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Research Article

Alternative Grazing Management Strategies Combat Invasive Grass Dominance

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ABSTRACT

Changes in historical disturbance regimes have increased the susceptibility of Great Plains ecosystems to various threats, including invasive species. Kentucky bluegrass (*Poa pratensis*; hereafter bluegrass) invasion in the northern Great Plains is displacing native species and has created novel ecosystems with no historical precedent to guide management. Traditional season-long (SL) grazing management increases bluegrass abundance, so we conducted a field experiment to determine if alternative early-intensive (EI) and patch-burn (PB) grazing management strategies can elicit more desirable outcomes in bluegrass-invaded pastures. Alternative EI grazing involved a triple of the stock density, compared to SL and PB stock density, for the first third of the grazing season while PB grazing incorporated SL grazing with prescribed burns. We randomly assigned treatments to 16-ha pastures with three replicates per treatment ($n = 3$). We conducted vegetation cover surveys and collected aboveground biomass samples to analyze plant community dynamics and production at study initiation and after 4 y of treatment. The SL grazing treatment increased bluegrass abundance by approximately 20% after 4 y of treatment while the alternative EI and PB grazing maintained its abundance at study initiation levels. Compositional differences also indicated increased native plant species cover with alternative management. Annual aboveground biomass production was not affected by management, but has the potential to become less variable with PB grazing in bluegrass-dominated pastures. Our results provide promising evidence that alternative grazing management may help combat bluegrass invasion by preventing additional increases in its abundance in the northern Great Plains.

Index terms: grazing systems; Kentucky bluegrass; mixed grass prairie; patch burn; plant community composition

INTRODUCTION

Patterns and interactions among fire and grazing shaped the Great Plains ecosystems as heterogeneous grasslands (Fuhlendorf and Engle 2004; Samson et al. 2004). Successional recovery, according to time since fire and grazing pressure, drove differences in plant communities and vegetation structure that shifted across the landscape over time (Coppedge and Shaw 1998). European settlement subdued this historical spatial and temporal disturbance pattern as fire suppression and strategic cattle grazing were introduced to the Great Plains region (Umbanhowar 1996; Samson et al. 2004). Traditional grazing management strategies were developed to promote uniform, moderate forage utilization within pastures (Fuhlendorf and Engle 2001). Consequently, these strategies typically resulted in reoccurring heavily disturbed and undisturbed areas as cattle preferentially graze areas of low stature regrowth while avoiding tall stature, mature plants (Bailey et al. 1998; Coppedge and Shaw 1998; Fuhlendorf and Engle 2004; Limb et al. 2010). Exacerbated by the lack of fire, the legacy of traditional management and its use still today have simplified Great Plains ecosystems and increased their vulnerability to ecosystem threats, especially invasive species, as biodiversity declines (Fuhlendorf and Engle 2001; Toledo et al. 2014).

Kentucky bluegrass (*Poa pratensis*; hereafter bluegrass) invasion is actively altering grassland ecosystems in the northern Great Plains (Toledo et al. 2014). Increased bluegrass abundance (Miles and Knops 2009) is threatening the biodiversity of grasslands by displacing native plant species (Cully et al. 2003). Evidence suggests that the expansion of bluegrass invasion over the last 30 y may be a response to the combination of multiple factors including outdated land management strategies, alterations to historical fire regimes, and increases in growing season length, atmospheric CO_2 concentrations, and average precipitation (Toledo et al. 2014; DeKeyser et al. 2015; Limb et al. 2018). The origin of bluegrass in the United States is not clear (Carrier et al. 1916), but bluegrass was not a part of the region's historical plant community (Barker et al. 1986) and was likely introduced to the region in the 1800s (DeKeyser et al. 2015). Furthermore, the US Department of Agriculture (USDA) lists bluegrass as a naturalized, invasive species throughout the tallgrass, mixed-grass, and shortgrass prairie ecoregions (USDA-NRCS 2018). The reason bluegrass has become so problematic is likely related to its biology. Bluegrass is a cool-season perennial grass that begins growth in early spring before the native cool-season species that dominate native prairie in the northern Great Plains. Bluegrass also develops a dense root and thatch layer that restricts the establishment and growth of other species (Pierson et al. 2002; Toledo et al. 2014) by reducing light penetration of the understory and altering surface hydrology (Taylor and Blake 1982), soil structure (Herrick et al. 2001), and nutrient cycling (Badra et al. 2005). Ecosystem alterations resulting from bluegrass invasion have developed novel ecosystems throughout most of the northern Great Plains that may be irreversible (Toledo et al. 2014). Controlling the invasion of bluegrass by reducing or preventing additional increases in its abundance will be necessary to limit its inhibitory effects and preserve native biodiversity.

The development of novel bluegrass-invaded grasslands in the northern Great Plains has gone largely unnoticed until recently, likely because bluegrass can provide adequate forage for livestock production, especially during early spring and wet years (Toledo et al. 2014). However, the forage value of bluegrass drastically declines with heat and water stress of the summer months and periods of drought that induce dormancy (Hockensmith et al. 1997). These periods are projected to increase in length as precipitation timing shifts under future climate scenarios with varying consequences on ecological communities (Alexander et al. 2006; Deguines et al. 2017). Bluegrass dominance, therefore, may leave the forage supply susceptible to climate change within and across seasons. Conversely, a diverse community would provide adequate forage across a wider range of conditions. Regardless of grazing intensity, bluegrass increases under season-long (SL) grazing management in the absence of fire (Smith and Owensby 1978; Cully et al. 2003; Murphy and Grant 2005; Limb et al. 2018), which implies that its invasion expands under uniform disturbance regimes. A change in grazing management practices will be necessary in the northern Great Plains to combat bluegrass invasion, maintain native biodiversity, and sustain forage production in an increasingly variable climate.

There is a lack of empirical evidence for effective grazing management strategies that will control (i.e., reduce the increase and/or abundance of) bluegrass in the cool-season mixed-grass prairie ecoregion of the northern Great Plains. Early-intensive (EI) grazing combined with annual burning can stabilize or decrease bluegrass populations in warm-season prairie (Smith and Owensby 1978). This strategy has not been investigated for bluegrass control in cool-season mixed-grass prairie, although it has the potential to reduce the early productivity of bluegrass before native warm-season species emerge. Moreover, the northern Great Plains is a fire-adapted system and evidence suggests that fire has the potential to control bluegrass invasion by reducing its cover. Studies in the tallgrass (Anderson et al. 1970; Owensby and Smith 1979; Smith and Knapp 1999) and mixed-grass (Engle and Bultsma 1984; Bahm et al. 2011; Kral et al. 2018) prairie ecoregions indicate that burning alone can decrease bluegrass. However, reductions are highly dependent on precipitation and soil type (Engle and Bultsma 1984) and are short term in mixed-grass prairie (Bahm et al. 2011; Kral et al. 2018). Patch-burn (PB) grazing management (i.e., fire followed by grazing) may reduce bluegrass cover for a longer period than burning alone in mixed-grass prairie but remains uninvestigated. Furthermore, PB grazing is known to promote structural and compositional diversity (Fuhlendorf and Engle 2001, 2004; Powell et al. 2018) as the northern Great Plains evolved under, and is resilient to, the interactions among fire and grazing (Samson et al. 2004; Fuhlendorf et al. 2009; Vermeire et al. 2011; Strong et al. 2013; Gates et al. 2017; Powell et al. 2018). Analyzing the effects of EI and PB grazing management strategies on bluegrass-invaded pastures of the northern mixed-grass prairie is relevant to determine if these alternatives can help control bluegrass where traditional SL grazing does not.

The consequences of bluegrass invasion remain largely unknown for northern Great Plains ecosystems and are highly speculative (Toledo et al. 2014). Therefore, investigating how bluegrass-invaded communities respond to management is necessary to identify sustainable methods to reduce or prevent further increases in bluegrass and quantify differences that can help improve the management of these novel ecosystems. In our study, we hypothesized that SL grazing and two alternative grazing strategies, EI and PB, would generate differences in plant community metrics (richness, evenness, and diversity) and composition. Because native Great Plains plants evolved with fire and grazing (Samson et al. 2004), we also hypothesized that native species abundance would be greatest with PB grazing while aboveground biomass production would be unaffected by management. To address these hypotheses, our specific objectives were to quantify the influence of each grazing management strategy on plant community dynamics, plant community composition, the cover of specific species groups, bluegrass cover in relation to other individual species of concern, and annual aboveground biomass production. Improving our understanding of how different land management strategies influence invaded ecosystems can elucidate ways to conserve biodiversity and simultaneously maintain essential ecosystem services in the face of increasing global change.

METHODS

Site Description

Our field experiment took place at the North Dakota State University (NDSU) Central Grasslands Research Extension Center (CGREC) located near Streeter, North Dakota, USA (46°44′N, 99°27′W). The center lies within the Missouri Coteau ecoregion and the region's vegetation is representative of mixed-grass prairie historically dominated by green needlegrass (*Nassella viridula*) and western wheatgrass (*Pascopyrum smithii*), both native cool-season grass species. However, nonnative cool-season grass species bluegrass and smooth brome (*Bromus inermis*), and the native shrub western snowberry (*Symphoricarpos occidentalis*), are important drivers of biodiversity changes and currently dominate the area's rangelands (Limb et al. 2018).

Central North Dakota has a continental climate with the coldest mean monthly temperature (–17 °C) occurring in January and the warmest (27 °C) occurring in July and August. Mean annual precipitation is 468 mm and about 70% falls between May and September (NDAWN 2018). Total annual and growing season monthly precipitation for each year of the study (2014, 2015, 2016, and 2017) are summarized in Figure 1 (NDAWN 2018; NOAA NCEI 2018). The growing season allows the growth of both cool- and warm-season plant species and has increased by an average of 12 d in parts of North Dakota from the historical average of 120 frost-free days (Badh et al. 2009; DeKeyser et al. 2013). Fine-loamy mollisols and irregular rolling plains dominate the region from the collapse of "superglacial" sediment (Bluemle 1991).

Experimental Design

We initiated a completely randomized design experiment on nine approximately 16-ha bluegrass-invaded pastures with three treatments each replicated three times ($n = 3$) in 2014. Treatments followed traditional and alternative grazing management strategies to compare their influence on vegetation

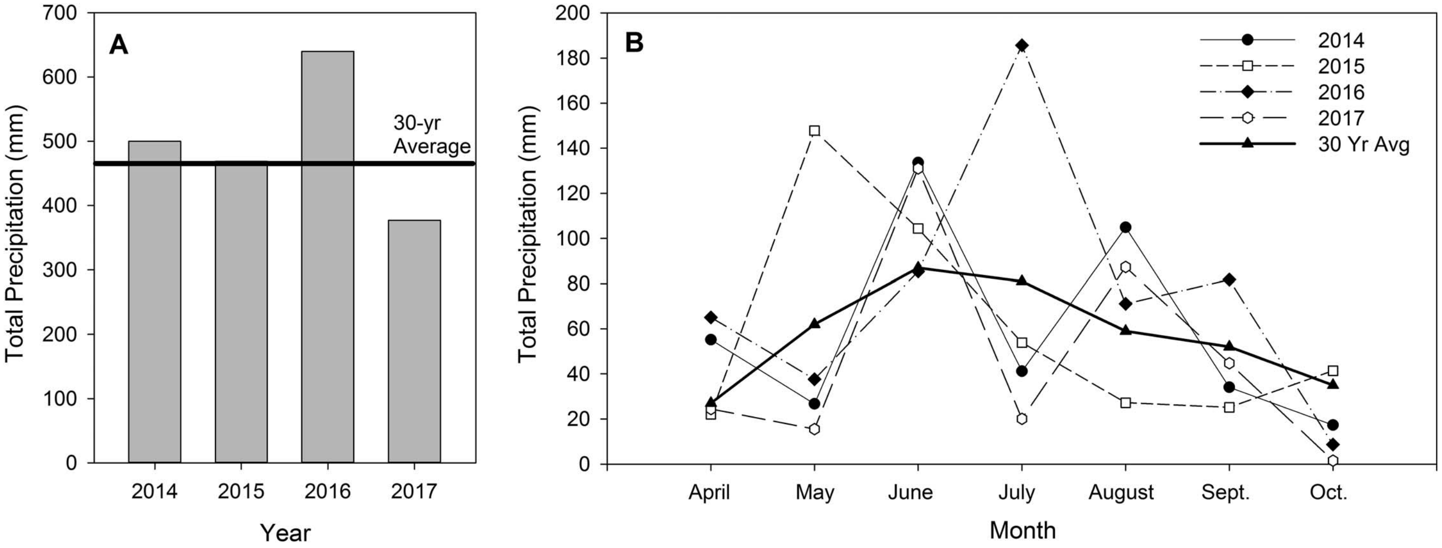

Figure 1.—Total annual precipitation (A) and monthly precipitation for each growing season (B) in Streeter, North Dakota, USA, for 2014 through 2017. Weather data acquired at the STREETER 7 NW, ND US USC00328415 station (46.7154°N, –99.4475°W) and retrieved from the National Environmental Satellite Data and Information Service of the National Oceanic and Atmospheric Administration (NOAA NCEI 2018). Thirty-year monthly averages for Streeter, North Dakota, were retrieved from the North Dakota Agricultural Network (NDAWN 2018).

dynamics. Each grazing treatment incorporated moderate herbivory with yearling mixed-breed heifers on three pastures. We stocked each pasture with the same animal unit months (1.85 AUM/ha) since all pastures were similar in carrying capacity. Grazing treatments and cattle did not rotate among pastures within or across years. Fencing existed only on the exterior boundary of each pasture to allow cattle to make their own foraging decisions within each pasture. Prior to treatment initiation, all pastures were grazed uniformly with a short-duration rotational grazing system for the previous 25 y, which is not a method tested in this study.

Grazing treatments emulated practices typical of, or alternative to, the study region. Traditional grazing management involves stocking pastures for the duration of the growing season. Thus, we stocked pastures in early May for 12 wk to establish a SL grazing treatment. Our alternative grazing management treatments consisted of EI and PB grazing. EI grazing involved stocking pastures at triple stock density for the first third of the growing season (4 wk: early May to early June). Therefore, EI grazed pastures were grazed intensely over a short duration but still received the same stocking rate as SL and PB pastures (1.85 AUM/ha). PB grazing combined the SL grazing strategy with a prescribed-fire regime where we burned a different fourth (4 ha) of each PB grazed pasture annually after a hard frost using a ring-fire technique on cured fuels. Weather conditions for burn operations were generally 50–70 °C with wind speeds $<$10 km/hr and relative humidity 40–60%. The first burn occurred in October 2014, so SL and PB pastures received the same grazing treatment in 2014. Grazing intensity does not influence plant community composition at our study location (Limb et al. 2018) and one partial season of grazing would not elicit changes in data collected at treatment initiation in 2014. Three patches of each PB pasture were burned prior to the 2017 growing season, so the effects of all grazing strategies were independent based on treatment in 2015–2017. Although not all patches were burned before the final round of data collection, 2017 represented the full completion of the PB grazing treatment because time-since-fire differed among each patch (i.e., patches included no burn, 1 y post-burn, 2 y post-burn, and 3 y post-burn) and grazing occurred over a total of four growing seasons.

Data Collection

We standardized sampling intensity among pastures by arbitrarily dividing (i.e., without fencing) all pastures regardless of grazing treatment into four equal patches that aligned with the PB treatment plan. Plant community surveys occurred between late July and early August in 2014 and 2017 to test for differences among grazing treatments at their initiation and completion. Surveys involved estimating the canopy cover of all vascular plants in each patch by subsampling with quadrats every 2 m along a 40-m transect (3 grazing strategies × 3 pastures × 4 transects × 20 quadrats = 720 sampled quadrats each year). We placed transects at new locations randomly each year with boundaries beginning and ending within loamy ecological sites, the dominant ecological site of our study location. Transects never intersected and were at least 50 m from one another and treatment perimeters. We identified and recorded the canopy cover of all plants within 0.25-m^2 frames using a modified Daubenmire cover class system (1 = trace–1%, 2 = 1–2%, 3 = 2–5%, 4 = 5–10%, 5 = 10–20%, 6 = 20–30%, 7 = 30–40%, 8 = 40–50%, 9 = 50–60%, 10 = 60–70%, 11 = 70–80%, 12 = 80–90%, 13 = 90–95%, 14 = 95–98%, 15 = 98–99%, 16 = 99–100%; Daubenmire 1959). We converted cover classes to midpoint values for statistical analyses. Plant nomenclature, status, and distribution followed the USDA National Plant Database (USDA-NRCS 2018).

Annual aboveground biomass production was measured at the peak of each growing season (mid-July) to compare the annual amount of forage available to livestock with each treatment. We placed three 1 × 1 m grazing exclosures,

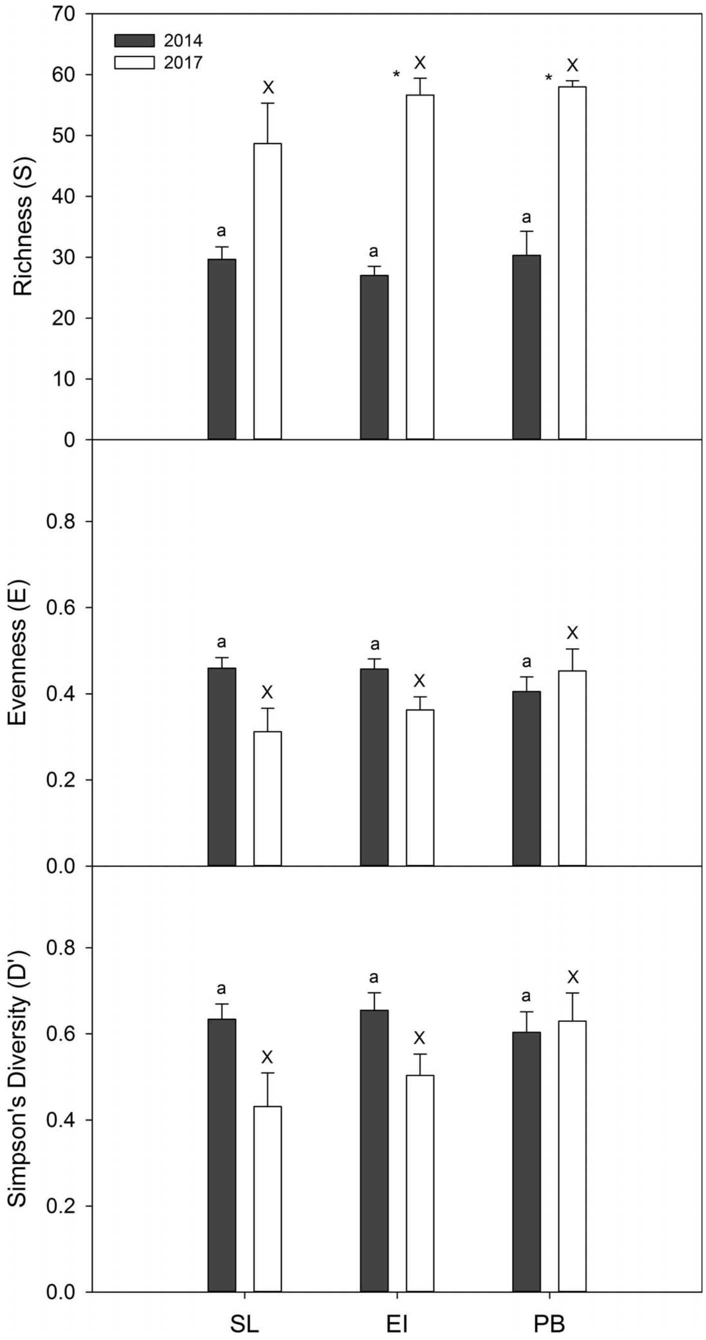

Figure 2.—Summary statistics of plant community dynamics at study initiation (2014) and completion (2017). Richness, evenness, and Simpson's diversity averages for pastures treated with season-long (SL), early-intensive (EI), and patch-burn (PB) grazing ($n = 3$). Different scaling among graphs. Treatment means with a common letter are not statistically different within each year ($p > 0.05$). Asterisks (*) represent a statistical difference between 2014 and 2017 means for each treatment ($p \leq 0.05$). Error bars represent standard error of the mean.

constructed of wire panels with 15-cm mesh, randomly at new locations in each patch of every pasture during the preceding dormant season in 2014 and 2017 (3 grazing strategies × 3 pastures × 4 patches each × 3 exclosures = 108 grazing exclosures placed randomly each year). Sampling involved clipping all standing biomass, including live and dead material, to the soil surface from one random 0.5 × 0.5 m quadrat placed within each exclosure. Samples were oven dried at 60 °C for 48 hr and weighed.

Data Analysis

We evaluated treatment effects by comparing 2014 and 2017 data as each year represented the status of all pastures at treatment initiation and full treatment implementation, respectively. We first determined mean species richness, evenness, and diversity (Simpson's diversity $D' = 1/\sum p_i^2$ where p_i refers to the proportion of each species i to the total number of species) for each pasture in 2014 and 2017. Invasive species and compositional shifts can contribute to values of richness, evenness, and diversity (Prach and Hobbs 2008), so we then tested for differences in species composition among treatments within and across years using all plant species cover data from the 80 quadrats in each pasture with nonmetric multidimensional scaling (NMS) procedures in PC-ORD 6.0. Based on the nature of our data, we used the Euclidean (Pythagorean) distance measure in autopilot mode to determine the best fit (lowest stress) in a possible 1 to 6 dimensional solution (McCune and Mefford 2011). To test for differences among treatments within and across years, we used permutation-based nonparametric multivariate analysis of variance (perMANOVA) in PC-ORD 6.0 also with the Euclidean (Pythagorean) distance measure.

We then analyzed the compositional data more directly by combining species into predefined groups to explain compositional shifts (native grasses, nonnative grasses, native forbs, nonnative forbs, native shrubs). To compare treatment effects on species groups, we determined the mean difference between treatment completion and treatment initiation averages for each group by subtracting 2014 from 2017 averages (e.g. 2017 − 2014). Positive differences reflected an increase in cover of a species group in 2017 while negative differences reflected a decrease. Not all nonnative species are classified as invasive and not all invasive species are nonnative, so we compared the cover of individual invasive species of concern among treatments within and across years including bluegrass, smooth brome, and western snowberry (Limb et al. 2018). We also examined the cover of green needlegrass and western wheatgrass, as they are the region's most dominant native cool-season species. In addition, we included little bluestem (*Schizachyrium scoparium*), a warm-season grass species known to increase under PB grazing management (Limb et al. 2011a). Finally, we examined aboveground biomass production data to identify if and how grazing management practices affected the forage supply for livestock. We analyzed all data with one-tailed *t*-tests (with a specified value of zero) or ANOVA and post hoc Tukey's B means separation with normally distributed and untransformed data in the IBM-SPSS Statistics software package (Version 25; IBM 2017).

RESULTS

We recorded 96 plant species from all the quadrats surveyed over both sampling years (2014 and 2017). Grazing treatments did not differ in species richness, evenness, or diversity at their initiation or the end of the study ($p > 0.05$; Figure 2). Between

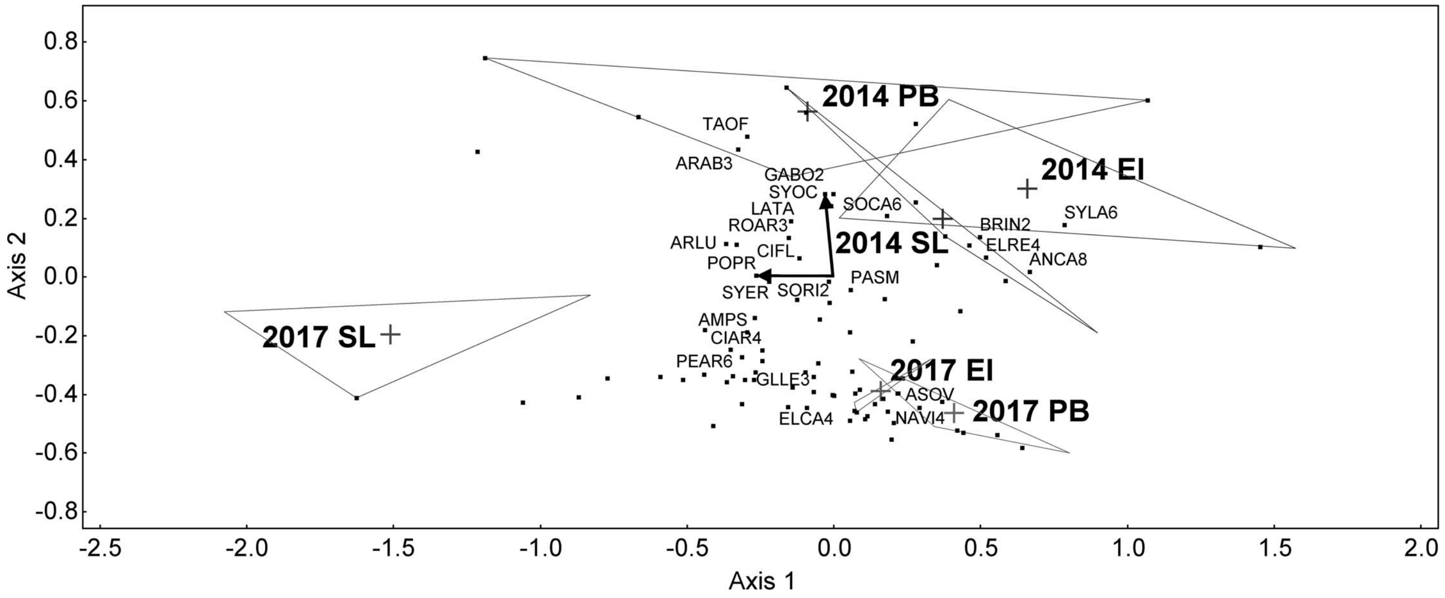

Figure 3.—NMS ordination of all 2014 and 2017 species along axes 1 and 2 with Euclidean (Pythagorean) distance separation showing convex hulls for each year by treatment. Greater separation between convex hulls suggests greater differences in plant community composition and less separation implies greater similarities. Plant species composition was collected at the study's initiation (2014) and completion (2017) on pastures treated with early-intensive (EI), patch-burn (PB), and season-long (SL) grazing management strategies at the North Dakota State University Central Grasslands Research Extension Center near Streeter, North Dakota. Points represent individual plant species' scores with the most dominant species identified according to USDA Plants Database symbols. Vectors indicate the species that have the strongest relationship with axis 1 (POPR, $r = -0.998$) and axis 2 (SYOC, $r = 0.987$).

years, only species richness with EI and PB grazing increased significantly in 2017 compared to 2014. Conversely, we found year ($p = 0.005$), treatment ($p = 0.047$), and year × treatment interaction effects ($p = 0.013$) in the plant community composition data (Figure 3). The NMS analysis of all 2014 and 2017 plant species data was useful (stress = 1.553) and resulted in a 2-dimensional solution (Figure 3). Axis 1 explained 90% of the variability while axis 2 explained 9.7% (99.7% cumulative). Generally, treatment by year centroids were oriented along the secondary axis according to year with shifts along the primary axis in 2017 according to grazing treatment. This indicates that plant community composition was similar among all treatments in 2014 while composition in EI and PB grazed pastures was similar and different from SL composition in 2017, so we explored annual differences between treatments more explicitly. Plant community composition was similar across all grazing treatments at their initiation in 2014 ($p = 0.51$) and a one-dimensional ordination was found (stress = 7.8) that explained 93.9% of the variability in the data. We also found a one-dimensional ordination that was useful for 2017 data (stress < 0.01) that explained 99.0% of the data's variability. In contrast to the 2014 results, the perMANOVA procedure on 2017 data revealed that composition was affected by grazing treatment at the study's completion ($p = 0.02$). Pairwise comparisons further revealed that EI and PB grazing treatments resulted in similar community composition that differed from the SL grazing treatment in 2017.

Treatment differences in plant community composition were attributed to changes in the cover of native and nonnative graminoids, native forbs, and native shrubs between 2014 and 2017. The cover of nonnative grasses, primarily bluegrass, increased in pastures treated with SL grazing without affecting the cover of any other defined species group (Figures 4 and 5). Specifically, canopy cover of bluegrass increased from 28% in 2014 to 48% in 2017 with SL grazing management while smooth brome was unaffected by management (Figure 5). The NMS analysis also indicated that bluegrass was strongly negatively correlated with axis 1 (Figure 3). EI and PB grazing did not alter nonnative grass or nonnative forb cover after 4 y of treatment. However, both alternative management strategies slightly increased the cover of native grass species and significantly decreased the cover of native shrubs. Furthermore, EI and PB grazing strategies neither increased nor reduced bluegrass cover but rather kept the cover near the study initiation (2014) average in 2017 (30% and 27%, respectively; Figure 5). Increased native grass cover with alternative EI and PB grazing management strategies was likely attributed to slight increases in the cover of several species collectively including green needlegrass, western wheatgrass, and little bluestem among others (Figure 5). Decreased native shrub cover was largely attributed to a reduction in western snowberry, which was the species most strongly correlated with axis 2 (Figures 3 and 5). Interestingly, the cover of native forbs increased from 0.2% in 2014 to 3.6% in 2017 with PB grazing management (Figure 4). We did not identify differences in aboveground biomass production among treatments at their initiation or completion (Figure 6). However, production was reduced from the 2014 average in 2017 with SL grazing management.

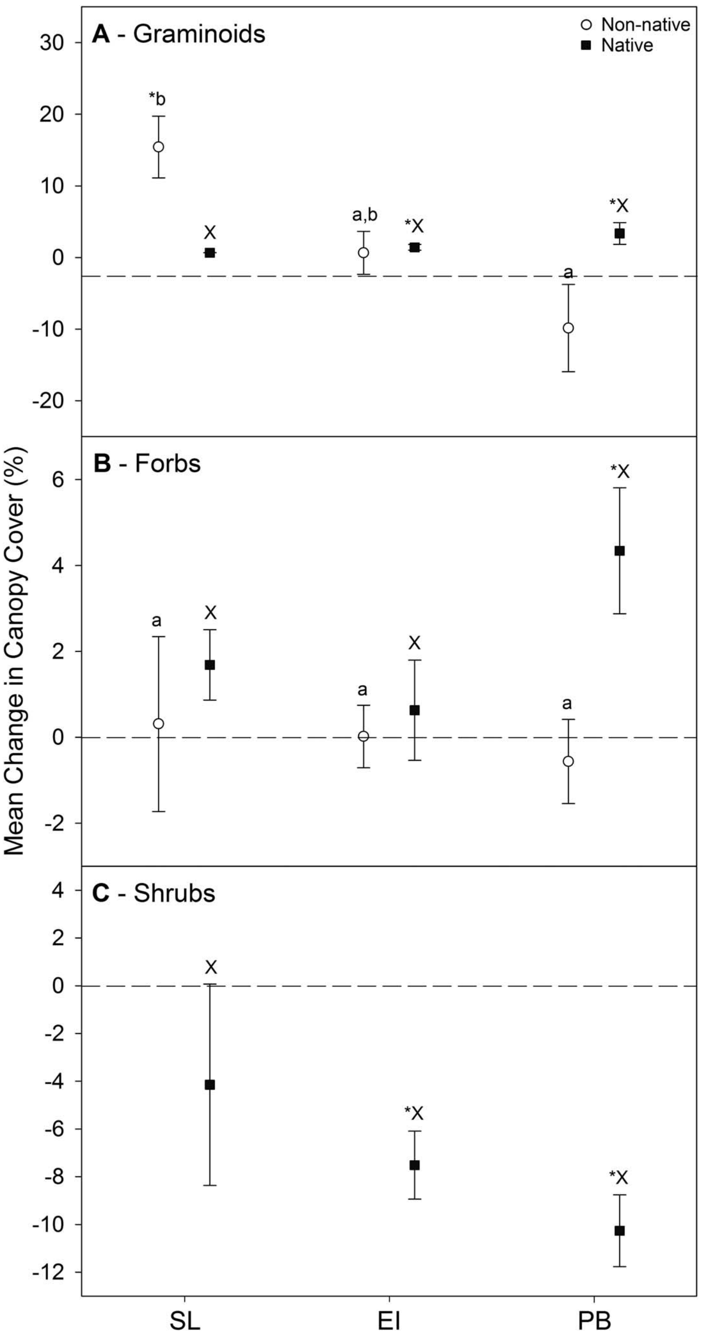

Figure 4.—Mean differences in the canopy cover of species groups between 2014 and 2017 for pastures treated with season-long (SL), early-intensive (EI), and patch-burn (PB) grazing at the North Dakota State University Central Grasslands Research Extension Center near Streeter, North Dakota ($n = 3$). Dashed lines represent no difference between 2014 and 2017 averages for each species group by treatment while asterisks (*) represent a statistical increase (above dashed line) or decrease (below dashed line) of the 2017 average from the 2014 average. Letters (a,b) represent statistical differences between grazing treatments within nonnative species groups ($p \leq 0.05$). Similar letters (X) within native species groups indicate no statistical differences among treatments ($p > 0.05$). Error bars represent standard error of the mean.

DISCUSSION

Land managers can be seriously challenged when faced with novel ecosystems as they often require modifications in management goals and practices to conserve biodiversity and the ecosystem services it supports (West 1993; Hobbs 2007; Kirkman et al. 2013). Accordingly, we found that alternative grazing management strategies may produce more desirable outcomes than traditional SL grazing management in novel bluegrass-dominated ecosystems of the northern Great Plains. In agreement with other studies, we found that SL grazing increased the abundance of bluegrass over time (Smith and Owensby 1978; Cully et al. 2003; Murphy and Grant 2005; Limb et al. 2018). On the contrary, alternative EI and PB grazing management strategies kept bluegrass cover from increasing beyond treatment initiation levels. We also identified differences in community composition after 4 y of treatment that suggest native species abundance increased in pastures grazed with EI and PB grazing management. We argue that our study serves as a call to action, despite its short 4 y duration, to continue investigating the potential for alternatives to improve the management of bluegrass-dominated grasslands in the northern Great Plains.

Pastures grazed with traditional SL and alternative EI and PB management strategies differed in community composition but not richness, evenness, or diversity at the end of our study. However, EI and PB grazing did elicit a slight increase in richness and the cover of native grasses and forbs after 4 y of treatment (Figure 4). These management strategies may have released native species from the inhibitory effects of bluegrass to drive these changes. EI grazing reduces the early productivity of bluegrass (Bryan et al. 2000) and increased grazing intensity reduces dead biomass accumulation (Naeth et al. 1991; Biondini et al. 1998). Thus, EI grazing may have stunted the early growth of bluegrass and limited its ability to establish a thickened thatch layer before the emergence of native species. Burning also releases native plants from the inhibitory effects of bluegrass by removing accumulated thatch (Kral et al. 2018) and regrowth increases grazing pressure in recently burned patches (Fuhlendorf and Engle 2004). Thus, we propose that increased grazing intensity following prescribed burning may have limited thatch accumulation post-fire and continued to release more later-emerging species from the inhibiting effects of accumulated thatch. Therefore, grazing after burning may decrease the ability of bluegrass to inhibit the growth of other species throughout more of the growing season than burning alone. Grazing during drought reduces the growth potential and vigor of bluegrass (Dong et al. 2011, 2014) and may have further amplified the release of native species in EI and PB grazed pastures with below average precipitation in 2017. Developing strategies that simultaneously promote natives and control invasive species is important for both biodiversity and ecosystem service conservation in novel ecosystems (Hobbs et al. 2009).

The cover of our native species of concern—western snowberry, green needlegrass, western wheatgrass, and little bluestem—did not differ among grazing treatments at the end of the study (Figure 5). However, western snowberry decreased from 2014 to 2017 in pastures treated with EI and PB grazing management. Woody encroachment by western snowberry,

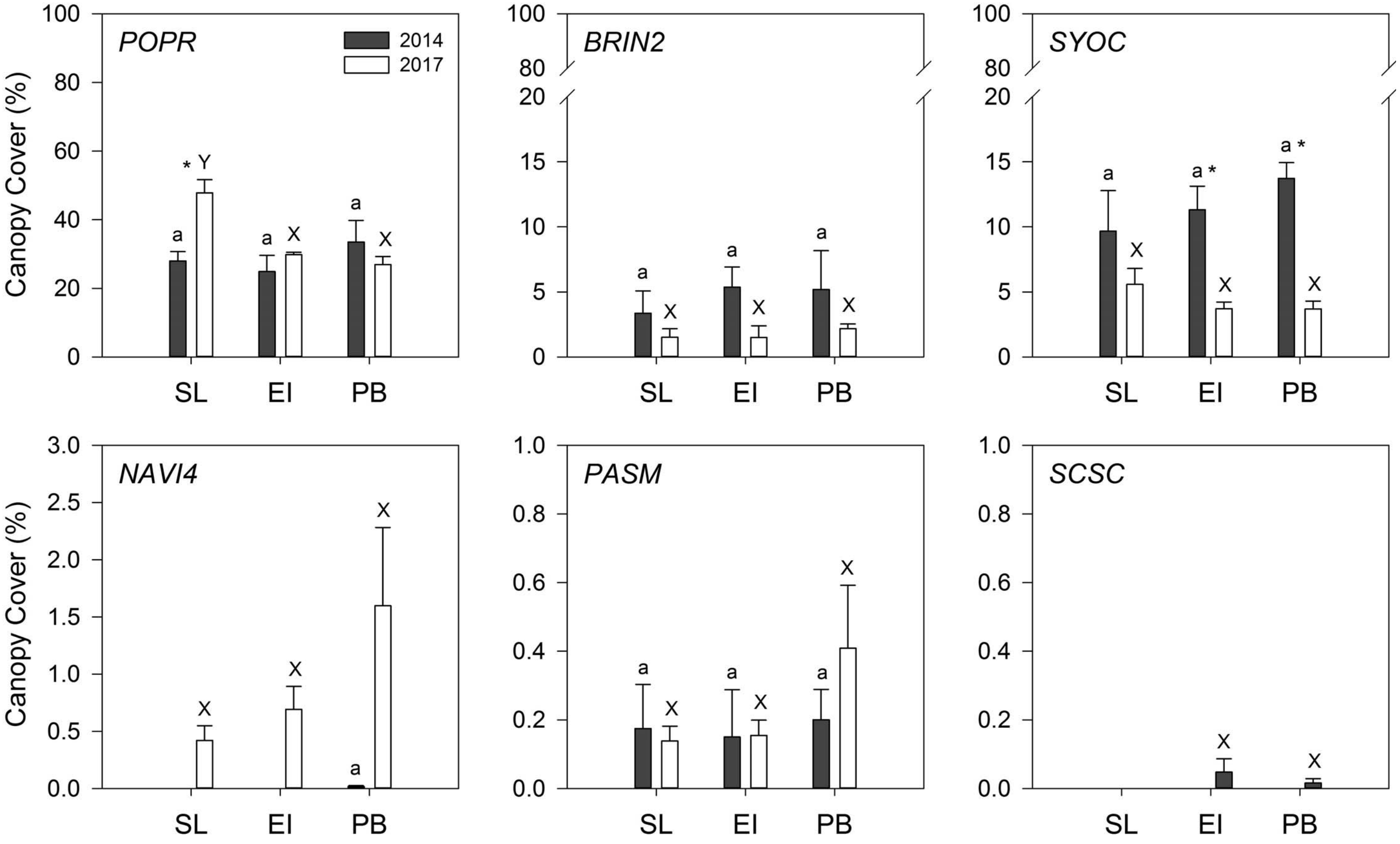

Figure 5.—Average canopy cover (%) of six species of concern on pastures treated with season-long (SL), early-intensive (EI), and patch-burn (PB) grazing at study initiation (2014) and completion (2017) ($n = 3$). Species are labeled according to the USDA plants database and include POPR - Kentucky bluegrass, BRIN2 - smooth brome, SYOC - western snowberry, NAVI4 - green needlegrass, PASM - western wheatgrass, and SCSC - little bluestem. Different scaling among graphs. Treatment means with a common letter are not statistically different within each year ($p > 0.05$). Asterisks (*) represent a statistical difference between 2014 and 2017 means for each treatment ($p \leq 0.05$). Error bars represent standard error of the mean.

among other trees and shrubs, reduces biodiversity and forage availability for livestock and wildlife habitat, so a reduction is desirable (Van Auken 2000; Grant et al. 2004). EI grazing may have altered the distribution of livestock and increased browse and trampling pressures on western snowberry by increasing the amount of grazed area early in the season (Brock et al. 2017). Similarly, PB grazing management may have reduced western snowberry cover by changing herbivore selection from plant- to area-specific and increasing its browse. Research has correlated this phenomenon with increased palatability of western snowberry regrowth following fire in PB grazed pastures at our study site (Johnson et al. 2019). Although not significant, observed differences in the cover of green needlegrass, western wheatgrass, and little bluestem from 2014 to 2017 warrant further investigation into the potential for these species to increase over time with PB grazing management (Figure 5).

As hypothesized, grazing management strategies did not differentially influence average aboveground biomass production in grazing exclosures at the end of our study (Figure 5). Therefore, PB grazing in mixed-grass prairie invaded by bluegrass neither reduced nor increased forage availability for livestock, as compared with SL and EI grazing, after 4 y of treatment. Research from the tallgrass prairie ecoregion suggests that increased spatial variability with fire and grazing management (e.g., PB grazing) reduces the temporal variability of aboveground biomass production (McGranahan et al. 2016). Moreover, unburned areas in PB grazed pastures provide a supplemental forage supply, or “grass bank,” during periods of drought where production may be limited in recently burned areas (McGranahan et al. 2014). Our study was not long enough to draw similar conclusions for the mixed-grass prairie, but observed changes in variance among treatments over time suggests further investigation is warranted.

Returning a novel ecosystem to a historical reference state is, generally, unattainable and defining what is “historical” or “natural” is often unclear (Hobbs et al. 2009). Given the extent of bluegrass invasion and its biology, we also cannot expect to feasibly eradicate bluegrass from the northern Great Plains (Toledo et al. 2014). Therefore, increasing native species abundance and limiting the cover of bluegrass are more appropriate management goals for the conservation of biodiversity and the ecosystem services it supports. Increasing native species abundance with alternative grazing management strategies, like EI and PB grazing, may not only benefit livestock production by increasing the predictability of forage production across years, but may also support other important areas of conservation concern. PB grazing, for example, increases heterogeneity of the landscape in a manner that provides a broad

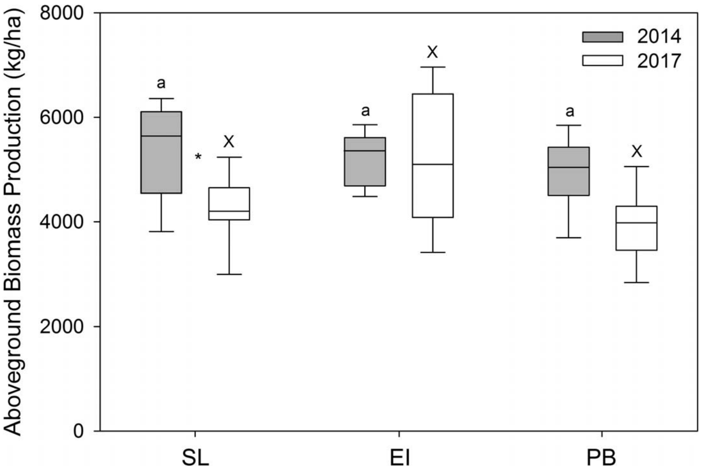

Figure 6.—Boxplot depicting annual aboveground biomass production (kg/ha) for pastures treated with season-long (SL), early-intensive (EI), and patch-burn (PB) grazing at their initiation (2014) and completion (2017). Production data was collected from grazing exclosures in three pastures for each treatment ($n = 3$) at the North Dakota State University Central Grasslands Research Extension Center near Streeter, North Dakota. Treatment means with a common letter are not statistically different within each year ($p > 0.05$). Asterisks (*) represent a statistical difference between 2014 and 2017 means for each treatment ($p \leq 0.05$). Error bars represent standard error of the mean.

range of habitat types for several species of insects, birds, and small mammals (Fuhlendorf et al. 2009; Scasta et al. 2016). EI grazing, on the other hand, may not produce this result as it increases the areal coverage of grazing even more than SL grazing (Brock et al. 2017). Therefore, EI grazing may homogenize vegetation structure and habitat type throughout a pasture and decrease insect and wildlife diversity. Confirming the effects of PB grazing management over larger spatial and temporal scales on these species groups in bluegrass-dominated grasslands and comparing those effects with SL and EI grazing management will be necessary to understand which strategy best supports biodiversity in all its forms before irreversible thresholds are crossed.

Management Implications

Our study suggests that, at least in the northern mixed-grass prairie, transitioning away from traditional SL grazing management to alternative strategies may be necessary to combat the deleterious effects of bluegrass invasion by limiting additional increases in its dominance. Alternatives PB and EI grazing, however, may affect livestock production differently despite their similar effects on plant community metrics in our study. EI grazing management results in decreased livestock weight gains (Grings et al. 2002) while PB grazing weight gains are either unchanged or improved (Limb et al. 2011b) as compared with weight gains associated with SL grazing in mixed-grass prairie. Improving the consistency and predictability of forage production across years will also become increasingly important for producers as climate change continues to alter precipitation patterns (Allred et al. 2014) and PB grazing may be a more desirable strategy to increase the consistency in aboveground biomass production across years (McGranahan et al. 2016). Additional research is necessary to confirm these findings over extended timespans and identify management specifics in bluegrass-invaded mixed-grass rangelands. Nevertheless, our results join a growing body of evidence that suggests PB grazing is a viable management strategy that can meet economic and biological objectives simultaneously in fire-dependent grassland ecosystems including those invaded by bluegrass in the northern Great Plains (DeKeyser et al. 2013; Scasta et al. 2016).

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