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Montane Meadow Plant Community Response to Livestock Grazing

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Abstract We examined long-term (10 years) meadow plant community responses to (1) livestock grazing under riparian grazing utilization limits; (2) suspension of livestock grazing; and (3) meadow site wetness and precipitation on the Inyo National Forest, California. Observed trends in meadow plant species richness, diversity, and frequency of soil stabilizing species were not significantly different between grazed ($N = 16$) and non-grazed ($N = 9$) study sites ($P > 0.12$ in all cases). Modest increases in richness and diversity were observed over the study period, but frequency of soil stabilizing species was constant. These results suggest that riparian conservation grazing strategies implemented during the study period neither degraded nor hampered recovery of meadow plant community conditions relative to non-grazed conditions. Meadow site wetness was negatively correlated to richness ($P < 0.01$) and diversity ($P < 0.01$), but was positively correlated to soil stabilization ($P = 0.02$). Precipitation was not a significant predictor for plant community responses.

Keywords Sierra Nevada · Riparian standards and guidelines · United State Forest Service · Public lands · Inyo National Forest · Kern Plateau

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Introduction

Livestock grazing on public lands is a controversial issue across the western United States, and policy-makers and managers are progressively charged with balancing diverse societal goals on these national lands (Armour et al. 1991; Beschta et al. 2013; Brunson and Steel 1996; Fleischner 1994). Livestock grazing on mountain meadows in California's Sierra Nevada began during the mid-1800s (Dull 1999; Odion et al. 1988). Since the early 1900s, the United States Forest Service (USFS) has administered this land-use via permit-based grazing allotments (Ratliff 1985). Mountain meadows on these public lands provision productive, high quality summer forage for local ranches—at a time when low-elevation annual grasslands have entered summer drought and have low forage nutritive quality (Huntsinger et al. 2010; Sulak and Huntsinger 2002). These meadows, which represent less than 10 % of the Sierra Nevada (Ratliff 1985), also contribute many other ecosystem functions and services to the forest landscape, including enhanced biodiversity, carbon sequestration, and flood attenuation (Hammersmark et al. 2008; Kuhn et al. 2011; Norton et al. 2011).

The mechanisms by which livestock can degrade meadow function have been well documented (Fleischner 1994; Belsky et al. 1999; Trimble and Mendel 1995)—excessive grazing can destabilize wetland areas by (1) reducing plant vigor, reproductive capacity, and competitiveness; and (2) triggering shifts in meadow plant communities from wetland to upland species. These functional changes in plant community attributes can lead to reduced rooting mass and reduced soil stability, which impact hydrologic function, and resistance to soil erosion (Kauffman and Krueger 1984; Ratliff 1985; Kleinfelder et al. 1992; Dwire et al. 2004; Micheli and Kirchner 2002;

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Manning et al. 1989). In a recent review, George et al. (2011) found strong evidence that riparian and meadow resources can be protected through conservation strategies controlling the intensity, timing, and spatial distribution of livestock grazing.

In this paper, we report on analyses of plant community data collected between 2001 and 2010 at 25 montane meadow monitoring sites across four grazing allotments on the Kern Plateau region of the Inyo National Forest (INF), California. During the latter part of the nineteenth century, meadows on the Kern Plateau experienced heavy livestock grazing—to the detriment of plant communities and hydrologic function (Dull 1999). During the 1980s and early 1990s, livestock grazing management in place on the Kern Plateau was identified as a driving impediment to meadow restoration and aquatic species conservation (Odion et al. 1988; Knapp and Matthews 1996; Matthews 1996). These concerns played a role in the suspension of grazing from the Whitney and Templeton grazing allotments in 2001. Grazing continued on the adjacent Monache and Mulkey allotments under new riparian grazing utilization standards that limited meadow vegetation consumption and stream bank disturbance by livestock. The objectives of this study were to (1) examine long-term (10 years) meadow plant community responses under grazed and non-grazed conditions; and (2) examine abiotic factors—specifically, meadow site wetness and precipitation (Allen-Diaz 1991; Dwire et al. 2004; McIlroy and Allen-Diaz 2012)—potentially driving plant community responses.

Materials and Methods

Study Area

This study included 25 long-term meadow plant community monitoring sites located across four grazing allotments on the Kern Plateau (36°14′N latitude and 118°15′W longitude), Inyo National Forest (INF) in the southeastern Sierra Nevada (Fig. 1). Study allotments ranged in elevation from 2,180 to 2,960 m (Table 1). Thirty-year mean annual precipitation (1981 through 2010) across the study allotments ranged from 44 to 52 cm, with the majority of precipitation falling as snow between December and April. The landscape is a mix of montane meadows, rock outcrops, and montane forests dominated by *Pinus contorta* ssp. *murrayana*, *Pinus balfouriana*, and *Pinus jeffreyi.* Meadow plant communities commonly include *Carex* spp., *Juncus* spp., *Trifolium* spp., *Poa pratensis*, *Deschampsia cespitosa*, and *Agrostis idahoensis*. Mean herbaceous meadow vegetation cover across study sites was 94 %. Relative frequency of annual, perennial grass/grass-like, and forb species was 4, 33, and 54 % across study sites, respectively.

Grazing Management

Similar to most mountain public grazing lands throughout the western United States, the study allotments were historically grazed by domestic cattle and/or sheep during the summer growing season. For the decade 1991 through 2000, which preceded the meadow plant community monitoring period examined in this paper (2001 through 2010), all study allotments were grazed by commercial cow–calf pairs (Table 1). For the decade 2001 through 2010, the Templeton and Whitney allotments received no grazing ($N = 9$ non-grazed study sites). Grazing continued on the Monache and Mulkey allotments ($N = 16$ grazed study sites) during the period 2001 through 2010, subject to annual riparian grazing standards that (1) restricted herbaceous vegetation biomass consumption; (2) required minimum residual herbaceous vegetation heights; (3) restricted browse on riparian willow species (*Salix* spp.); and (4) restricted livestock hoof damage to streambanks (Table 1) (Clary and Leininger 2000; Clary and Webster 1990; Hall and Bryant 1995). Grazing management utilized to achieve riparian standards included improved livestock distribution and rotational grazing with herding, and annually variable timing of rest and grazing for meadows across the allotments.

Data Collection

This was a-10 year longitudinal survey of 25 plant community monitoring sites established between 1999 and 2001 at key meadow grazing areas across four grazing allotments. Key meadow grazing areas selected for monitoring were sites preferentially grazed by livestock due to relatively high forage value and proximity to drinking water. Three parallel, 20 m long monitoring transects were permanently established 5 m apart in a location representing the dominant plant community in each key meadow grazing area (i.e., study site). Along each transect, twenty 0.01 m^2 quadrats were established at 1 m intervals for a total of 60 quadrats per study site. All plant species rooted within each 0.01 m^2 quadrat were identified following Baldwin et al. (Baldwin et al. 2012). Plant community composition based on relative species frequency was determined at each study site at sample years 1, 5, and 10. Sample year 1 represented initial plant community structure and composition at the time grazing was suspended on 9 study sites and new riparian grazing standards were being introduced on the other 16 study sites (Table 1).

Plant species relative frequency data for each sample year was used to calculate the number of species (richness)

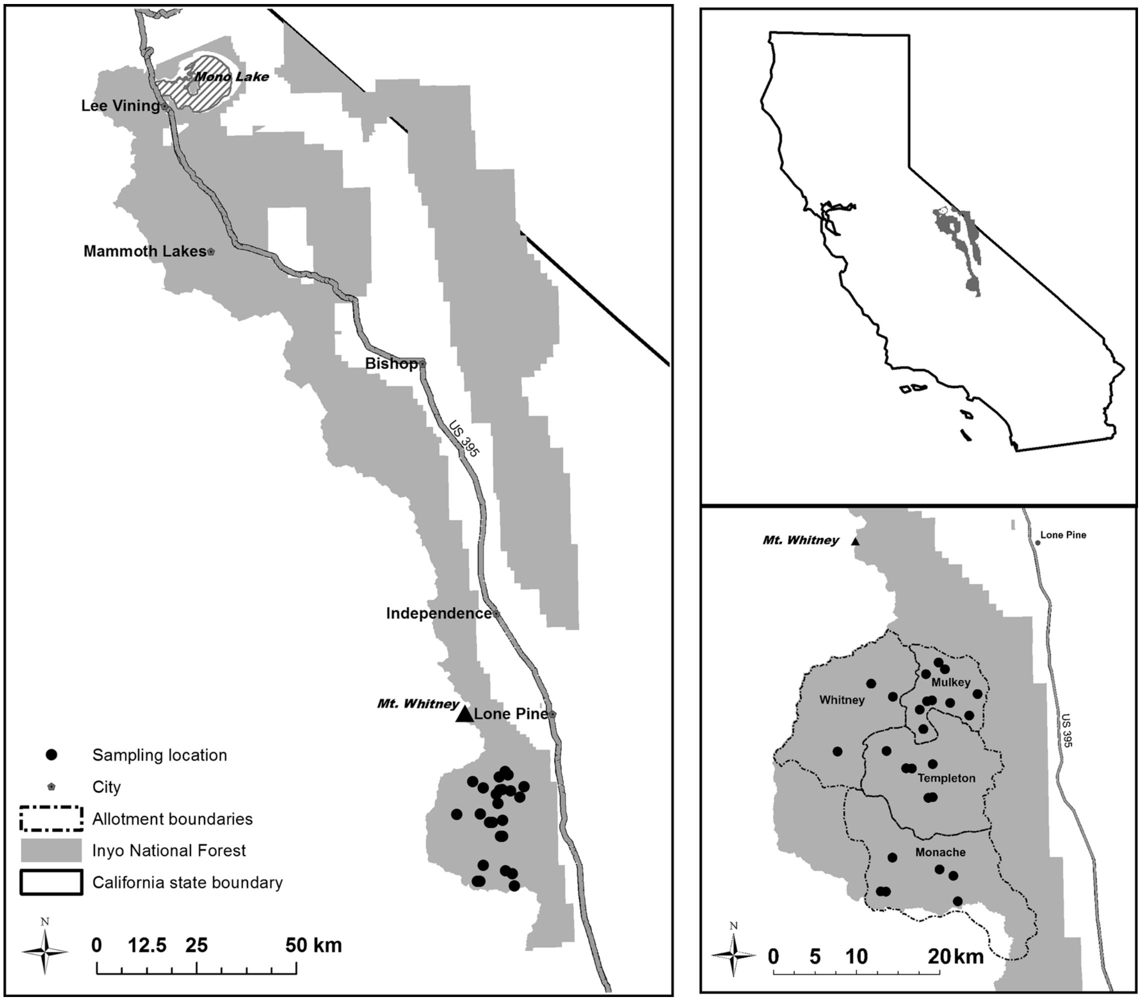

Fig. 1 Long-term meadow plant community study sites and allotments on the Inyo National Forest

and Shannon–Wiener index (diversity; H′) for each study site (McCune and Grace 2002). Each plant species was assigned to a soil stabilization category following Winward (2000), Burton et al. (2010), and Baldwin et al. (2012). Soil stabilization score (1–10) was calculated for each study site each sample year based on relative frequency of species in stabilization capacity categories (1–2.9 = very low, 3–4.9 = low, 5–6.9 = moderate, 7–8.9 = high, 9–10 = excellent) following Winward (2000). This metric heavily weights perennial, deeply rooted, clonal grass-like species commonly dominant in wetter meadows. We calculated the ratio of perennial grass-like species to forb species (PGL:F) for sample year 1 at each study site to serve as a proxy for meadow site wetness. Increased relative frequency of perennial grass-like species (i.e., increased PGL:F) is indicative of wetter (i.e., shallow, persistent water table) meadow site conditions (Allen-Diaz 1991; McIlroy and Allen-Diaz 2012). Five-year mean annual water year (October through September) precipitation was estimated for each study site for the periods prior to sample years 1, 5, and 10 using Zonal Statistics (ESRI 2010) and raster imagery of monthly precipitation totals (PRISM 2013).

Data Analysis

We used linear mixed model (LMM) and generalized linear mixed model (GLMM) regression analyses to test for relationships between sample year (1, 5, 10), livestock grazing (grazed with riparian standards, non-grazed), precipitation (5-year mean annual precipitation prior to each sample year), meadow site wetness (sample year 1 PGL:F), species richness and diversity, and soil stabilization score. LMM, based on the normal probability distribution function, was used for regressions with diversity, and soil stabilization score as dependent variables. GLMM, based on the Poisson probability distribution function, was used for regressions with species richness as the dependent variable. For each regression analysis, we used a backward stepwise procedure with the following fixed effects included in each

Table 1 Study site numbers, topographic, precipitation, livestock grazing characteristics, and riparian grazing utilization standards for grazing allotments

	Allotment			
	Monache	Mulkey	Templeton	Whitney
No. study sites	6	10	6	3
Area (ha)	22,100	7,700	17,700	18,200
Mean elevation (m)	2,660	2,640	2,180	2,960
5-year mean annual P prior to sample year 1 (cm)	55	47	51	65
5-year mean annual P prior to sample year 5 (cm)	55	44	42	44
5-year mean annual P prior to sample year 10 (cm)	49	42	43	46
30-year mean annual P (1981–2010)	52	44	45	52
Years grazed 2001–2010	10	10	0	0
Mean annual no. cow–calf pairs 1991–2000	738	241	505	325
Mean annual no. cow–calf pairs 2001–2010	747	235	0	0
Herbaceous grazing utilization standard (%)[a]				
Prior to 1996	60–70	50–65	No standard	50–65
1996–2003	<35	<35	No standard	50–65
2004–2010	<35	<35	No grazing	No grazing
Willow grazing utilization standard (%)[b]				
Prior to 1996	No standard	No standard	No standard	No standard
1996–2003	40	40	40	40
2004–2010	20	20	No grazing	No grazing
Streambank damage standard (%)[c]				
Prior to 1996	20	20	20	20
1996–2003	10	10	10	10
2004–2010	10	10	No grazing	No grazing
Stubble height standard (cm)[d]				
Prior to 1996	No standard	No standard	10	No standard
1996–2003	No standard	No standard	10	No standard
2004–2010	No standard	No standard	No grazing	No grazing

[a] Maximum allowable herbaceous vegetation consumption (% annual biomass production)

[b] Maximum allowable browse on riparian willow species (% annual leader growth)

[c] Maximum allowable streambank trampling and physical hoof damage (% streambank length damaged)

[d] Minimum required residual herbaceous vegetation height (cm)

initial model: (1) sample year, to determine if plant community trended over the study period, (2) grazing by sample year interaction, to determine if plant community trends differed between study sites grazed with riparian standards and non-grazed study sites; (3) PGL:F, to test for meadow site wetness influence on plant community responses; and (4) 5-year mean annual precipitation prior to each sample year, to test for precipitation effects on plant community responses. Wald-type tests were used to determine significant fixed effects, with a $P < 0.05$ required for inclusion in final models (Rabe-Hesketh and Skrondal 2008). Allotment identity and study site identity were specified as hierarchal random effects, with study site nested within allotment. Standard diagnostics were used to check assumptions associated with these analyses. All analyses were conducted with the STATA/SE 11.1 statistical package (StataCorp 2013). To examine changes in overall plant community composition, we used non-metric multidimensional scaling (NMDS). NMDS scores were calculated using a Euclidean dissimilarity matrix (McCune and Grace 2002). We examined 6-D through 1-D solutions, and selected the optimal number of dimensions via a plot of final stress versus number of dimensions (i.e., scree plot; McCune and Grace 2002). We also used multi-response permutation procedures (MRBP) to test significance of plant community differences, blocking observations by study site (McCune and Grace 2002). Analysis was conducted in the R software environment using the metaMDS routine from the vegan package (Oksanen et al. 2007; R Development Core Team 2010).

Table 2 Results of mixed model regression analyses to test for relationships between meadow plant community, sample year, livestock grazing, precipitation, meadow site wetness, species richness and diversity, and soil stabilization score

Fixed effect	*P* value		
	Species richness	Shannon diversity index	Soil stabilization
Site wetness	<0.01	<0.01	0.02
Year 1	–	–	–
Year 5	<0.01	<0.01	0.36
Year 10	<0.01	0.04	0.81
Grazing *X* year	0.86	0.56	0.12
Precipitation	0.24	0.44	0.59

P value for year is for the comparison of sample year 5 and 10 to sample year 1 for each response variable

Table 3 Meadow plant species richness and diversity, and soil stabilization score at sample years 1, 5, and 10 pooled across all sample sites

Response variable	Year	Mean (1 S.E.)		
		Overall	Grazed	Non-grazed
Species richness	1	14 (0.9)	13 (1.4)	15 (1.1)
	5	19 (1.4)*	20 (2.0)	19 (1.9)
	10	18 (1.1)*	17 (1.5)	18 (1.5)
Shannon diversity index	1	2.1 (0.08)	2.1 (0.12)	2.2 (0.08)
	5	2.4 (0.09)*	2.4 (0.12)	2.4 (0.12)
	10	2.3 (0.08)*	2.3 (0.10)	2.2 (0.12)
Soil stabilization	1	5.1 (0.2)	5.2 (0.3)	4.9 (0.2)
	5	4.9 (0.2) *ns*	5.1 (0.2)	4.6 (0.3)
	10	5.0 (0.2) *ns*	4.9 (0.3)	5.2 (0.3)

Asterisk (*) and *ns* indicate sample year 5 and 10 mean is significantly ($P \leq 0.04$ in all cases) or not significantly ($P \geq 0.36$ in all cases) different from sample year 1 for each response variable, respectively

Results

Mean annual precipitation across all study sites from 2001 through 2010 was 46 cm, or 96 % of the 30 year average (1981 through 2010) (Table 1). The livestock grazing (grazed, non-grazed) by sample year (1, 5, 10) interaction was not a significant predictor of meadow plant species richness ($P = 0.86$), diversity ($P = 0.56$), or soil stabilization score ($P = 0.12$) (Table 2). This indicates that trends observed for these plant community metrics were not significantly different between grazed and non-grazed study sites (Table 3). Species richness and diversity were significantly higher in both sample years 5 and 10 compared to sample year 1 (Table 3). Mean soil stabilization score was constant across sample years 1, 5, and 10 ($P > 0.36$) (Table 3). Meadow site wetness (PGL:F) was negatively correlated to richness ($P < 0.01$) and diversity ($P < 0.01$), but was positively correlated to soil stabilization ($P = 0.02$) (Table 2; Fig. 2). Five-year mean annual precipitation prior to each sample year was not a significant predictor for any of the plant community metrics examined in this study ($P > 0.24$) (Table 2). This is likely due to the relatively narrow precipitation range observed across the study sites (Table 1). NMDS analysis of both grazed and ungrazed sites revealed significant ($P < 0.01$) changes in overall plant community composition between years 1 and 10 of the study period (Fig. 3), which supports previous results of the species richness and diversity analyses (Table 3).

Discussion

Livestock Grazing and Plant Community

Trends in meadow plant species richness, diversity, and soil stabilization were not different between study sites excluded from grazing and sites grazed with riparian standards in place. Regardless of grazing management, species richness and diversity increased by 4 and 0.2 (on a log scale) over the 10 years of study, respectively—and there was a significant shift in overall plant community composition between sample years 1 and 10. The modest increases in richness and diversity were driven by increases in native, perennial forb species. Soil stabilization scores were constant over the study period under both grazing regimes. Overall, soil stabilization scores were at the lower end of Winward's (2000) moderate stability class.

Riparian response to livestock exclusion has been shown to vary substantially depending upon initial site conditions and plant community composition, as well as livestock grazing management pre- and post-exclusion (George et al. 2011; Sarr 2002). For example, previous studies have reported increased (Bowns and Bagely 1986), decreased (Ratliff 1985; Holland et al. 2005), and static (Kauffman et al. 1983; Lucas et al. 2004) species richness and diversity following livestock exclusion. Certainly, positive meadow plant responses have been well documented following removal of relatively heavy riparian grazing (Kondolf 1993; Kauffman et al. 1983; Schulz and Leininger 1990; Green and Kauffman 1995; Leege et al. 1981). Meadow vegetation recovery from past grazing management in the study area (Dull 1999; Odion et al. 1988) appears to be gradual, as has been previously observed in this area (Kondolf 1993) and others (Sarr 2002).

Our results indicate that livestock exclusion did not lead to greater rates of meadow plant community recovery compared to grazing management to achieve riparian

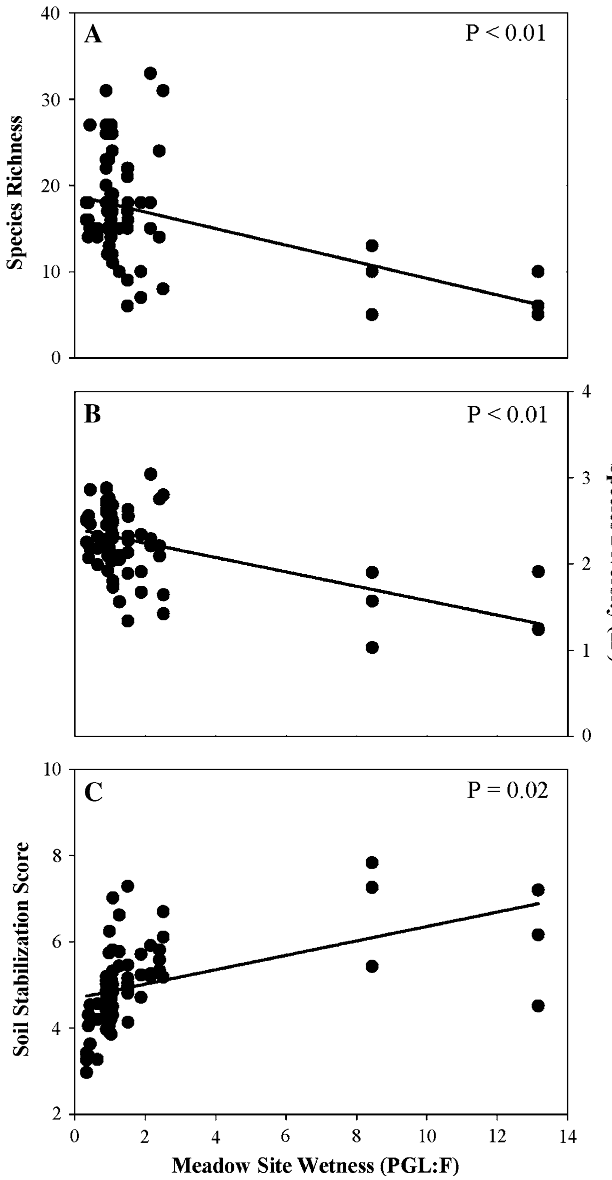

Fig. 2 Scatter plot and predicted relationships between meadow site wetness (calculated as the ratio of perennial grass-like species to forb species (PGL:F) in sample year 1) and **a** plant species richness, **b** plant species diversity (Shannon–Wiener index; H′), and **c** soil stabilization score

grazing standards. Almost 30 years ago on the Templeton Allotment, Odion et al. (1988) found significantly greater herbaceous plant densities inside a two-year old livestock exclosure compared to adjacent grazed areas. The riparian grazing regimes these authors studied at that time resulted in 75 % browse on annual willow growth (*Salix* spp.), a considerably different riparian grazing regime than the current maximum allowable 20 % browse on riparian woody species (Table 1). Implementation of riparian grazing standards over the past two decades appears to have established grazing regimes that are currently much different than those studied in the 1980s and early 1990s.

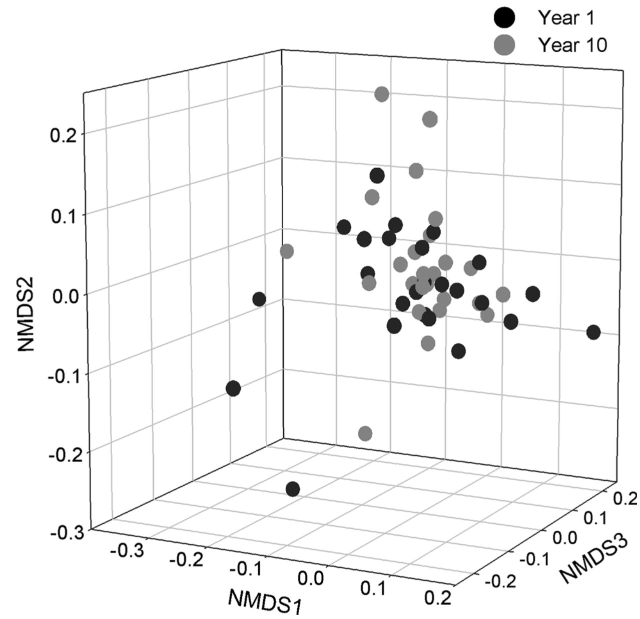

Fig. 3 NMDS ordination of all study sites based on plant community composition. Plant community composition was significantly different ($P < 0.01$) between sample years 1 and 10

Our findings support the adoption of a policy of riparian grazing utilization limits on these public grazing lands (Clary and Webster 1990; Clary and Leininger 2000; Hall and Bryant 1995). Our management scale results agree with Clary (1999), who experimentally demonstrated that grazing management implemented to achieve riparian grazing standards was compatible with stream and riparian vegetation enhancement. He found comparable recovery trajectories for treatments of livestock exclusion, light grazing (20 % consumption of herbaceous vegetation, 15 cm herbaceous vegetation height), and moderate grazing (35 % consumption of herbaceous vegetation, 10 cm herbaceous vegetation height).

Meadow Site Wetness and Plant Community

Plant species richness and diversity decreased with increasing meadow site wetness, while stability increased along this same gradient (Table 2; Fig. 2). These findings agree with previous research demonstrating the linkages between meadow water table–soil moisture dynamics and meadow plant communities (Allen-Diaz 1991; Lowry et al. 2011; Castelli et al. 2000; Loheide and Gorelick 2007). Species richness is negatively correlated with depth to the water table (Dwire et al. 2006), and wet meadows are generally less plant species rich than drier sites (Dwire et al. 2004; McIlroy and Allen-Diaz 2012). Perennial grass-like species tolerant of wet meadow conditions are commonly clonal and competitively dominant, resulting in reduced species richness and diversity with increased meadow wetness. The positive relationship between stability and meadow wetness is driven by the heavy weight placed on frequency of perennial, deeply rooted, clonal grass-like species commonly dominant in stable wet meadows.

Conclusions

We observed an overall moderate increase in meadow plant species richness and diversity, with no change in the frequency of soil stabilizing species. The observed trends in meadow plant community did not vary between non-grazed study sites and grazed study sites. These results suggest that livestock grazing compliant with USFS riparian grazing standards did not degrade or hamper recovery of meadow plant communities relative to livestock exclusion. Our results indicate that riparian centric grazing management to achieve limited riparian grazing is an important component of any strategy to conserve meadows on grazed national forest lands.

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