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Research Article

Assessing Large Herbivore Management Strategies in the Northern Great Plains using Rangeland Health Metrics

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ABSTRACT

Maximizing rangeland health has become a popular theme in North American grassland management. Quantifying rangeland health is particularly important when attempting to compare different management strategies ongoing in priority conservation areas, such as those within the Northern Great Plains (NGP). We investigated the response of five vegetative components of rangeland health to three large grazer management strategies on individual sites within public and private rangelands in northeastern Montana: Bureau of Land Management allotments that continuously maintained rotational cattle (*Bos taurus*) grazing, US Fish and Wildlife Service designated wilderness areas where cattle were removed, and lands managed by the American Prairie Reserve where cattle were removed and bison (*Bison bison*) were reintroduced. We then compared sites relative to historical climax plant community (HCPC) conditions—our management target. Our bison-restored site had exotic plant abundances most similar to the HCPC, and significantly lower than our other sites. Our cattle-removal site maintained litter cover most similar to the HCPC, while other sites were lower than target conditions. Overall, differences among our bison-restored and cattle-retention sites were slight relative to the HCPC. Although our treatments were represented by a single site, no single management strategy achieved all five vegetative measures of rangeland health based on HCPC targets. We observed several differences between sites that could inform future grazer management in this region. We provide a novel process to quantifiably compare rangeland health across areas with different grazer management approaches. Without quantitative assessments and long-term monitoring, assessing rangeland health can be overly subjective, and may not inform ongoing concerns surrounding grazing management in the NGP.

Index terms: bison; cattle; cattle removal; vegetation; prairie; restoration; rangeland management

INTRODUCTION

Grasslands are among the most imperiled ecosystems globally (Hoekstra et al. 2005), and debate is ongoing about how best to manage or restore rangelands across the United States (Freese et al. 2014; Fuhlendorf et al. 2018). U.S. federal agencies manage more than 149 million hectares in the Northern Great Plains (NGP) and have typically developed rangeland management plans focused on the use of domestic cattle to achieve rangeland condition targets and objectives (USBLM 2018b; USFWS 2018). However, more recently, two separate but competing management strategies have emerged with the objective of restoring NGP grasslands to historical conditions through the removal of domestic livestock or replacement of domestic livestock with American bison (*Bison bison* L.; McMillan et al. 2019). Bison reintroduction is widely popularized because of their hypothesized keystone effects on prairie diversity in the Great Plains (Knapp et al. 1999; McMillan et al. 2019). However, there are also concerns that year-round bison grazing can negatively impact grassland plant communities (Ware et al. 2014; Ranglack et al. 2015; USBLM 2018a). These concerns limit ongoing efforts to allow bison restoration projects on publicly managed rangelands in the NGP (Geddes 2018), and quantitative measures of bison effects on rangelands are lacking in the region. In particular, the impact of bison restoration on vegetation is a primary concern listed in public scoping of bison restoration projects ongoing in the NGP (USBLM 2018a). Thus, there is a need to quantitatively assess which approach best meets management objectives for NGP plant communities, and potentially broader ecosystem function.

Rangeland health monitoring is a standardized approach to assess rangeland condition using 17 indicators developed by the US Bureau of Land Management, Natural Resource Conservation Service, US Geological Survey, and the Agricultural Research Service (Pyke et al. 2002). Rangeland health is broadly defined as the status of a rangeland's air, soil, water, vegetation, and relevant ecological processes at some point in time (Pyke et al. 2002). Soil and hydrological measurements make up 10 of the 17 (59%) indicators, the rest being related to vegetative composition, abundance, and structural measurements (7 of 17; 41%). Five of these vegetative rangeland health measures are readily quantifiable (invasive plant abundance, litter abundance, plant community composition and distribution, and functional group abundance) and represent a potentially important baseline from which to monitor responses to management alternatives in the NGP. While using indicators to assess general range condition is a common monitoring approach (Karl and Herrick 2010), we know of no research that has utilized the rangeland health framework to compare how different grazer-management strategies affect progress toward a defined, quantifiable restoration objective.

Our objective for this study was to compare the five quantifiable vegetative indicators of rangeland health outlined by Pyke et al. (2002) among three management approaches in the NGP. Because quantifiable objectives or targets among our management approaches relative to rangeland condition were limited, and because historically relevant conditions are reported as desirable by some (American Prairie Reserve 2014, 2018; Freese et al. 2014), we set the rangeland health targets as the Historical Climax Plant Community (HCPC) from those reported by Bestelmeyer and Brown (2010). We first wanted to test the hypothesis that vegetation community composition and species abundances at our bison restoration site were more similar to HCPC conditions across all measures compared to cattle retention or cattle removal. We predicted that both bison restoration and cattle removal treatments would differ from the site where cattle were retained, and have comparable abundances of desirable perennial bunchgrasses, litter abundance, average functional group abundances, bare ground cover, and overall abundance of exotic plant species to the HCPC.

METHODS

Study Area

Our study took place across three study areas in the NGP region of the United States, in a portion of southern Phillips County, Montana (Figure 1), where the dominant management actions consisted of bison-restoration, sustained cattle grazing (hereafter cattle-retention), and cattle-removal. Although precise records were lacking prior to 1980, we found that it was reasonable to assume that all three study areas maintained similar grazing histories (pre-treatment) overall (McMillan et al. 2019). Our bison-restored site was located within a 12,545 ha bison reintroduction area managed by the American Prairie Reserve (APR; Figure 1). We specifically conducted our sampling in the 7092 ha Telegraph and Box Elder creek drainages, where cattle were removed in early 2004 prior to the release of 16 bison in October 2005 (M. Kohl, Utah State University, pers. comm., February 2017). This area has been managed with year-round bison grazing with the bison population growing to roughly 600 animals (including juveniles and sub-adults) by 2015 (18.75% growth per year with an average 29 imported animals per year; American Prairie Reserve 2018). Therefore, bison had grazed Box Elder for 10 y by the time of our sampling (Figure 1), during which bison grazing intensity was maintained below 0.39 Animal Unit Months (AUM) ha^{-1}. Our cattle retention site was located on land managed by the Bureau of Land Management within the 8303 ha Fourchette Creek grazing allotment adjacent to our bison-restored site (Figure 1). Grazing intensity in our cattle-retention site was maintained below a threshold of 0.33 AUM ha^{-1} from 1 May to 30 October during years 1983–2016, with no grazing outside of those months, across a five-pasture design with a variable rotation schedule (B.J. Rhodes, BLM, pers. comm., October 2015). Finally, our cattle-removal site was a 4059 ha portion of the Charles M. Russell National Wildlife Refuge, where cattle were removed in 2004. We focused our sampling within an allotment named Telegraph Creek Pasture Five where cattle had been absent for 10–11 y, located just to the south of our bison-restoration site (Figure 1). Stocking rates at each of our sites was typical for our study area, but is light relative to more productive grasslands in North America (e.g., McGranahan et al. 2012).

Site Selection, Field Sampling, and Design

To compare vegetative communities among our three sites, we first restricted our sampling to a single ESD wherein we assumed soil condition, slope, aspect, elevation, and other abiotic variables would be similar (see McMillan et al. 2019 for more detail). We selected the Shallow Clay 11–14" ESD (hereafter SC; https://esis.sc.egov.usda.gov) given that it was the most widely distributed and productive upland ESD that occurred across our three study areas, where each study area represented single treatment. We randomly selected 10 sample points within SC in each study area. We collected field data from June to August 2016, corresponding to the quantifiable vegetative measures of rangeland health (see Pyke et al. 2002; i.e., functional group composition and abundance, bare ground cover, exotic species abundance, and litter cover) at 10 randomly established 0.1 ha modified Whittaker plots within each of our three sites (McMillan et al. 2019).

Assessing Functional Group Abundance and Overall Composition Divergence from HCPC among Sites: We used two separate approaches to assess how plant functional group (i.e., perennial bunchgrasses, shrubs and subshrubs, sedges, and forbs) abundances and composition compared across our three sites relative to the HCPC. We first used a hierarchal cluster analysis to visually analyze how compositionally different our three sites were from the targeted HCPC composition (Gardener 2014). We calculated the average abundance for each species across all of our sites, and further divided each site into three functional group categories. Our functional groups followed those listed in the SC ESD: forbs, shrubs/subshrubs, or grasses/sedges. We determined the average abundances for the HCPC by averaging the range of abundances for each species listed in the SC ESD, and using those in our analyses. The resulting table was similar to a plant community data table, except the rows (normally as sites only) were further broken down into functional groupings by site. We used the R 3.3.2 package *pvclust* to calculate a dissimilarity matrix (Suzuki and Shimodaira 2006; Gardener 2014; R Core Team 2019) and then performed a hierarchical clustering analysis using 1000 bootstrap replicates (R Core Team 2019). We visually and statistically assessed the results of our hierarchical clustering by generating a dendrogram and by calculating approximately unbiased (*au*) and bootstrap probability (*bp*) values (i.e., *p*-values) to illustrate the relative relatedness of each site to the climax community (Suzuki and Shimodaira 2006; Gardener 2014). Cluster significance was considered at $\alpha = 0.05$. In our second approach, we used the *vegdist* function in the package *vegan* in R to further analyze how our sites compared based on functional group abundance by calculating the difference (i.e., the distance) between each site and the HCPC (Oksanen et al.

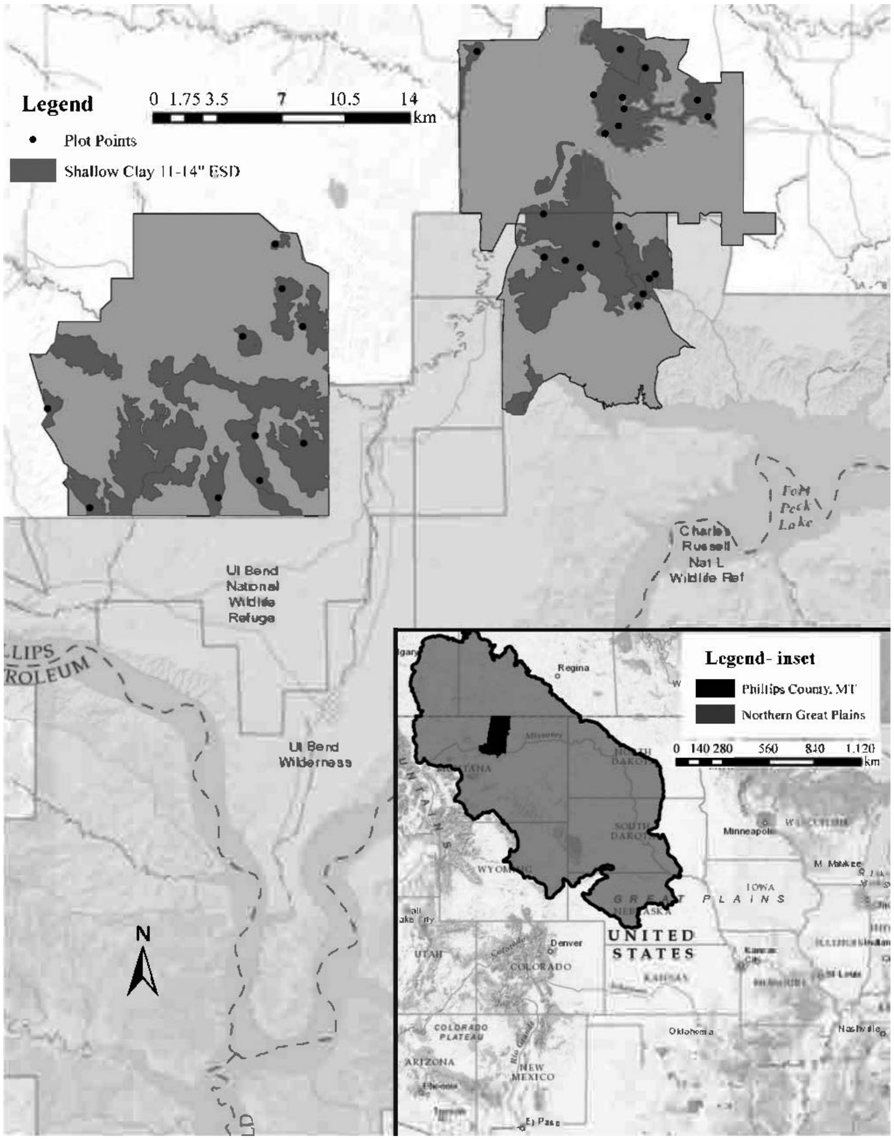

Figure 1.—Map (1) of our study area, located in Phillips County, Montana, showing the extent of the Shallow Clay 11–14" ESD within our study area, as well as the spatial arrangement of our three sites: (A) bison-restored, (B) cattle-retention, and (C) cattle-removed. Plots are represented as filled circles. Inset map (2) shows the general location of our study area (Phillips County, Montana) within the Northern Great Plains region.

2016). We then conducted a least-square mean *t*-test (LSM contrast) for each difference to assess whether the functional group abundances in each site were statistically different from the HCPC. We also directly assessed whether our three sites differed in mean functional group abundances using an ANOVA with each plot as a block within site. When significant effects were detected among sites we further explored differences using pairwise LSM contrasts.

Evaluating Differences in Litter, Exotic Plant, Bare Ground, and Perennial Bunchgrass Cover among Sites: We summed our species-level data for all species falling within either perennial bunchgrass or exotic plant groups, then calculated the mean abundance of each group per plot. While Pyke et al. (2002) specify that managers measure strictly *invasive* species abundances (native or nonnative), we broadened our assessment to include all exotic species (regardless of invasive potential) and thus attempt to provide a clearer assessment of each site relative to the HCPC. We then log-transformed our data to meet normality assumptions needed for further parametric statistical testing. We used an ANOVA with each plot as a block within site to determine whether our sites differed from one another in litter, exotic plant, bare ground, and perennial bunchgrass abundances. When we detected significant effects across our three sites, we further explored the relationship with pairwise LSM contrasts. We also used pairwise LSM contrasts to compare the mean litter, exotic plant, bare ground, and perennial bunchgrass abundances of the HCPC to the mean abundances that we observed in our three sites. We calculated the mean litter, exotic plant, bare ground, and perennial bunchgrass cover for the HCPC by calculating the mean of the abundances reported in the SC ESD. Significance for all tests was declared at $\alpha < 0.05$.

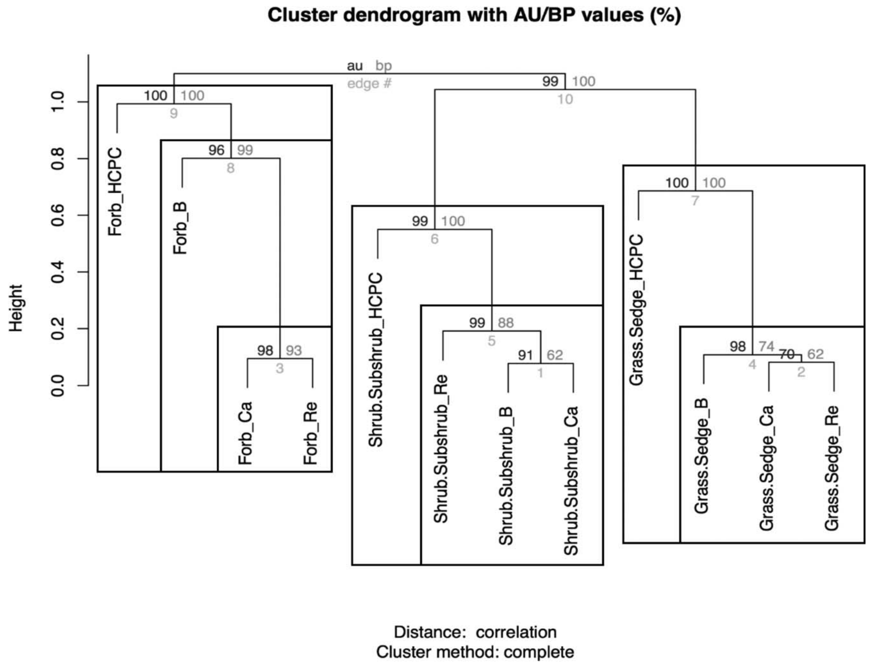

Figure 2.—Results of a hierarchical clustering analysis portrayed as a dendrogram, showing the similarity of each of our management sites from 2016—i.e., bison-restored (B), cattle-retention (Ca), and cattle-removal (Re)—to the historical climax plant community (HCPC) per functional group. Approximate unbiased (*au*) *p*-values (displayed as 1–*P* * 100) and bootstrap estimates (*bp*) are displayed above each grouping. Cluster significance was considered at $\alpha = 0.05$, and bootstrapping was performed with 1000 replications. Height represents the amount of dissimilarity between clusters, as determined following the "correlation" dissimilarity index in the "pvclust" command in the statistical software R 3.3.2. Boxes are placed around significant groupings ($p \leq 0.05$) below 1.0 height.

RESULTS

Assessing Functional Group Abundance and Overall Composition Divergence from HCPC among Sites

In contrast to our hypothesis, we did not find that vegetation within the bison-restored site had functional group compositions more closely related to the HCPC than vegetation in the cattle-removal or cattle-retention sites using hierarchical clustering (Figure 2). We found that forb composition and abundance at the bison-restored site clustered with both the cattle-removal and cattle-retention sites (bootstrap resampling, $p = 0.04$, $au = 0.96$, $bp = 0.99$; Figure 2), and with the HCPC ($p < 0.0001$, $au = 1.0$, $bp = 1.0$; Figure 2). However, the cattle-removal and cattle-retention sites were also separately, and significantly, clustered in their forb composition and abundance ($p = 0.02$, $au = 0.98$, $bp = 0.93$) and were approximately 60% and 80% dissimilar from the bison-restored site and HCPC, respectively (Figure 2). The bison-restored, cattle-removal, and cattle-retention sites were also clustered in their shrub and subshrub composition and abundance ($p = 0.01$, $au = 0.99$, $bp = 0.88$) and were approximately 20–30% similar to the HCPC (Figure 2). Similarly, the bison-restored, cattle-removal, and cattle-retention sites were significantly clustered with each other in terms of grass and sedge composition and abundance ($p = 0.01$, $au = 0.98$, $bp = 0.74$) and were approximately 15–20% similar to the HCPC (Figure 2).

Following our second approach, we did not find that native functional group abundances in our bison-restored site differed from others (shrubs and subshrubs: $F = 1.49$, $p = 0.24$, df = 2; forbs: $F = 0.48$, $p = 0.63$, df = 2), except for grasses and sedges ($F = 6.19$, $p < 0.01$, df = 2). We observed that mean native grass and sedge abundances in both the bison-restored ($t = -1.73$, $p = 0.12$, df = 19) and cattle-retention sites were similar to those in the HCPC ($t = 0.82$, $p = 0.12$, df = 19), while the cattle-removal had 14% higher mean native grass and sedge abundances that differed from HCPC ($t = 3.54$, $p = 0.0064$, df = 19; Figure 3). While we did not observe a significant difference in forb abundance among our three sites, only the bison-restored site had forb abundances statistically similar to the HCPC ($t = 1.80$, $p = 0.10$, df = 19), while the cattle-retention site was 4% higher ($t = 3.01$, $p = 0.015$, df = 19) and the cattle-removal site was 7% higher ($t = 3.01$, $p = 0.015$, df = 19) in mean native forb abundances compared to the HCPC (Figure 3). Mean native shrub and subshrub abundances were approximately 12% higher in both the bison-restored ($t = 3.64$, $p = 0.0054$, df = 19) and cattle-retention sites ($t = 5.00$, $p = 0.0007$, df = 19), and was approximately 24% higher in the cattle-removal site than those expected from the HCPC ($t = 5.84$, $p < 0.001$, df = 19; Figure 3).

Evaluating Differences in Litter, Exotic Plant, Bare Ground, and Perennial Bunchgrass Cover among Sites

Our cattle-removal, cattle-retention, and bison-restored sites significantly differed in litter cover ($F = 9.73$, $p < 0.001$, df = 27,2). The cattle-removal site contained 31% and 23% higher litter cover compared to the cattle-retention (LSM, $t = 4.25$, $p = 0.0001$, df = 27; Figure 4) and bison-restored ($t = 3.15$, $p = 0.002$, df = 27) sites, respectively, and did not statistically differ from

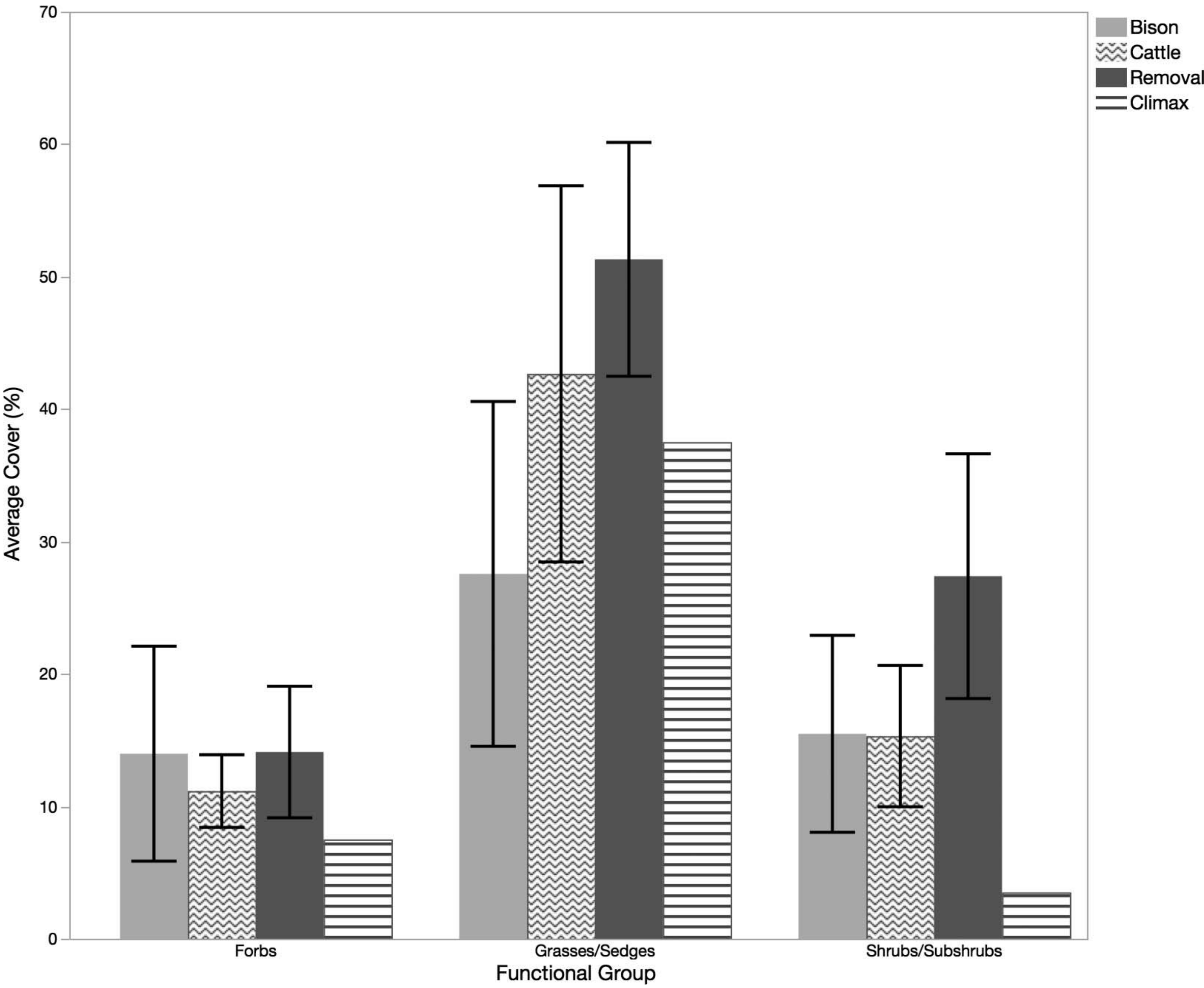

Figure 3.—The mean abundance (Cover %) of forbs, grasses and sedges, and shrubs and subshrubs detected within our bison-restored (Bison), cattle-retention (Cattle), and cattle-removal (Removal) sites from 2016. Also depicted is the expected mean forb, grass and sedge, and shrub and subshrub abundance within the historical climax plant community (Climax) reported within Montana's Shallow Clay 11–14" Ecological Site Description. Error bars represent a 95% confidence interval of the mean ($\alpha = 0.05$).

the HCPC ($t = -0.63$, $p = 0.54$, df $= 9$). However, the bison-restored and cattle-retention sites did not differ in litter cover ($t = -1.10$, $p = 0.14$, df $= 27$). Furthermore, both the cattle-retention ($t = 8.41$, $p < 0.001$, df $= 9$) and bison-restored ($t = 3.14$, $p = 0.011$, df $= 9$) sites had significantly lower litter cover overall (28% and 20% lower, respectively) than is predicted to occur in the HCPC.

We found that the bison-restored, cattle-removal, and cattle-retention sites significantly differed in exotic plant abundances ($F = 4.23$, $p = 0.025$, df $= 27, 2$). In support of our hypothesis, the bison-restored site had lower exotic plant abundances than the cattle-retention (LSM, $t = 1.80$, $p = 0.042$, df $= 27$) and cattle-removal sites ($t = 2.88$, $p = 0.0039$, df $= 27$; Figure 4), and had closer exotic plant abundances to our HCPC management target ($\bar{x} = 4.95\%$; Table 1) than the other sites. Specifically, the cattle-retention and cattle-removal sites contained 18.95% higher ($\bar{x} = 23.9\%$ cover) and 11.05% higher ($\bar{x} = 16\%$ cover) exotic plant abundances than the bison-restored site, respectively (Figure 4). The cattle-removal and cattle-retention sites were not significantly different in exotic plant abundances ($t = 1.08$, $p = 0.29$, df $= 27$; Figure 4).

Although we did not detect differences among the three sites in bare ground cover ($F = 0.80$, $p = 0.46$, df $= 22, 2$), only the bison-restored site did not significantly differ from the HCPC ($t = -0.36$, $p = 0.73$, df $= 4$). Both the cattle-retention ($t = -2.47$, $p = 0.036$, df $= 9$) and cattle-removal ($t = -2.33$, $p = 0.045$, df $= 9$) sites had 6–7% lower bare ground cover than expected within the HCPC (Figure 4).

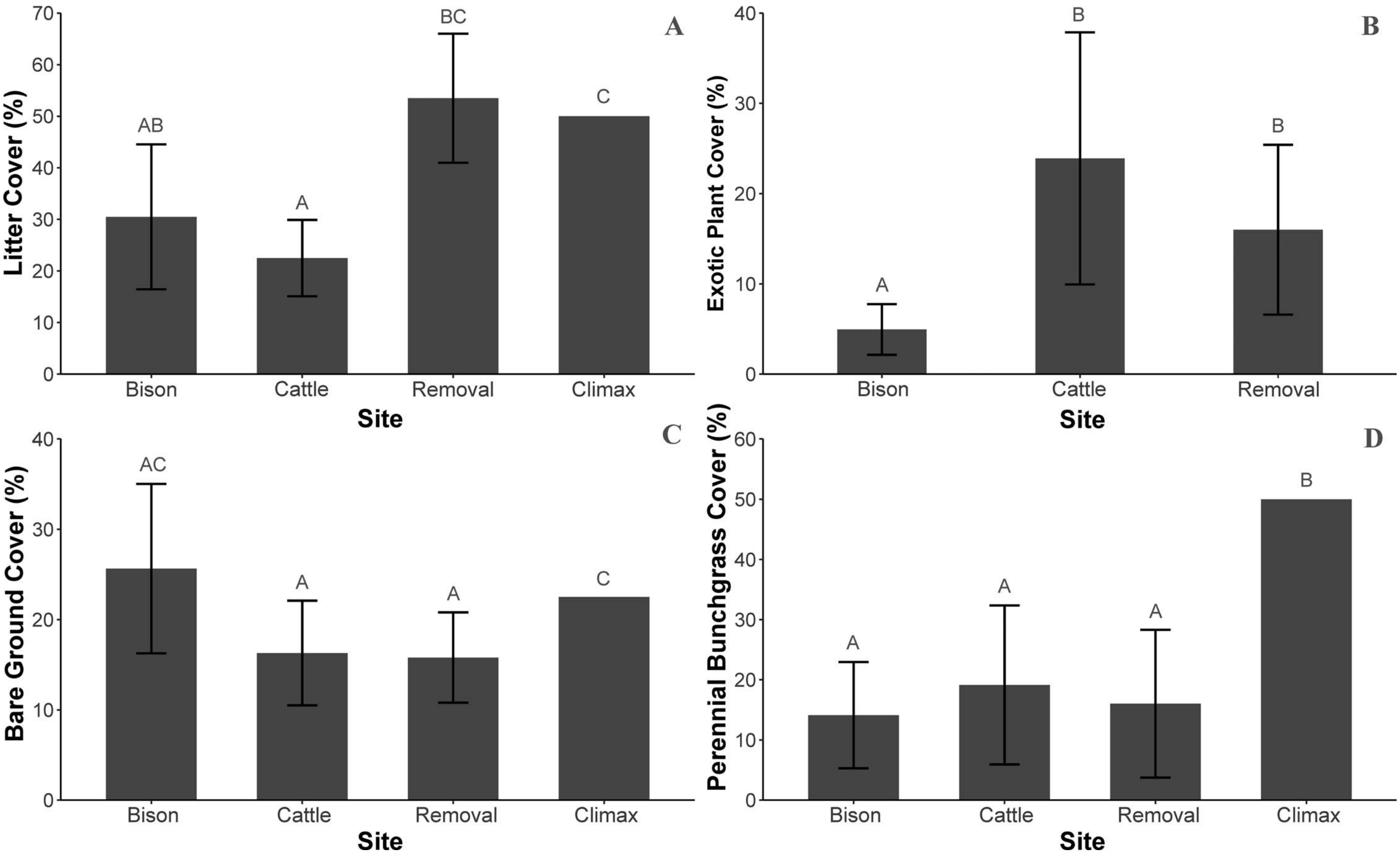

Figure 4.—The mean percent litter cover (A), percent exotic plant cover (B), percent bare ground cover (C), and percent perennial bunchgrass cover (D) for our bison-restored, cattle-retention, and cattle-removal sites from 2016. Also depicted is the expected mean percent litter, bare ground, and perennial bunchgrass cover for the historical climax plant community reported within Montana's Shallow Clay 11–14" Ecological Site Description (A, C, and D). Error bars represent a 95% confidence interval of the mean ($\alpha = 0.05$). The historical climax plant community lacks exotic species (i.e., cover = 0%), and thus is not represented in panel B of this figure. Letters above the bars represent significant differences between each of our sites and the HCPC, calculated using pairwise LSM contrasts.

We did not find higher abundances of perennial bunchgrasses in the bison-restored or cattle-removal sites compared to the cattle-retention site ($F = 0.25$, $p = 0.78$, df = 27, 2). All of the three sites had lower perennial bunchgrass abundance compared to the HCPC, with both the bison-restored ($t = -5.38$, $p = 0.0004$, df = 9) and cattle-removal sites being 6% lower ($t = -5.50$, $p = 0.0004$, df = 9), and the cattle-retention site being 5% lower ($t = -4.18$, $p = 0.0024$, df = 9) than the HCPC (Figure 4).

DISCUSSION

Our results suggest that if HCPC conditions are used as a rangeland health target, no individual large herbivore management strategy fully achieved a predicted HCPC state 10 y post-treatment in the SC ESD. Further, bison restoration in the short term may only be marginally more effective at moving toward rangeland health targets within the SC ESD relative to continued management with cattle. This finding adds to an expanding line of evidence that suggests bison restoration effects are likely highly context-specific and potentially take extended periods of time to occur (Knapp et al. 1999; Towne et al. 2005; Fuhlendorf et al. 2009; Allred et al. 2011; Ware et al. 2014; McMillan et al. 2019). Although our treatments were only represented by a single site, at a minimum our findings suggest that bison restoration is not moving vegetation away from HCPC targets of rangeland health within the SC ESD. Thus, our study provides an additional context where bison restoration may not negatively affect grassland community dynamics (McMillan et al. 2019) and rangeland health (this study) compared to cattle retention.

Two of our assessed management approaches had only been recently employed by the time of our study (~10 y), and it is possible that significant differences among our three sites may become more apparent through time as vegetative changes in semi-arid environments often occur across long time horizons (Augustine et al. 2017; Porensky et al. 2017). However, our data suggest that some measures are likely to reflect differences in large grazer management strategies more quickly than others. For example, that litter cover was much higher in the cattle-removal site than those in the other two sites suggests that this measure is one of the first to respond to large herbivore management, and further supports evidence reported elsewhere in the Great Plains that grazing disturbances work in part (often with others; e.g., fire) to regulate litter accumulation, decom-

Table 1.—The subset of rangeland health indicators and the quantitative measurement we used for each from Pyke et al. (2002), along with a summary of the averages (± SE) found in our plots from 2016 within bison-restored (Bison), cattle-retention (Cattle), and cattle-removed (Removal) sites compared with our management target. Each indicator we quantitatively measured is shown along with the corresponding management target (i.e., the historical climax plant community; HCPC) for the ecological site description (ESD) labeled Shallow Clay 11–14" (obtained via https://esis.sc.egov.usda.gov/). Significance was measured at $\alpha = 0.05$.

Quantitative measurement	HCPC target	Bison (mean %; ± SE)	Cattle (mean %; ± SE)	Removal (mean %; ± SE)
Percent bare ground	15–30%	26 (4)**	16 (3)	15 (2)
Percent cover	70–90% grass and sedge:			
	•10–20% sod forming	–	–	–
	•40–60% bunchgrass	14 (4)	19 (6)	16 (5)
	•3–7% sedges	–	–	–
	15% shrub and subshrub	–	–	–
	1–5% forb	–	–	–
Percent composition by structural or functional group, and group richness	Grasses and sedges dominant (40% of total richness)	28 (6)**	43 (6)**	51 (4)
	Forbs sub-dominant (5–10%)	13 (4)**	10 (1)	12 (2)
	Shrub and subshrubs sparse (1–5%)	15 (3)	15 (2)	27 (4)
Percent litter cover	40–60% litter cover	30 (6)	22 (3)	53 (5)**
Percent cover of exotic plant species	<1% of canopy cover	5 (1)	24 (6)	16 (4)

** Not significantly different from HCPC ($p > 0.05$).
– Not measured.

position, and nutrient cycling in temperate grasslands (Knapp et al. 1999; Fuhlenforf and Engle 2004; Anderson 2006). Similarly, exotic species abundance was the only measure we observed to support our hypothesis that bison restoration moves plant communities toward HCPC. Thus, our findings may suggest that exotic plants are more likely to respond quickly to large herbivore management than other functional groups of plant species.

Our study also only considered a subset of conditions likely to occur throughout our study area, and further research should expand to examine the ability of each management strategy across long temporal scales, varying ESDs, and rangeland management practices (e.g., stocking rate, fire management) in meeting quantifiable restoration or management objectives. Further, since the bison herd we studied underwent growth between the time of this study and the initial reintroduction, the relevance of our results are limited to sites where reintroductions are relatively recent. We recommend that future studies address how sustained grazing by bison, and perhaps also stocking rate, may impact rangeland health relative to other management strategies. Our study area lacked detailed management records prior to 1980, so it is possible that some of our results could reflect unaccounted for historical variance in management among our sites during the homesteading (1900–1920) and post-homesteading eras, and thus a need for further replicated study in the future. Lastly, we only focused on vegetative measures of rangeland health, and other rangeland health measures (potentially along with explicit manager-specific targets that differ from HCPC conditions) should be incorporated in future long-term monitoring.

We provide a novel approach to quantitatively assess and monitor rangeland plant communities between differing livestock management strategies by integrating portions of the rangeland health indicator framework (Pyke et al. 2002) with predicted historical climax plant community conditions as defined by an area's ecological site description (Bestelmeyer and Brown 2010). While we did not observe across-the-board differences in vegetative rangeland health measures among our treatments, this approach (particularly when expanded across management and environmental conditions) may serve as an example of how adaptive management (Holling 1978; Holling and Meffe 1996) can inform rangeland conservation and restoration projects in the Northern Great Plains. Without such quantitative assessments linked with long-term monitoring, assessing rangeland health can be overly subjective, and may not directly inform ongoing concerns surrounding grazing management in the NGP.

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