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RESEARCH ARTICLE

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Effects of Patch-burn Grazing on Vegetative Composition of Tallgrass Prairie Remnants in Missouri

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ABSTRACT: Patch-burn grazing is a grassland management approach that recouples the processes of fire and grazing altering animal distribution and creating structural heterogeneity for the benefit of flora and fauna biodiversity. Our objective was to investigate the effects of patch-burn grazing on plant species composition in tallgrass prairie remnants on conservation lands in Missouri. Each grazed unit consisted of three patches with each patch approximately one-third the size of the grazed unit. We burned a different patch annually and stocked the entire unit at a moderate rate of 0.45 AUM/ha with yearling stocker steers. We established paired plots of treatment (grazed) and control (grazing excluded) and sampled vegetative composition during a 3-yr graze-burn cycle and a fourth year of no treatments. Species richness, diversity, and floristic quality of exclosures were compared between plots using the Shannon, Simpson, and Floristic Quality indices. Our results indicated a year × treatment interaction. Specifically, species richness and floristic quality were significantly reduced in grazed plots during the first growing season post-burn ($P < 0.05$). However, after the first growing season post-burning and focal grazing, grazed plots no longer differed in diversity or floristic quality from ungrazed plots. Species richness remained elevated in grazed plots beyond the first growing season after burning and focal grazing. Species diversity showed no significant change in grazed plots throughout the study ($P < 0.05$) but results indicated a similar trend, declining during the first growing season post-burn followed by a spike the second year and recovery in subsequent years. More research is necessary to understand effects of long-term implementation of patch-burn grazing and on individual species. Despite the results of a companion study on horizontal structure, our results suggest that after one year of intense grazing following fire, any changes in vegetative composition were minor to nonexistent after three years of patch-burn grazing and one year of rest

Index terms: diversity, floristic quality, patch-burn grazing, pyric herbivory, tallgrass prairies

INTRODUCTION

Conventional prairie management on conservation lands has included periodic prescribed fire and mowing to remove litter, maintain or improve plant species diversity, and prevent woody encroachment. While these practices may effectively achieve those goals, they unfortunately create uniform height and density of vegetation (Fuhlendorf et al. 2012) and can promote dominance of perennial C_4 grasses (Kucera and Koelling 1964; Collins 1987). Prescribed burning is essential to managing grasslands to restrict woody plant growth, release nutrients bound in plant litter, and maintain plant species diversity (Steinauer and Collins 1996). However, research has indicated that fire alone is not a sufficient solution for restoring or maintaining prairie biodiversity (Knopf and Samson 1997; Collins et al. 1998).

Prior to European settlement, interaction between fire and grazing by native herbivores created a mosaic of heterogeneous vegetative structure (Fuhlendorf and Engle 2001). Diversity of flora and fauna has been shown to increase in grasslands following reintroduction of grazers in a manner that mimics the presettlement interaction of fire and grazing (Hartnett et al. 1996; Hayes and Holl 2003; Fuhlendorf et al. 2006; Engle et al. 2008; Burns et al. 2009). Pyric herbivory is a term ecologists have used to describe the spatial and temporal interactions between grazing and fire that promote a shifting pattern of disturbance across the landscape (Fuhlendorf et al. 2009). Implementing this management paradigm involves rotationally burning patches of a management unit and allowing grazers free access to the entire unit. There are no interior fences and the higher quality forage of the most recently burned patch attracts the majority of grazing pressure (Coppedge et al. 1998; Vermeire et al. 2004; Collins and Smith 2006; Allred et al. 2011a; Leis et al. 2013). As different portions of the unit are burned in successive years, grazing pressure shifts spatially among patches. Grazing pressure is reduced as time since fire increases with only minimal grazing in patches that have not been burned in more than two to three years (Allred et al. 2011a). The practice is referred to as patch-burn grazing (PBG) and is an attempt to return ecological processes to the grassland system that have been disrupted since European settlement (Fuhlendorf et al. 2009; Fuhlendorf et al. 2012).

Managing grazing distribution of cattle has traditionally been accomplished by using

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fences and strategic placement of other resources such as watering points or mineral supplement (Owensby 1994; Derner et al. 2009; Holecheck 2011; Fuhlendorf et al. 2012). Most traditional approaches to grazing management have sought to reduce unequal forage utilization across a pasture to maintain forage production in areas that would be threatened by overgrazing (Owensby 1994; Holecheck 2011). Duvall and Whitaker (1964) used prescribed burning to enhance forage palatability of coarse perennial grasses and shift grazing pressure among patches within longleaf pine–bluestem rangeland in Louisiana. However, the technique was not widely adopted because burning only a portion of a grazed area was often viewed as counterproductive toward achieving the goal of even forage utilization throughout the unit (Vallentine 1989:186). Patch-burn grazing contrasts with traditional grazing management in that equal forage utilization is distributed across a longer period (e.g., three years) rather than within a single growing season. Recent studies have brought more attention to this management technique and highlighted benefits to co-occurring grassland bird, insect, plant, and small mammal species by achieving greater heterogeneity in habitat structure (Fuhlendorf and Engle 2004; Coppedge et al. 2008; McGranahan et al. 2012a; Moranz et al. 2012; Winter et al. 2014).

Missouri Department of Conservation (MDC) land managers were interested in implementing patch-burn grazing on tallgrass prairie remnants to diversify the structure of grassland cover to enhance habitat quality for grassland birds. Using bison to restore grazing to grassland systems is not reasonable in the majority of tallgrass prairie remnants due to expensive infrastructure and resources necessary in managing a herd. Many prairie remnants are small tracts (<300 ha) with public access. Safety concerns also prevent the use of bison in managing prairie remnants owned by public agencies. Therefore, land managers and ecologists in the Midwest and Great Plains increasingly rely on cattle to emulate natural processes on conservation lands (US Fish and Wildlife Service 2012). Cattle have been found to exhibit preference for recently burned prairie and can be used to create the shifting mosaic of vegetative structure similar to that created by bison (Hobbs et al. 1991; Fuhlendorf and Engle 2004; Vermeir et al. 2004; Allred et al. 2011a; Allred et al. 2011b). While studies have documented similarities in diet between cattle and bison (Hartnett et al. 1997; Towne et al. 2005), concerns remain about the potential for negative effects of cattle grazing on prairie communities. Research has demonstrated that improper grazing can reduce species richness and shift the floristic composition of tallgrass prairie remnants to favor dominance of exotic and early successional species (Drew 1947; Kucera 1956; Nyboer 1981).

At the inception of this study, anecdotal observations of the effects of patch-burn grazing on habitat structure suggested that it could be used to achieve management goals (McGranahan et al. 2013); however, no local data were available to support this assertion or objectively evaluate potentially important changes to the vegetation community. The objective of this 4-year study was to document the influence of PBG on the floristic quality, species diversity, and species richness of high-quality tallgrass prairie remnants within the Osage Plains Region of Missouri.

STUDY AREA

Our study was conducted at five prairie remnants in the Osage Plains Ecoregion of Missouri (Nigh and Schroeder 2002): Bethel Prairie Conservation Area (Barton County), Hi Lonesome Prairie Conservation Area (Benton County), Niawathe Prairie (Dade County), Taberville Prairie Conservation Area (St. Clair County), and Wah'Kon-Tah Conservation Area (Cedar County). These tallgrass prairie remnants are located in west-central Missouri and owned by MDC and The Nature Conservancy (Figure 1). Prairies within this region are unglaciated and characterized by well-drained soils of poor fertility. The shallow and rocky soil of this region has deemed much of the area unsuitable for cultivation, sparing many prairie remnants from conversion to cropland. Several small and scattered prairie remnants in this ecoregion have persisted post-settlement as hay or cattle pastures under light to moderate grazing pressure. Mean annual precipitation for the region is 99–107 cm (39–42 in) with a mean growing season length of 210 days (Nigh and Schroeder 2002).

The study sites are considered dry–mesic with sandstone and shale parent material. Chert and bedrock were often exposed. Dominant grasses of the community type included big bluestem (*Andropogon gerardii* Vitman), Indian grass (*Sorghastrum nutans* [L.] Nash), little bluestem (*Schizachyrium scoparium* [Michx.] Nash var. *scoparium*), prairie dropseed (*Sporobolus heterolepis*[A. Gray] A. Gray), and Scribner's panic grass (*Panicum oligosanthes* var. *scribnerianum* [Nash] Fernald) (Nelson 2005). Characteristic shrubs included lead plant (*Amorpha canescens* Pursh), pasture rose (*Rosa carolina* L.), and New Jersey tea (*Ceanothus americanus* L.) as well as a variety of long-lived perennial forb species endemic to tallgrass prairie remnants (Nelson 2005). While all sites had a management history that involved grazing, botanical surveys conducted prior to the study by MDC biologists revealed high plant species diversity that was thought to represent intact tallgrass prairie plant communities (Missouri Department of Conservation, unpub. data).

METHODS

Each site consisted of one grazing unit with infrastructure of exterior fencing and water sources. The size of each grazing unit ranged from 62 ha (154 ac) to 105 ha (261 ac) ($\bar{x}$ = 72 ha [179 ac]). We divided the grazing units into three patches that comprised approximately one-third of the total unit. We randomly assigned a 2005, 2006, or 2007 burn to patches within each unit with all patches receiving one burn. Burning was conducted between mid-March and mid-April using a ring fire technique by drip torch. Yearling cattle were stocked at a rate of 0.45 AUM/ha from mid-April to mid-August (120 days).

We monitored changes in vegetative composition in nine pairs of permanent plots per grazing unit with three paired plots randomly placed within each patch. Plots were arranged such that members of the pair shared slope and aspect, fell on approx-

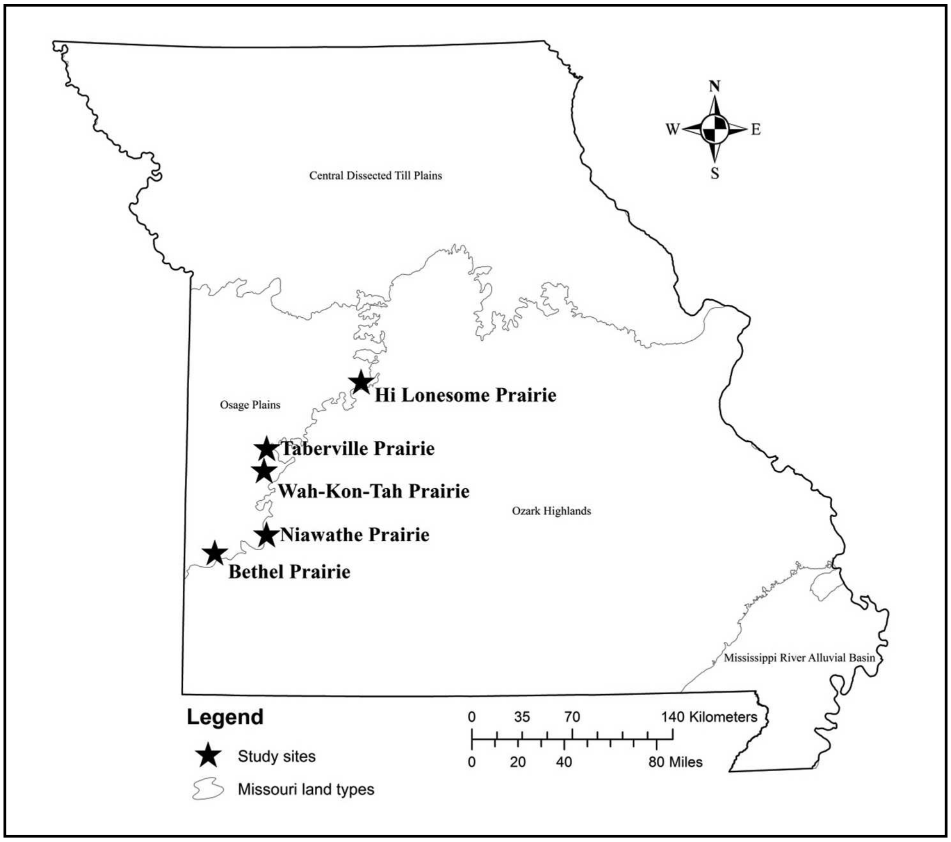

Figure 1. Study locations of five remnant prairies within the Osage Plains Region of Missouri.

imately the same contour, and appeared to encompass similar vegetation communities. One plot of each pair was randomly selected to serve as a control to isolate the effects of grazing. We determined which plot would serve as the control by the flip of a coin and constructed a grazing exclosure around that plot by using t-posts and woven wire fencing. A 1-m buffer was retained between the exclosure fence and the plot boundary. Plots within pairs were separated by a 15-m buffer to ensure that cattle paths that developed around exclosure fences did not influence vegetation in the unfenced treatment sites. Each 10 × 10 m plot employed a nested design of 1 x 1 m subplots at the four corners.

We estimated percent cover of all plant species within the subplot by using the Daubenmire Class Method (Table 1; Bonham 2013:141–143). We collected data during July 2005–2008. Therefore, the first three years of data collection occurred while cattle were present and the 4th and final year (2008) was considered a rest period with no grazing or burning.

Analysis

We assessed floristic quality using the Floristic Quality Index (FQI) on a plotwise basis. The FQI is an index weighted by a coefficient of conservatism value (CC) for each species within a sampling plot (Jog et al. 2006). Coefficient of conservatism values follow Ladd (2004). The values assigned to each species are based on the species' fidelity to a community type. Plants with greater fidelity to a community are assigned greater values and occupy a small ecological niche while plants with lower values are tolerant of many different conditions (Taft et al. 1997; Jog et al. 2006). The total number of native species recorded in a plot also plays an important role in the calculation. In the circumstance a plant could not be identified to species, particularly nonflowering individuals of the *Panicum* and *Carex* genera, we used the average CC value of species within the same genus and known to exist within the area as an adjusted CC value.

We assessed species diversity using the effective number of species derived from the Shannon entropy (H′) and Simpson concentration (D) indices (Jost 2006). The diversity indices were calculated for each subplot using cover class midpoints (Table 1). The index value and species richness/m² were determined as the mean value of the subplots. We also determined the richness/m² of native, exotic, and annual/biennial species for each plot as the mean value of the subplots. Species that could not be identified or could be identified only to genus, particularly the *Panicum* and *Carex* genera, were not included in the native, exotic, and annual/biennial species richness analysis.

We then calculated the difference within each pair between grazed and ungrazed plots so that a positive difference indicates greater value in grazed plots, while a negative difference indicates greater value in ungrazed plots. If the metric (FQI, species richness, H′, or D) of the grazed

Table 1. Cover classes and midpoints used to estimate plant species cover and calculate plant diversity indices at five prairie remnants within the Osage Plains Region of west-central Missouri from 2005 to 2008.

Cover class	% cover	Mid-point %
1	0.1–1	0.5
2	2–5	3.5
3	6–25	15.5
4	26–50	38
5	51–75	63
6	76–95	85.5
7	96–100	98

and ungrazed plots were similar, the difference was near 0. Since the grazed and ungrazed plots were specifically selected to minimize spatial variability in the various diversity measures, the plots could not be considered independent. Consequently, all analyses were performed on the difference (as described above) instead of analyzing two separate samples. We calculated 95% confidence intervals as the mean difference +/− [$t_{0.025,\,4}$ (SE)].

A repeated-measure, blocked, nested ANOVA was used to determine if differences between grazed and ungrazed plots varied significantly among the three patches during the four years of study (2005 to 2008). The five different prairie remnants acted as blocks since they all had different management histories. Finally, a blocked ANOVA was used to determine if differences in floristic quality, diversity, and total species richness was greater or less than 0 during the final year (2008) of our study (≥1 year of rest from burning and grazing for all patches). We used the statistical software R (R Core Team 2013) with the function lm and the Anova function from the car package for all repeated measures analyses.

It was important that our analysis accounted for variation among observers throughout the four years of our study. Our sampling teams remained constant within each year of sampling to minimize the variability known to occur among sampling teams (Kercher et al. 2003), but were different among sampling years. Therefore, analyzing the difference between grazed and ungrazed plots allowed us to minimize the variation within each sampling year but failed to reduce the variation among years due to the necessity of using different sampling teams each year.

RESULTS

All study sites indicated similar trends regarding the parameters analyzed. However, despite blocking our data to account for variation among locations, there was considerable variation in the FQI variable of the difference within pairs. The standard error among patches ranged from 0.24 to 0.61, while average difference values in FQI between grazed and ungrazed plots ranged from −1.39 to 0.81 (Figure 2). Average FQI values for grazed and ungrazed plots at each site are shown in Table 2.

Analysis of FQI showed an interaction of patch and year ($P < 0.001$). Floristic Quality Index values were less in grazed plots relative to ungrazed plots during the year of a burn (Figure 2). For example, in the 2005 burn patch, the mean difference value was lowest during the year that patch was burned, indicating FQI had decreased in the grazed plots. The declines in FQI of grazed plots appear to be a temporary effect as the difference between grazed and ungrazed plots was reduced by the second post-burn growing season. By 2008, all plots had experienced at least one year of rest from burning and grazing. No differences were detected in FQI ($P = 0.11$) between grazed and ungrazed plots as 95% CI overlapped 0 for all burn patches during the final year (2008) of our study (Figure 2).

A total of 361 species was encountered within the sampling plots across all blocks. The standard error in total species richness/m² ranged from 0.19 to 1.27, while average difference values among patches ranged from −0.48 to 2.75. Diversity measures were also considerably variable. The standard error of H' and D index values ranged from 0.29 and 0.36 to 2.25 and 2.21 with average difference values among patches ranging from −1.61 and −0.62 to 2.55 and 1.81, respectively. Mean site diversity indices and species richness for both grazed and ungrazed plots are shown in Table 2.

There was a significant interaction between patch and year for mean species richness and mean richness of native species ($P < 0.004$). We did not detect a difference in average number of annual/biennial species as few were found: 1.4/m² (1.24 SD) for a CV of 88% (Figure 3). We also did not find any significant factors or interactions for exotic species richness because the average number of exotic species found in study plots was very small (Table 2). Exotic species most often encountered included Canada bluegrass (*Poa compressa* L.), sericea lespedeza (*Lespedeza cuneata* [Dum. Cours.] G. Don), Korean clover (*Kummerowia stipulacea* [Maxim.] Makino), redtop (*Agrostis gigantea* Roth), and creeping bentgrass (*Agrostis stolonifera* L.). Differences in species richness among patches appear related to time

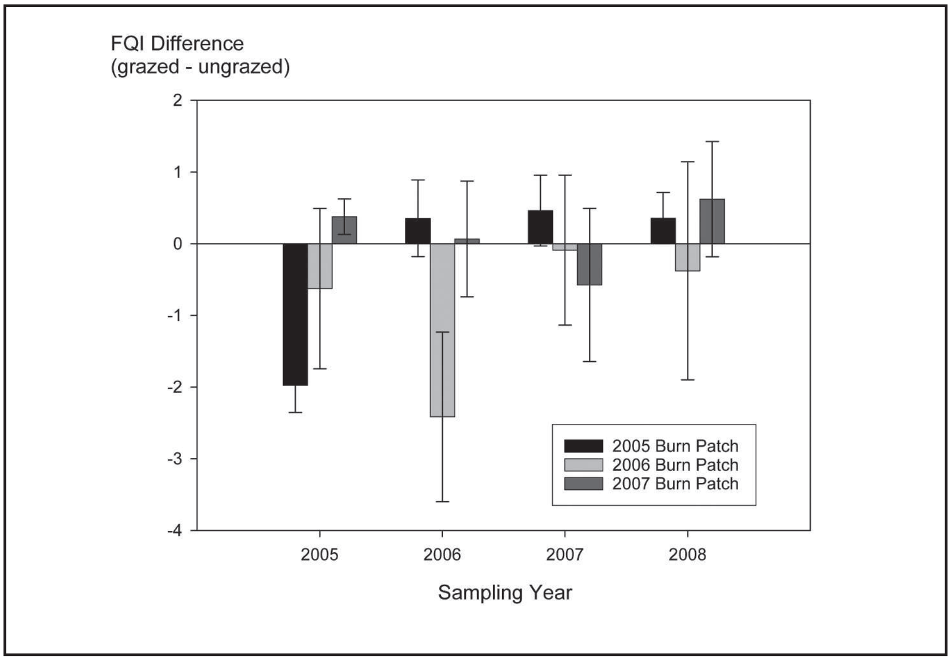

Figure 2. Mean differences in Floristic Quality Index values between sampling plots subjected to patch-burn grazing and ungrazed plots within exclosures during three years of patch-burn grazing followed by one year of rest in west-central Missouri tallgrass prairie. Error bars represent SE.

Table 2. Average index values for Floristic Quality (FQI), Shannon entropy (H′), Simpson concentration (D), and species richness of study subplots (1-m²) within grazed (G) and ungrazed (U) plots at five prairie remnants within the Osage Plains Region of west-central Missouri from 2005 to 2008.

Site	FQI		H′		D		Total species richness		Exotic species richness	
	G	U	G	U	G	U	G	U	G	U
Bethel	13.01	12.72	18.77	17.72	13.94	12.38	19.08	17.82	1.11	1.10
Hi Lonesome	12.02	12.10	19.46	17.58	13.91	12.55	20.10	18.13	1.42	1.25
Niawathe	15.95	16.20	21.73	20.18	15.74	14.53	21.28	19.49	1.11	1.19
Taberville	17.11	17.84	21.18	22.74	15.13	16.80	22.30	22.09	1.12	1.04
Wah'Kon-Tah	16.73	16.99	21.41	20.94	15.07	15.12	21.18	19.94	1.21	1.00

since fire. As with FQI, species richness immediately following fire was less in grazed plots (Figure 4). However, grazed plots had greater mean species richness than ungrazed plots 1–3 years post-burn. Interaction between patch and year was not found for H′ (*P* = 0.31) or D (*P* = 0.40) although data showed a pattern similar to total species richness (Figure 4). The blocked ANOVA analysis of the final year (2008) of our study showed no differences in total species richness (*P* = 0.26), H′ (*P* = 0.78), or D (*P* = 0.88) between grazed and ungrazed plots (Figure 5).

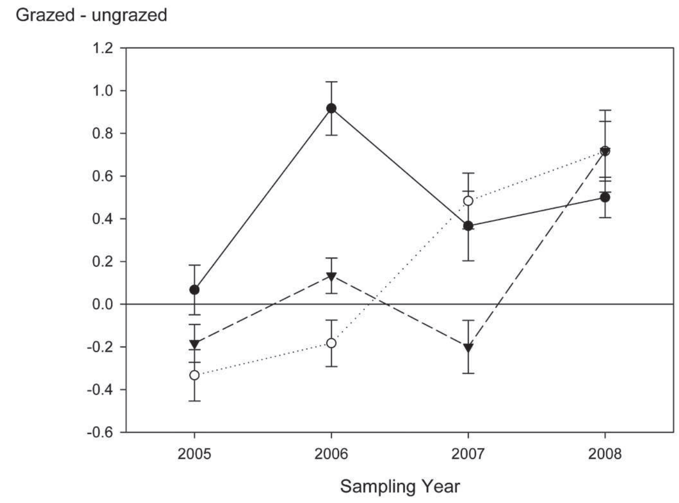

Figure 3. Mean differences between sampling plots subjected to patch-burn grazing and ungrazed plots within exclosures regarding plant richness of annual and biennial species in patches burned in 2005, 2006, and 2007 during a three-year patch-burn grazing rotation followed by one year of rest in west-central Missouri. The solid line represents mean differences within patches burned in 2005. The dotted and dashed lines represent mean differences in patches burned in 2006 and 2007, respectively. Error bars represent SE.

DISCUSSION

This PBG experiment was aimed at quantifying the effect of the fire and grazing interaction to the plant community over time. Pyric herbivory is based on the expectation that grazers will concentrate foraging in recently burned areas when given a choice (Duvall and Whitaker 1964; Fuhlendorf and Engle 2001; Fuhlendorf et al. 2009; Leis et al. 2013). To validate that the fire–grazing interaction followed our expectations, Jamison and Underwood (Missouri Department of Conservation, unpub. data) simultaneously measured vegetation structure at four of our study sites. They examined differences in visual obstruction, the standard deviation of vegetation height, and the coefficient of variation between grazed and ungrazed units during the same three-year PBG rotation and one year of rest (2005 to 2008). Jamison and Underwood concluded that vegetation height was shorter in patches within grazing units that were burned the same year data were collected. Vegetation height in patches within the grazing units that had not been recently burned were similar in vegetation height to that of patches within the adjacent ungrazed units indicating that cattle focused grazing pressure on the most recently burned patches as presumed. A strong grazing preference for recently burned patches was also reported by Vermeire et al. (2004) such that cattle reduced grass standing crop by 78% within burn patches compared to 19% outside the influence of the burn. Leis et al. (2013) reported similar findings when quantifying the effects of a fire–grazing interaction at Tallgrass Prairie National Preserve in Kansas. Additionally, McGranahan et al. (2012a) found that management through pyric herbivory can create patch contrast in vegetation structure across a broad range of precipitation and plant community types. Our data support the fire–grazing interaction as well in determining changes to the plant community within a most recently burned patch, followed by a return or returning trend to ungrazed conditions after one year post-burn.

Our assessment of FQI and diversity measures described in this paper was aimed

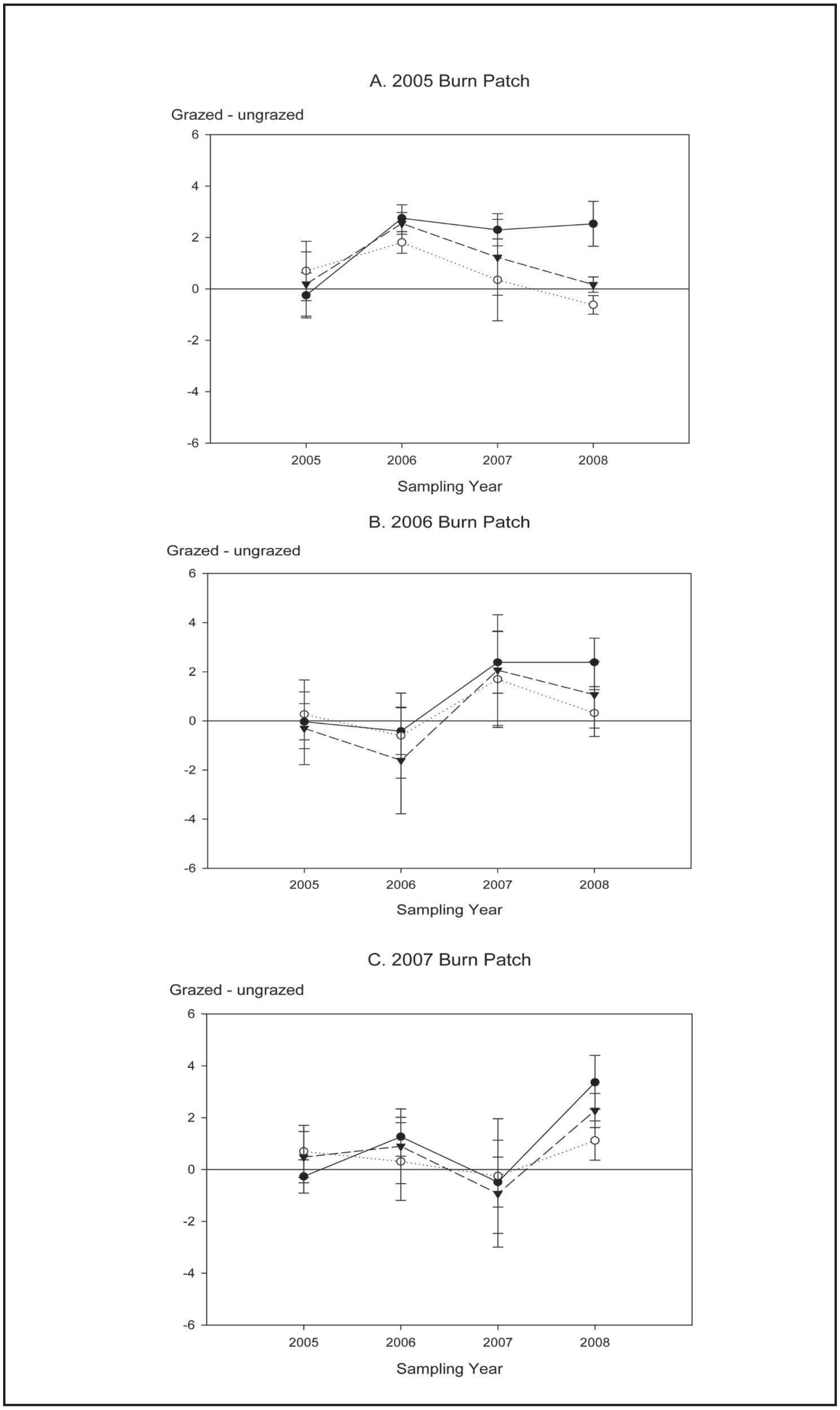

Figure 4. Mean differences between sampling plots subjected to patch-burn grazing and ungrazed plots within exclosures regarding plant species richness and diversity indices in patches burned in 2005 (A), 2006 (B), and 2007 (C) during a three-year patch-burn grazing rotation followed by one year of rest in west-central Missouri. The solid line represents mean differences in species richness. The dotted and dashed lines represent mean differences of Simpson and Shannon index values, respectively. Error bars represent SE.

at evaluating the effect of PBG on plant community dynamics. The FQI has been applied in a variety of research contexts to indicate changes in plant community composition (Jog et al. 2006). However, the index tends to be biased against species that are typically more tolerant of disturbance. For example, taxa with low fidelity to a habitat and adapted to severe disturbances where only brief periods are available for growth and reproduction are usually given a CC value of 0–1 (Taft et al. 1997) regardless of their ultimate importance to the grassland community. As cattle heavily grazed recently burned patches, we observed a temporary decrease in FQI and species richness. However, one year post-burn, species richness was greater in grazed plots and FQI returned to a state similar to that of ungrazed plots. The rapid recovery of FQI and species richness indicates that even though some plants were not observed during the year of a burn and focal grazing, the vegetative composition persisted and recovered the following year. As total species richness was strongly driven by native species, the increase in annual and biennial species with low CC values one year following a burn explains how species richness increased in grazed plots with only slight differences in FQI. This was evidenced when considering the difference in richness of annual/biennial species between grazed and ungrazed plots (Figure 3). While differences were less than 1 species/m² and not significant, the trend shows annual species richness increases one year post-burn across all patches.

While the FQI describes changes in the character of the plant community, diversity indices allow for standard comparisons of abundance and distribution (Magurran 2004). Patterns in our species diversity data highlight the temporal nature of plant community responses to the treatment. Although 95% CI of both mean species richness and diversity indicate grazing had no significant negative impact (Figure 5), interactions with patch and year show that effects of PBG on species richness within patches differed temporally. The temporal nature of these responses shows a shift in the expression of species rather than a shift in species presence. Data show that conditions in the most recently burned

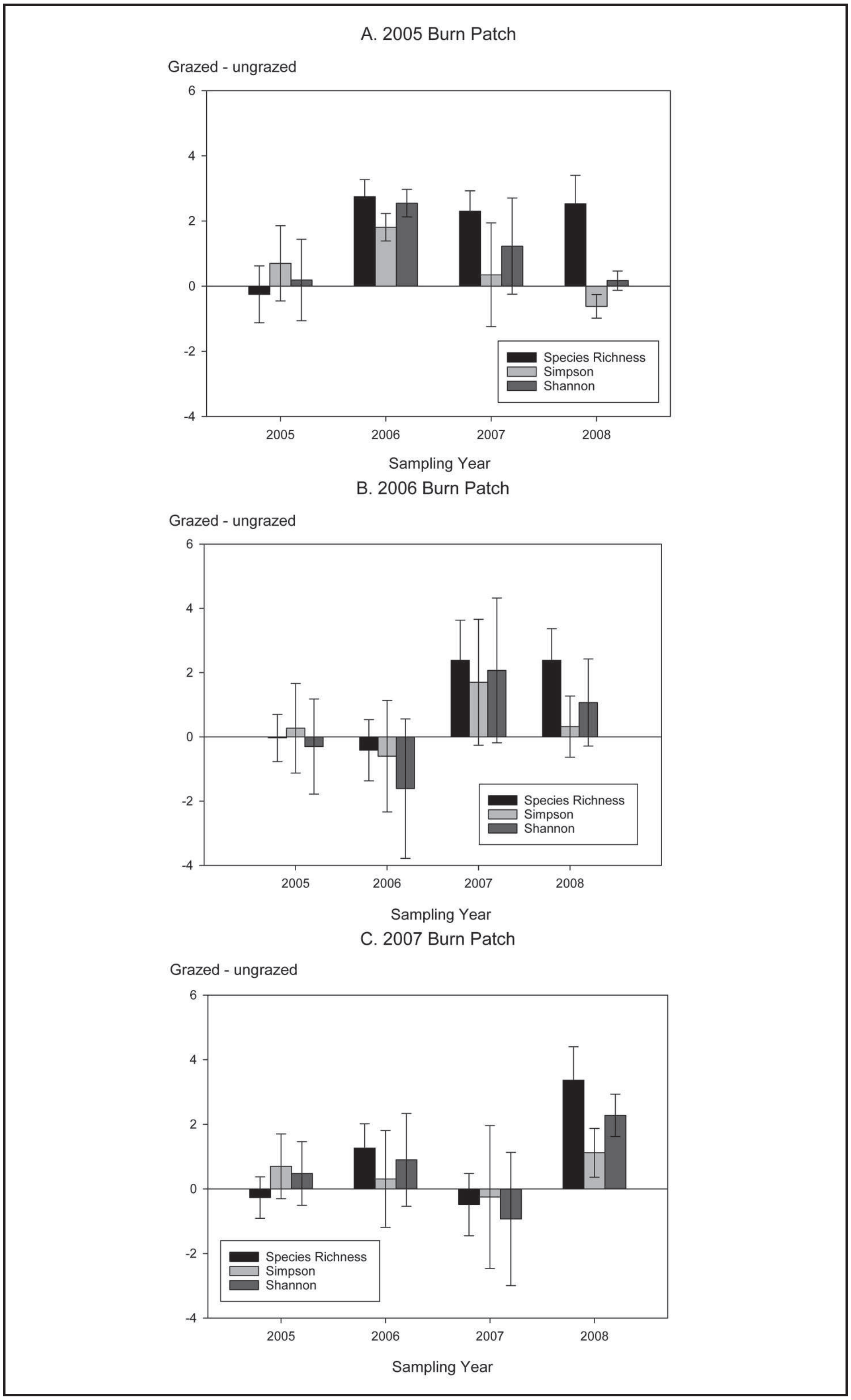

Figure 5. Mean differences between sampling plots subjected to patch-burn grazing and ungrazed plots within exclosures regarding plant species richness and diversity indices in patches burned in 2005 (A), 2006 (B), and 2007 (C) during a three-year patch-burn grazing rotation followed by one year of rest in west-central Missouri. Error bars represent SE.

patches resulted in greater expression of lower CC species such as annuals and those tolerant of disturbance (Figure 3). Species with greater CC values were recovering one year post-burn as conditions (i.e., increased bareground and sunlight) remained adequate for the lower CC species that benefited from the recent disturbance. We, therefore, saw a temporal increase in diversity and species richness one year post-burn. As time since focal grazing and fire increased, expression of these species along with the species of greater CC value returned to a baseline state similar to plots that were not grazed.

An increase in germination and vegetative reproduction may also explain the increase in species richness/m². Fuhlendorf and Engle (2004) found that recently burned and grazed patches are characterized by greater bare ground and forb cover while higher vegetation structure and litter occur in patches with more time since fire and reduced grazing pressure. Grazing was found to facilitate rare species establishment with the addition of seed while there was no effect of seed addition without grazing (Wilsey and Martin 2015). This was attributed to less build-up of aboveground biomass and litter in grazed plots (Wilsey and Martin 2015). Dynamics of the plant community and responses from fire are driven by germination (Van Staden et al. 2000; Jefferson et al. 2008) and vegetative reproduction such as rhizomes and stolons (Benson and Hartnett 2006). Smoke and heat may also increase water infiltration rates of seeds and the lower albedo of ash and bare ground contribute to soil warming. While we did not track seedlings or clonal ramets in our study, the increase in species richness/m² one year post-burn is consistent with these mechanisms.

Previous studies have also reported increases in floristic diversity and species richness from the use of fire and grazing to establish presettlement processes and natural disturbance patterns (Collins 1987; Hartnett et al. 1996; Collins et al. 1998; Hickman et al. 2004; Brudvig et al. 2007). Abundance of perennial and annual forb species have been found to increase from fire and grazing (Collins 1987; Fahnestock and Knapp 1994; Augustine and McNaughton 1998; Coppedge et al. 1998; Hayes and Holl 2003). Noy-Meir et al. (1989) reported that under light to moderate grazing, grass cover decreased to the benefit of species of a wide range of families and growth forms. This response was attributed to the opening of establishment gaps in the dense grass foliage and litter (Noy-Meir et al. 1989). Similar to cattle, other large grazing mammals have been reported to contribute to species diversity through nutrient cycling effects of urine deposition (Steinauer and Collins 1995). Moreover, Symstad and Jonas (2011) reviewed published studies on impacts of moderate grazing on tallgrass prairie and found reports of positive responses to species richness and diversity.

Deterioration of plant communities through the mismanagement of livestock grazing has been documented on tallgrass prairie remnants (Kucera 1956; Nyboer 1981; McGranahan et al. 2012b). While the presence of exotic species can affect FQI assessments, they are not directly included in the FQI calculation. However, an increase in exotic species could lead to a reduction of native plants within the plots thereby reducing FQI. Our data do not support this as few exotic species were recorded, and those that were had patchy distributions. Investigations of grazing impacts to prairie reconstructions have also shown no increase in exotic seedling emergence (Martin and Wilsey 2006). McGranahan (2011) found that the proportion of live tall fescue (*Lolium arundinaceum* Schreb. S.J. Darbyshire, syn. *Festuca arundinacea* (Schreb.) Darbysh.) increased with patch-burn grazing thereby reducing fire spread. Later research indicated that the reduction of fire spread and the presence of exotic species such as tall fescue can prevent patch contrast and the restoration of pyric herbivory (McGranahan et al. 2012b). Horizontal structure data gathered by Jamison and Underwood (unpub. data) and direct observations were that cattle had a strong fidelity to the burn patch in our study. This supports the species-level observations of low numbers of exotic species.

Our study shows that changes in floristic quality, species richness, or diversity following a three-year PBG cycle and one year of rest are temporal in nature. These metrics temporarily declined through the first burn-graze treatment followed by an increase in floristic quality and diversity during the next growing season when compared to nearby ungrazed plots. Our design does not provide the data necessary to determine impacts on individual species. Despite post-hoc analyses to determine treatment effects on 11 relatively common species that would be of interest (*Andropogon gerardi* Vitman*, Stylosanthes biflora* [L.] Britton, Stearns & Poggenb.*, Euthamia gymnospermoides* Greene ex Porter & Britton*, Echinacea pallida* [Nutt.] Nutt.*, Amorpha canescens* Pursh*, Baptisia bracteata* Muhl. ex Elliott var. *leucophaea* [Nutt.] Kartesz & Gandhi, *Helianthus mollis* Lam., *Lespedeza virginica* [L.] Britton, *Pycnanthemum tenuifolium* Schrad., *Ruellia humilis* Nutt., and *Viola sagittata* Aiton), significance was not detected as the degree of error was large. Species often have clumped distributions and can take many years for effects to manifest. Therefore, a greater sampling effort and longer time of study would be required to determine impacts to individual species.

MANAGEMENT IMPLICATIONS

Ecologists and managers are increasingly recognizing the significant interaction of fire and grazing and its importance to functioning grassland ecosystems. There is, however, concern regarding effects on the vegetation community when using cattle as a part of a grazing component in prairie remnant management. Our study did not address effects on rare plant species or long-term changes to the plant community following multiple rotations of PBG. Nor did it investigate how previous management histories may factor into vegetative changes from PBG. We evaluated PBG in remnant tallgrass prairie, so the potential role of PBG as a tool in prairie restoration and reconstruction also needs to be evaluated as well as the effects of using PBG when established populations of exotic species are present. However, we have found that the plant community as a whole retained its structure and character over a single three-year rotation on floristically intact tallgrass prairie. Managers will benefit from long-term monitoring of the plant community when implementing PBG.

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