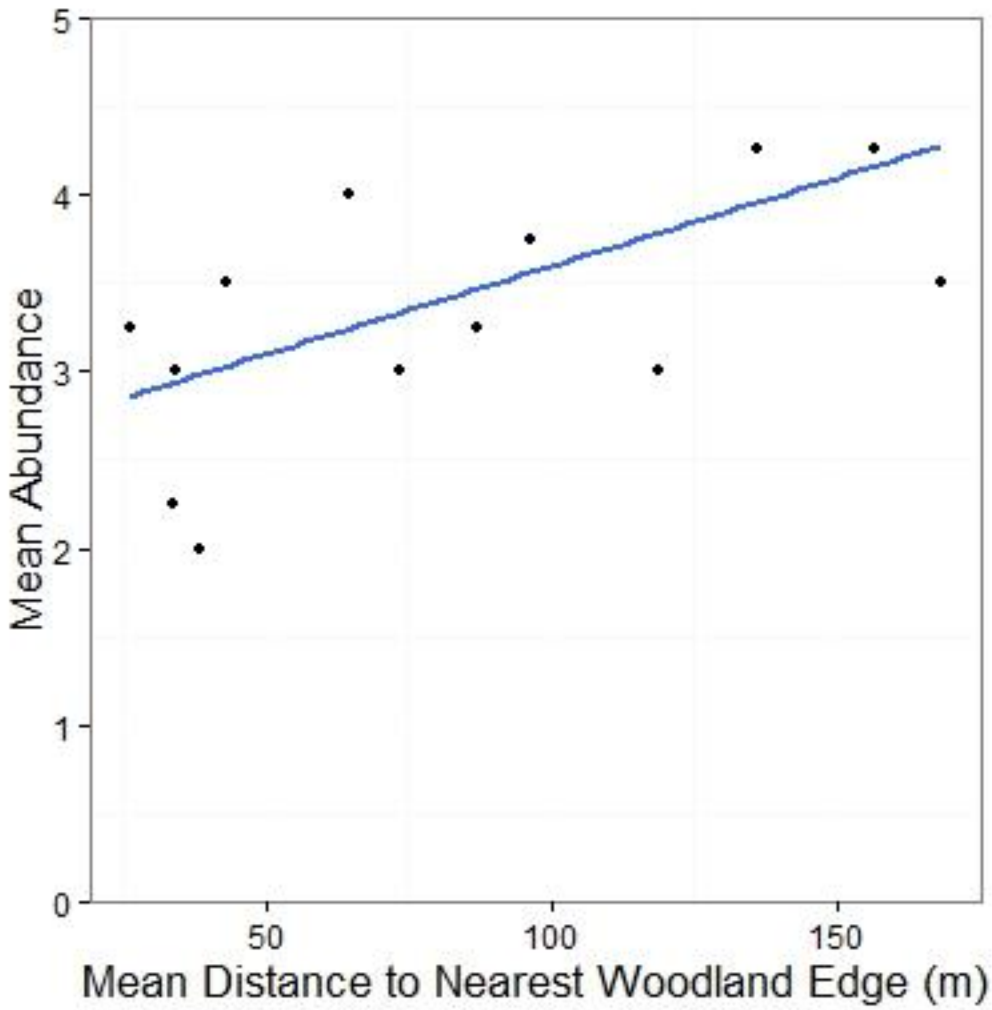


Figure 3. Dickcissel mean abundance to nearest woodland edge based on AICc best model (r^2 = 0.32).

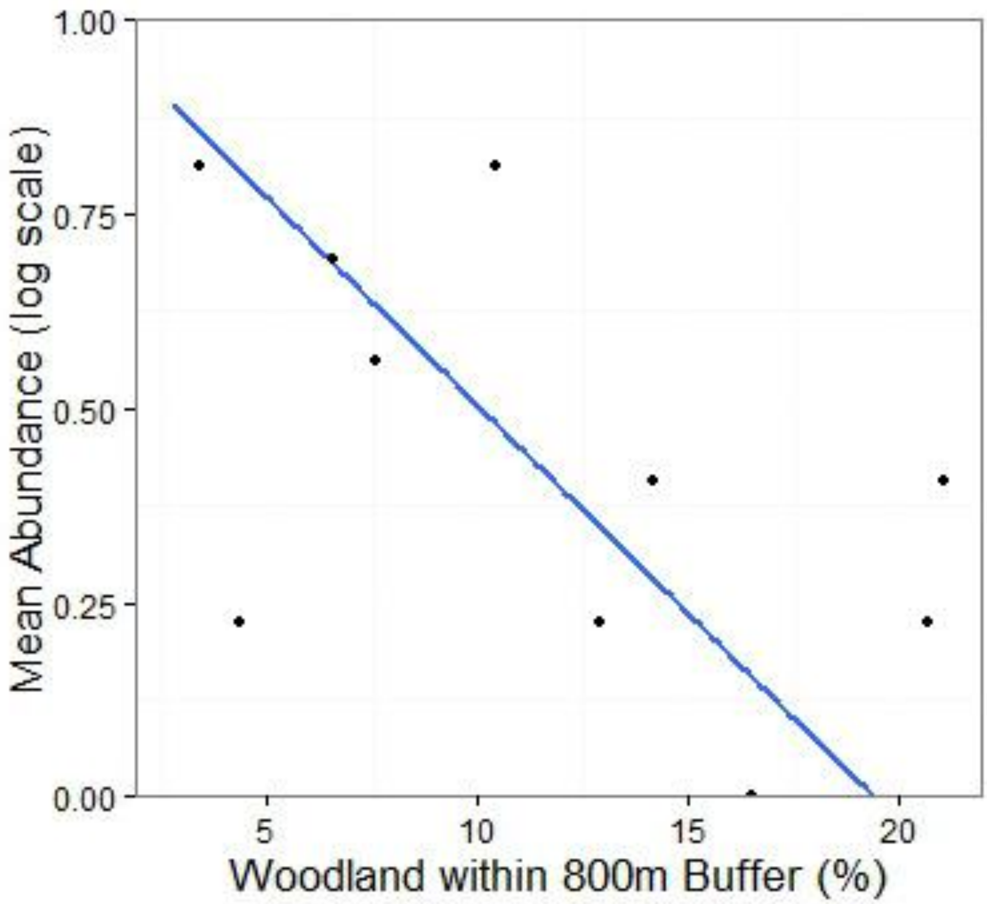
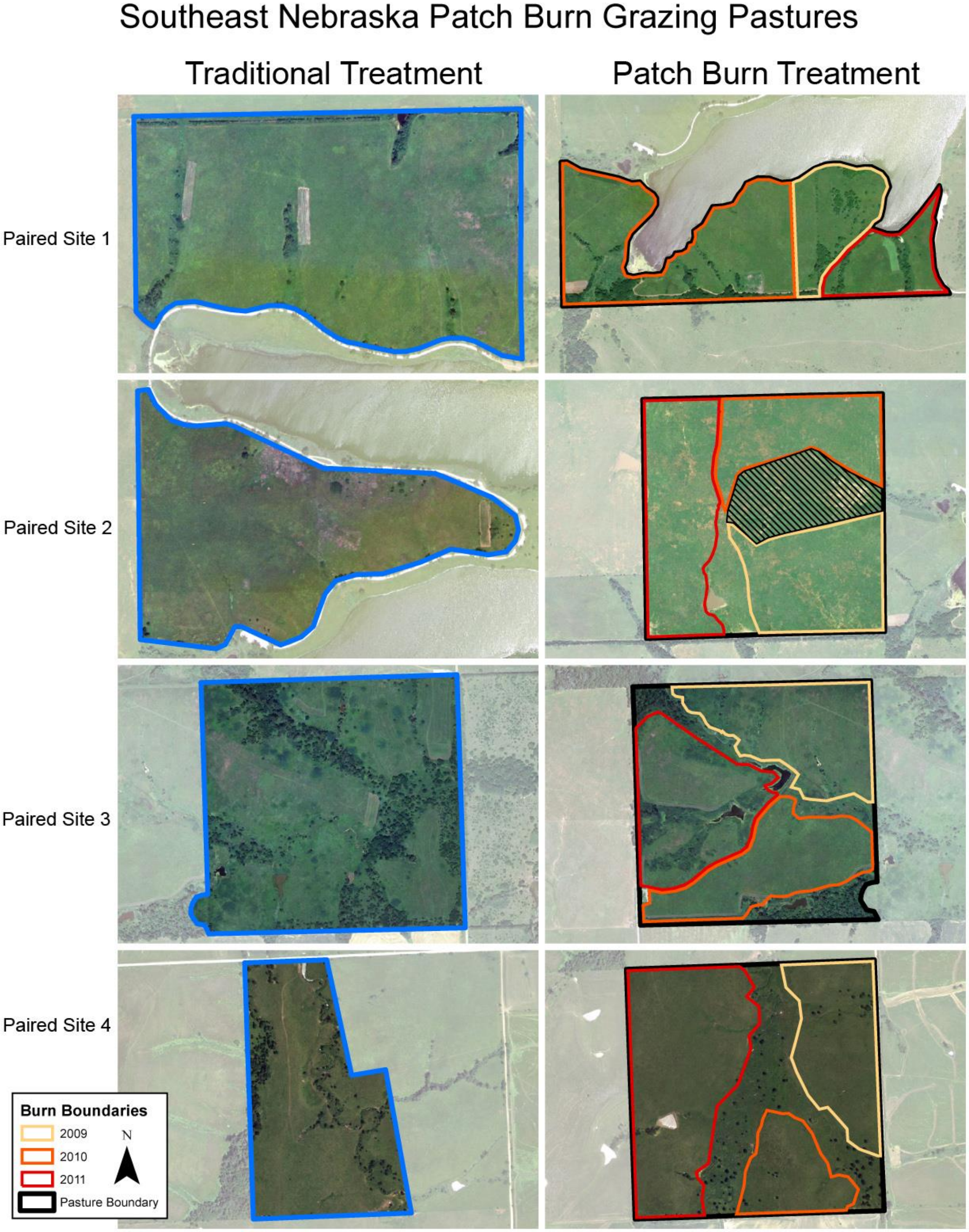


Figure 4. Eastern Meadowlark mean abundance on log scale to percent total woodland within 800m based on AICc best model (r^2 = 0.37).

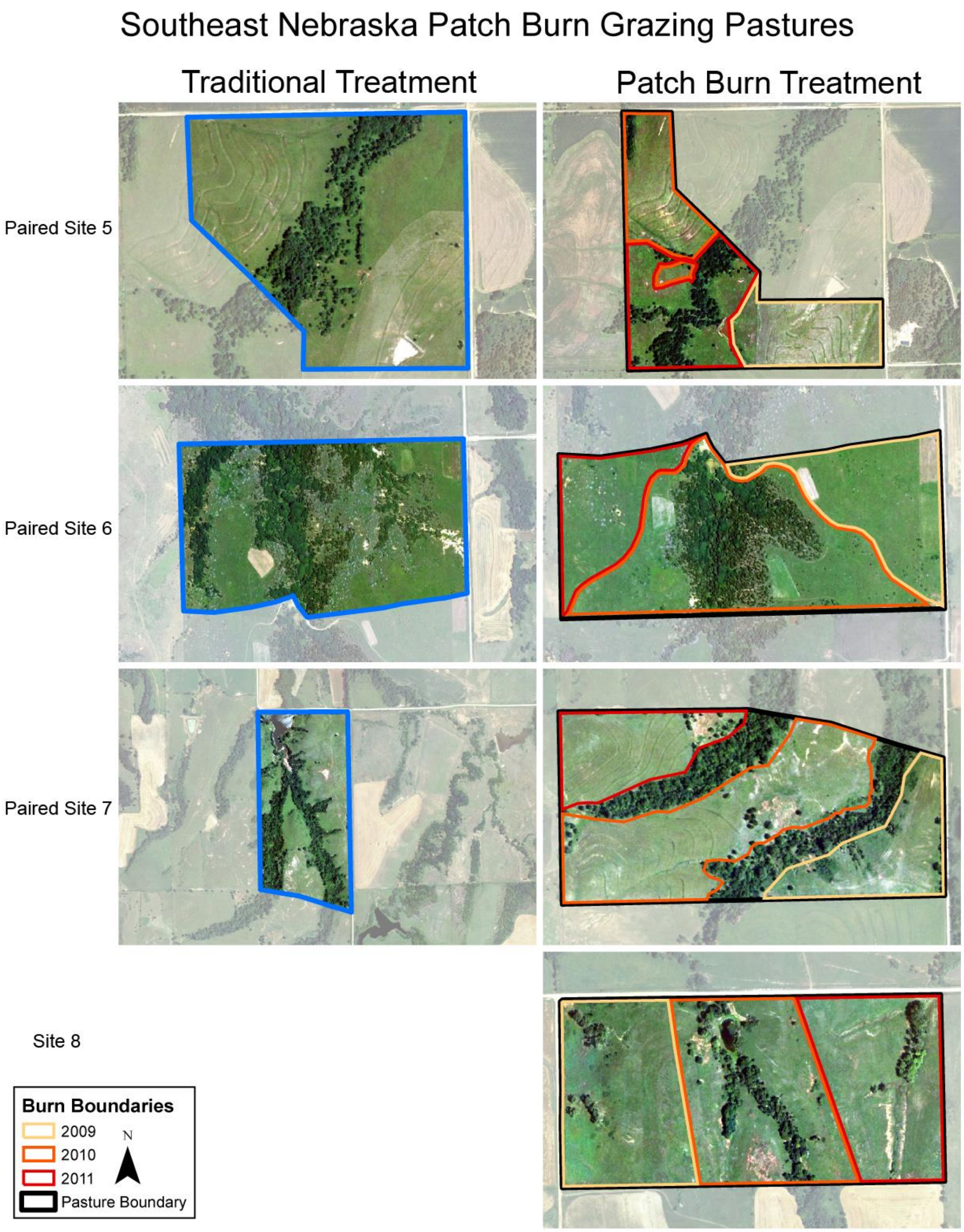
APPENDIX



Study pastures located in southeastern Nebraska. Study sites were paired based on proximity.

Different colors in the patch burn treatment pastures represent areas that received patch burns.

Traditional treatment pastures received an entire pasture burn the initial year of the study (2009).



Study pastures located in southeastern Nebraska. Study sites were paired based on proximity. Different colors in the patch burn treatment pastures represent areas that received patch burns. Traditional treatment pastures received an entire pasture burn the initial year of the study (2009).

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