Plant Community Responses to Livestock Grazing: an Assessment of Alternative Management Practices in a Semiarid Grassland

Matthew R. Loeser
Thomas D. Sisk
Timothy E. Crews

Abstract—One of the most prevalent land-use practices in the American Southwest, and one of the most contentious issues among land-use policymakers, is the grazing of domestic livestock. In an effort to contribute scientific understanding to this debate, we have designed experiments comparing the effects of alternative grazing regimes on plant communities. In a semiarid grassland of northern Arizona, we have implemented a replicated study of four treatments: (1) low-intensity, long-duration grazing rotations; (2) high-intensity, short-duration rotations (Holistic Resource Management-style grazing); (3) very high intensity, short duration grazing (to simulate herd impact); and (4) livestock exclosure. Beginning in 1997, we conducted annual surveys of the plant communities with Modified-Whittaker plots. Preliminary results suggest that interannual variability affecting all study plots is high, and that these alternative management strategies do not have dramatic short-term effects on the plant community. Comparisons of native and exotic species richness, as well as ground cover of grasses and forbs, showed no consistent pattern due to treatment over a 3-year period. Our results suggest that the effects of alternative livestock management styles in the semiarid grasslands studied are modest, at least in the short-term, and that future plant monitoring programs would greatly benefit from a multiscale sampling design.

Introduction

Semiarid grasslands are diverse and productive ecosystems that have been subjected to anthropogenic forces for over a century. Among the many perturbations affecting these systems, the grazing of domestic livestock has been recognized as the most pervasive (Fleischner 1994). Although the effects of grazing have been studied for over a century, little consensus has been reached with regard to grazing's impact on grassland composition and productivity, due in part to the difficulty of separating the effects of

livestock from tremendous interannual variation in climatic patterns. While some studies have found grazing to be the primary determinant of plant community composition (for example, Conley and others 1992; Fuhlendorf and Smeins 1997), others have found climatic patterns to outweigh grazing effects (for example, Gibbens and Beck 1988; Herbel and others 1972). Moreover, the interactions of climatic variation and grazing disturbances may exacerbate the changes in grassland plant communities (Fuhlendorf and Smeins 1997).

Within this sphere of unpredictable climatic events, humans endeavor towards sustainable land-use practices, but in many cases we fail due to our ignorance of the complex array of forces, both natural and anthropogenic, operating in all ecosystems. Certainly the last century of fire prevention programs in the forests of the American Southwest are a testament to our efforts to eliminate or control disturbance, but as a consequence, we have placed many forest ecosystems in tenuous states where they are prone to catastrophic fire (Covington and others 1994). Scientists and public alike recognize a need to restore disturbance cycles to many natural systems. However, efforts to restore grasslands have been slow at best, and degradation by land-use activities are so acute that they have become a global conservation priority (Samson and Knopf 1996). Noss and others (1995) concluded that grasslands were among the most endangered ecosystems in the United States, and of those ecotypes that were most severely degraded, grasslands suffered areal losses that were threefold greater than any other ecosystem. Likely explanations of this decline include their widespread use for agriculture and development, and the commonly assigned low aesthetic value that has translated into a low priority for conservation (Samson and Knopf 1996).

The Southwest has nearly one million acres of arid and semiarid grasslands, but unfortunately our understanding of their ecology is fairly superficial, especially of the effect of large mammal herbivory such as domestic livestock. By some accounts the number of cattle on rangelands of the Western United States doubled, from 25.5 to 54.4 million head between 1940 and 1990 (Belsky and others 1999), affecting approximately 70 percent of the land surface in the Western States (Fleischner 1994), and 86 percent of the land in Arizona (Mayes and Archer 1982).

Surprisingly little is known about the effects of varying intensities of livestock grazing on the biological diversity of semiarid grasslands. Although many studies have examined the effects of livestock on the soils, vegetation, and birds

of riparian areas (Berendse 1999; Saab and others 1995; Trimble and Mendel 1995), this information cannot be applied to grassland systems because upland grasslands differ substantially, both in structure and function, from riparian ecosystems. In addition, Hastings and Turner (1965) have presented compelling photographic evidence that the last century has seen a transformation over large areas of the Southwest, of grasslands to woody shrublands. While ecological impacts from current livestock grazing practices are widespread and ongoing, it is important to recognize that the greatest damage from overgrazing appears to have taken place in the 1890s (Wildeman and Brock 2000).

Because livestock grazing can be conducted at varying intensities, it is likely that some management techniques alter biological diversity more than others. The standard method for measuring such effects has been to establish exclosures and conduct pairwise comparisons of species richness and ground cover (Hughes 1996). Because of the tremendous variability in responses as seen in the literature, it is exceedingly difficult for land managers to confidently proceed with grazing management plans. In addition to the confounding effects of variable climatic patterns mentioned previously, numerous criticisms have been directed at previous grazing studies, including the small size of exclosures (Fleischner 1994), a lack of standard methods for measuring grazing intensity (Fleischner 1994), changes in the behavior of small herbivores and granivores (McNaughton 1983), and poor experimental design (Stohlgren and others 1998). Stohlgren and others (1998) address many of the sampling design concerns and draw specific attention to inadequate assessment of spatial heterogeneity of landscapes. In grassland ecosystems, choosing the appropriate spatial scale at which to measure a given response variable is not trivial, yet the majority of plant diversity studies have arbitrarily chosen sample plots of $1 m^2$ or less (Stohlgren and others 1998). Collins and Glenn (1997) have demonstrated that measures of local species richness generated from a $1 m^2$ plot often correlate poorly with landscape-scale species richness. Similarly, Stohlgren and others (1998) compared common rangeland vegetation sampling techniques and found the traditional, fine-scale measurements to greatly underestimate rare or patchily distributed species.

To address many of the problems associated with previous study designs, we initiated an experiment with both spatial and temporal replication. This paper addresses three key issues in the study of livestock grazing impacts:

- Does increasing intensity of livestock grazing have a consistent effect on the ground cover and species richness of both native and exotic grasses and forbs?
- Do grazing treatments cause a predictable shift in species composition of the overall plant community?
- Can our sampling techniques and models of plant species richness be scaled accurately to reflect conditions of larger landscapes?

Methods _____

Site Description

Located at 2,160 m elevation and approximately 32 km southeast of Flagstaff in north-central Arizona, the Reed Lake field site occupies Upper Great Basin grassland (Brown 1994) at the edge of ponderosa pine (*Pinus ponderosa*) forest. The dominant perennial grasses are *Agropyron smithii* and *Elymus elymoides*. A vertisol soil occurs across the entirety of the 25-ha study site. Annual precipitation averages between 300 and 460 mm and is approximately evenly divided between winter and the July–September monsoons (Brown 1994).

Experimental Design

Meaningful application of science to grazing issues will require comparisons of the effects of actual management practices ("treatments") as well as experimental treatments designed to elucidate the relationships between grazing and important ecosystem variables. Our experimental design, replicated in time and space, consists of four treatments in three blocks on the landscape (Sisk and others 1999). The four treatments are (abbreviations shown in parentheses): (1) conventional low-intensity, long-duration grazing rotations (CON); (2) high-intensity, short-duration rotations informed by the principles of Holistic Resource Management (HRM, see Savory 1988); (3) very high-intensity, short-duration grazing to simulate herd impact (VHI); and (4) livestock exclosure (EXC).

Stocking rates and rotations for the first two treatments are determined by ranchers on adjoining pastures, while the latter two treatments are implemented on experimental plots by researchers and ranchers. Instead of manipulating small plots in a way thought to mimic range management, we are working on active ranches, where we have carefully selected matched sites in which spatially and temporally extensive treatment effects can be complemented with 1-ha EXC and VHI treatments. Of the four treatments, only the VHI treatment is not a simulation of a current grazing policy; it serves as a critical upper-end treatment representing a ranching practice that could be implemented only with open-range herding. Its primary purpose is to help identify mechanistic relationships and inform interpretation of results from all treatments.

Plant Surveys

To measure plant community patterns, we established a Modified-Whittaker plot in each treatment plot (fig. 1; Stohlgren and others 1995). The Modified-Whittaker plot samples the heterogeneity of plant communities with a higher level of accuracy then either true random sampling or the original Whittaker plot design (Stohlgren and others 1995). This sampling design involves a $1,000 m^2$ plot with nested subplots consisting of one $100 m^2$, two $10 m^2$, and ten $1 m^2$ subplots (fig. 1). Our version of the Modified-Whittaker plot differs slightly from the design proposed by Stohlgren and others (1995) in that we have placed four of the $1 m^2$ subplots within the central $100 m^2$ plot. Each Modified-Whittaker plot has been measured annually since 1997, with surveys conducted between mid-July and early September. The $1 m^2$ plots were surveyed with a 50-point frequency frame (Mueller-Dombois and Ellenberg 1974) that provided a more objective measurement of ground cover by species and a more comprehensive species list
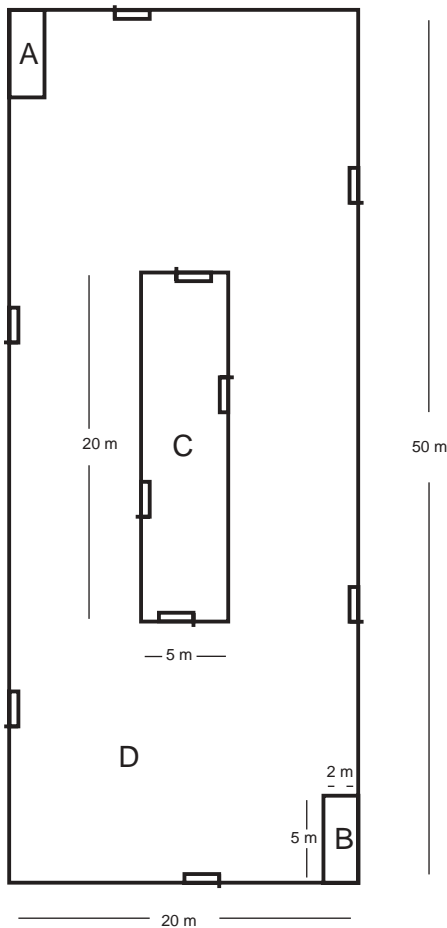
Figure 1—A schematic diagram of the version of Modified-Whittaker plot employed in this study showing multiscale plots in a nested design (see Stohlgren and others 1995).

than could be obtained with visual estimates. Only species lists were assembled for the 10 m², 100 m², and 1,000 m² plots. Plant identifications were referenced with the Deaver Herbarium located at Northern Arizona University in Flagstaff, AZ, and voucher specimens were retained in the Sisk Laboratory. Of the approximately 80 plant species found at the Reed Lake site, 15 could not be identified due to poor specimens but were considered unique species and therefore included in overall estimates of species diversity.

Grass Versus Forb—Three years of detailed plant community data have generated a wealth of information, and here we present only a subset. We calculated trends in ground cover estimates for the two primary functional groups, grasses and forbs, using data collected from the 1 m² plots over all 3 years. Data were transformed with an arcsine (square root) function prior to analysis (Sokal and Rohlf 1995), but are presented as percentages. A repeated-measures ANOVA was conducted for each functional group (Sokal and Rohlf 1995).

Native Versus Exotic—To address the issue of whether livestock grazing management techniques affect the establishment of exotic species, we calculated the number of native and exotic species in each treatment for each of the 3 years. In addition, we calculated the total ground cover of native and exotic species by treatment and year. A repeated-measures ANOVA was conducted for species richness and ground cover.

Community-Level Analysis—To determine whether treatment was having a predictable effect on overall plant diversity, we used 3 years of 1-m² subplot data to calculate three indices, as described in Magurran (1988). First, we calculated values for the Shannon-Wiener index of diversity based on the proportional abundance of species, with higher values suggesting more species and greater evenness. Second, we estimated evenness by comparing the observed diversity to the maximum potential diversity. Values fall between 0 and 1, with higher values indicating a more evenly distributed community. Third, we calculated the Simpson's index, which is weighted towards the more dominant species and is fairly insensitive to differences in rare species. All calculations were made based on Magurran (1988).

As suggested by Stohlgren and others (1998), we estimated the similarity of plant communities between treatments using Jaccard's Coefficient (signified as *J*, Krebs 1989). Jaccard's Coefficient (or index) is a standard similarity index that indicates the amount of overlap between the communities of any pair of sites. Within each replicate, we compared the CON treatment to each of the other three treatments and calculated the corresponding Jaccard's Coefficient. We chose to use the CON treatment as the "control" because it represents the most common grazing management style for Southwestern rangelands, both historically and currently (Kruse and Jemison 2000), while other treatments are more recent deviations from this management. The *J* value was then calculated for each 1,000 m² plot comparison and averaged across all three replicates. A *J* value of 1.0 would represent complete overlap of species between two sites.

Another useful product from the nested design of the Modified-Whittaker plots is the ability to generate species-area curves. Species-area curves have the very practical application of extrapolating total species richness to areas much larger then the plots in which the plant surveys were conducted. One standard variant of the species-area curve is defined as:

$$y = m(\log \text{Area}) + b$$

where y is the number of species, m is the slope of the line, and b is a constant. However, Stohlgren and others (1997) have demonstrated that a corrected species-area curve provides a far more accurate extrapolation of plant species to large areas:

$$y = (m/J) (\log \text{Area}) + b$$

where *J* is the mean Jaccard's Coefficient. We calculated the species-area curve for each treatment based on 1999 data from the 1 m², 10 m², and 100 m² plots. We then calculated the *J* value based on the species lists of each 1,000 m² plot

within a treatment and used the average J value to correct the original species area curves. The estimated species richness for an area of 3,000 m^2 was calculated using both the standard (uncorrected) species-area curve and the corrected species-area curve and compared to the observed species richness of the three 1,000 m^2 plots summed together.

The final analysis of the plant community data was conducted with a relatively new ordination technique of demonstrated utility, nonmetric multidimensional scaling (NMDS, Minchin 1987). Based upon NMDS, Minchin (1987) developed a simulation procedure that makes it possible to model patterns of variation in community composition along ecological gradients. We used species abundance data from the 1 m^2 plots of all 3 years to compare communities of each plot to one another.

Results

Grass Versus Forb Cover

If cattle graze grasses more commonly than forbs (Stuth 1991), then a moderate intensity of grazing may limit grasses from becoming the dominant ground cover. In this scenario, forbs may have higher germination success in areas where some grazing occurs because grasses aren't able to hoard resources such as light and nutrients (Collins and others 1998). We analyzed ground cover data from our 1-m^2 plots for total grass and forb cover across all treatments and years. Since the initiation of the study in 1997, grass cover appears to have increased roughly 10 percent. However, this change was not statistically significant (fig. 2; df = 3, F = 0.540, p = 0.668). In contrast, forb cover appears to be decreasing by approximately 20 percent, but again this trend is not statistically significant (fig. 2; df = 3, F = 0.697, p = 0.579). Although neither trend demonstrated clear statistical significance, both suggest changes that occurred due to yearly patterns rather than treatment effects.

Native Versus Exotic Richness and Cover

Exotic plants are generally considered to be successful colonizers of disturbed habitats, and high intensities of livestock grazing have been suggested to increase invasibility (Hobbs and Huenneke 1992). To explore this hypothesis, we first computed the total native and exotic species richness from our 1 m^2 plots for each treatment by
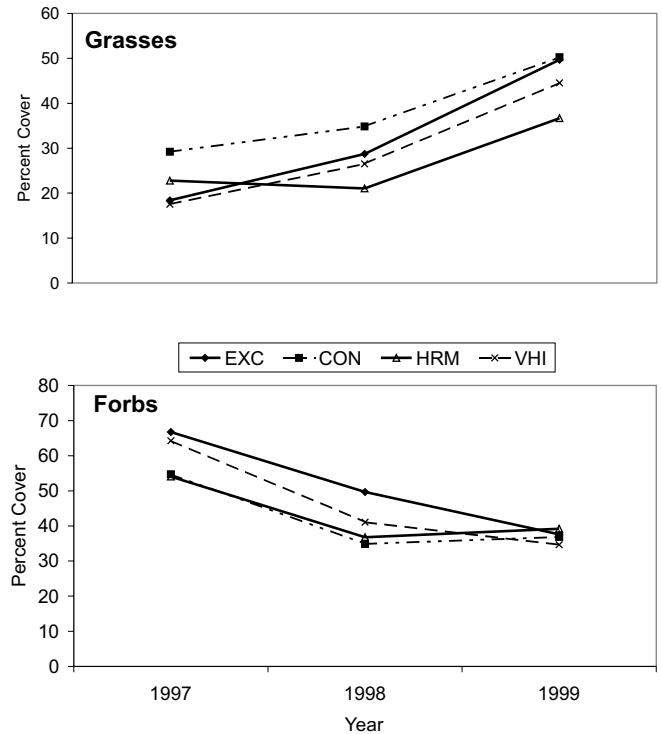
Figure 2—Percent ground cover by year and treatment for grasses and forbs as calculated from 1 m^2 subplot data. Treatment shows no significant effect for either grasses or forbs, but climatic patterns appear to be benefiting grasses to the detriment of forbs. Standard error bars are not shown because they overlap for all treatments within any given year indicating no statistically significant differences (grasses: df = 3, F = 0.540, p = 0.668; forbs: df=3, F = 0.697, p = 0.579).

year. No statistically significant differences were found within years (table 1; 1997: df = 3, F = 1.933, p = 0.165; 1998: df = 3, F = 1.274, p = 0.317; 1999: df = 3, F = 0.917, p = 0.455). Similarly, ground cover by natives and exotics from our 1 m^2 plots did not show significant differences among treatments within years (table 1; 1997: df = 3, F = 1.525, p = 0.213; 1998: df = 3, F = 2.434, p = 0.103; 1999: df = 3, F = 0.254, p = 0.952).

Table 1—A comparison of species richness and ground cover of native and exotic plants from data collected in 1 m^2 subplots over 3 years. Means listed across from year and standard errors shown in parentheses below its respective mean.

Year	n	# of native species				# of exotic species				Cover of native species (%)				Cover of exotic species (%)			
		EXC	CON	HRM	VHI	EXC	CON	HRM	VHI	EXC	CON	HRM	VHI	EXC	CON	HRM	VHI
1997	3	10.0	8.3	9.3	7.0	3.7	3.7	3.0	3.3	54.3	54.2	58.8	65.7	29.7	29.9	17.9	16.1
		(1.2)	(0.3)	(0.7)	(0.6)	(0.7)	(0.9)	(0.6)	(0.3)	(8.4)	(8.6)	(5.6)	(1.9)	(1.6)	(10.6)	(5.2)	(4.3)
1998	3	16.3	13.0	13.3	12.0	4.7	4.3	2.7	4.7	43.3	44.8	50.3	45.0	24.0	33.7	7.6	22.6
		(0.7)	(2.1)	(3.0)	(1.0)	(0.3)	(0.3)	(0.9)	(0.3)	(10.5)	(9.5)	(3.0)	(3.7)	(6.4)	(11.6)	(2.3)	(3.2)
1999	3	24.7	22.3	27.3	22.7	7.7	8.0	7.3	9.0	54.0	55.2	56.2	54.4	32.8	31.8	19.8	24.9
		(0.9)	(0.3)	(2.4)	(2.2)	(1.2)	(0.6)	(0.9)	(0.6)	(9.1)	(11.8)	(5.6)	(2.2)	(11.4)	(12.4)	(7.3)	(3.2)

Table 2—A comparison of measures of community composition (Shannon-Wiener, Evenness, and Simpson's Diversity) for 3 years of plant community data collected from 1 m^2 subplots. Mean values provided with standard errors shown in parentheses.

Year	Treatment	Shannon-Wiener		Evenness		Simpson's Diversity^{-1}	
1997	EXC	2.0	(0.2)	0.7	(0.04)	5.9	(1.0)
	CON	1.8	(0.1)	0.7	(0.03)	4.7	(0.7)
	HRM	1.8	(0.2)	0.7	(0.05)	4.8	(0.9)
	VHI	1.6	(0.1)	0.7	(0.05)	4.0	(0.2)
1998	EXC	1.9	(0.2)	0.7	(0.05)	5.1	(1.1)
	CON	1.9	(0.0)	0.7	(0.03)	5.1	(0.3)
	HRM	1.7	(0.2)	0.7	(0.05)	4.4	(0.7)
	VHI	1.8	(0.0)	0.7	(0.03)	5.0	(0.2)
1999	EXC	2.1	(0.2)	0.6	(0.06)	5.1	(1.2)
	CON	2.1	(0.2)	0.6	(0.05)	5.4	(1.0)
	HRM	2.1	(0.2)	0.6	(0.03)	5.5	(0.9)
	VHI	2.2	(0.1)	0.6	(0.02)	5.2	(0.2)

Diversity Indices

Assessing diversity can be done with many different analytical approaches, but perhaps the most common methods involve the calculation of diversity indices. Although ubiquitous in the ecological literature, diversity indices can be difficult to interpret because they are often codependant on species richness and species evenness. We have calculated the means and standard errors for each treatment, by year, for the Shannon-Wiener index, Evenness, and Simpson's Diversity. As displayed in table 2, treatments are fairly similar, regardless of index or year. These data are provided as descriptive information and no statistical tests were conducted.

Furthermore, to determine if alternative management styles, namely EXC, HRM, and VHI treatments, exhibited predictable patterns of unique species we calculated the Jaccard's Coefficient by comparing each alternative treatment plot with its partnered CON treatment plot. There was moderate similarity between plant communities in treatments, with an average of 60 percent overlap of species and less than 6 percent standard deviation (fig. 3).

Community-Level Comparisons

In addition to difficulties in analysis and interpretation, it is difficult to graphically display information from diversity indices. Instead, we display community data through ordination techniques, and we statistically tested the patterns with nonmetric multidimensional scaling. While no treatment effects were found to be significant, a strongly significant pattern was discovered for plant communities when analyzed by year. The data suggest that the plant communities differ significantly among years regardless of treatment type (fig. 4).

Species-Area Curves

It is commonplace in the ecological literature to extrapolate results from small-scale studies to larger landscapes. While most sampling methods fail to provide a framework for extrapolating data, the nested design of the Modified-Whittaker plot is ideally suited to these efforts. Extrapolations employing the species-area equation, corrected with Jaccard's Coefficient, were more accurate than extrapolations with the uncorrected species-area curve. More specifically, the uncorrected species-area curve tended to underestimate the plant community by roughly 20 species, whereas the corrected species-area curve underestimated by as few as three and no more than 13 (table 3).

Discussion _____

The significance of livestock grazing in determining patterns of community organization is generally believed to be great, but the literature demonstrates a poor understanding
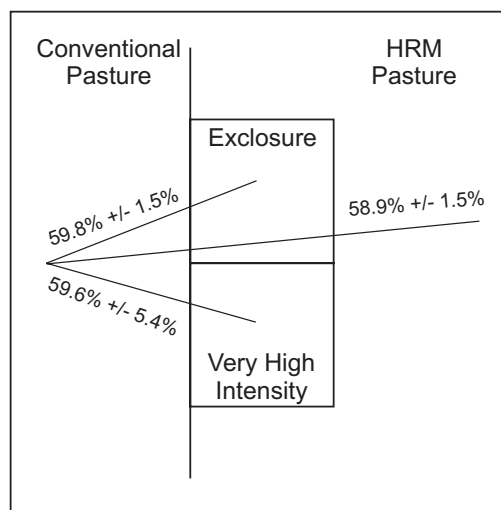
Figure 3—Species overlap between the CON treatment and each of the other three treatments. Jaccard's Coefficient values are averaged across the three replicates.
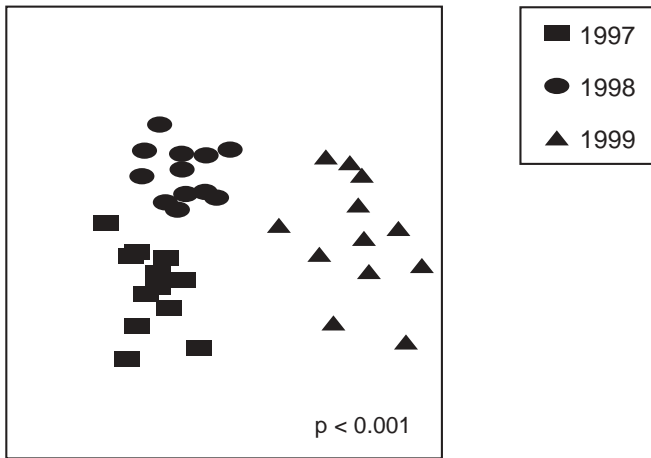
Figure 4—An ordination of plant community data determined with nonmetric multidimensional scaling. Each point represents the plant community of an individual plot from a specific year. Those communities that are more dissimilar will be found farther apart than those that are more similar. Treatments were not statistically significant. However groupings of plant communities by year that ignore treatment are statistically significant.

of specific patterns and processes. Milchunas and Lauenroth (1993) summarized 152 studies in grasslands and found sharply contrasting results with respect to the responses of species composition to large-herbivore grazing. The authors suggested that these conflicting findings may be due to one or more of the following explanations: (1) differing methodologies, including the intensity of grazing and the scale at which the response variable was measured; (2) the productivity of the grassland, which includes its soil resources and moisture; and (3) the evolutionary history of each grassland type (Milchunas and Lauenroth 1993). This paper most directly addresses the first, that is, the appropriate methodology for measuring grassland community responses to grazing treatments.

Grass Versus Forb Cover

Shifts in the proportional ground cover of plant functional groups as a function of grazing intensity have been well documented in highly productive sites. Long-term research conducted in the tallgrass prairies of Kansas demonstrated that perennial grasses created a near monoculture by outcompeting forbs when disturbance events were eliminated (Collins and others 1998). In comparatively less productive ecosystems, such as semiarid grasslands, the interactions of grasses and forbs are poorly understood. In one study of a semiarid grassland, Brady and others (1989) found ground cover increased for both grasses and forbs following the removal of cattle. In contrast, Rambo and Faeth (1999) found grass cover increased by 20 percent in two of three sites when grazing was eliminated for at least 8 years. Our own work suggests no immediate differences in grass or forb cover due to alternative grazing treatments. However, across all treatments grass cover is increasing while forb cover is decreasing. Although it may be too soon to draw conclusions about the fairly recent treatments, EXC and VHI, we have a higher level of confidence in comparing CON and HRM treatments, which have been in place for a minimum of 12 years. With 3 years of ground cover data, the alternative HRM management style does not exhibit a markedly different level of grass or forb cover.

Native Versus Exotic Richness and Cover

Although livestock grazing has often been implicated as a dispersal agent for exotic species (Hobbs and Huenneke 1992; Mack 1981), several studies show no link between grazing and exotic species richness. A multi-State sampling effort in the Northern Rocky Mountains found no consistent differences in native or exotic species richness due to livestock grazing (Stohlgren and others 1999). Similarly, in a semiarid grassland, Rambo and Faeth (1999) found no differences in exotic species richness in either long-term or short-term grazing exclosures. Preliminary findings from our work agree with these latter studies, showing no statistical difference in either species richness or ground cover of native or exotic species due to treatment. Our results do show a consistent increase in the overall number of species

Table 3—Standard species-area curves compared with species-area curves corrected with Jaccard's Coefficient. Equations were determined from 1 m^2, 100 m^2, and 1,000 m^2 plots, and then validated against observed species in the sum of the three 1,000 m^2 plots in each treatment.

Treatment	Standard Equation Y = m (log Area) + b	Estimate of spp. in 3,000 m^2	Corrected Equation Y = m / J (log Area) + b	Mean J (S.E.)	Corrected Estimate of spp. in 3,000 m^2	Actual spp. in 3,000 m^2
EXC	y = 6.2581x + 7.6129	29.4	y = 6.2581/0.631 + 7.6129	0.631 (0.066)	42.1	45
CON	y = 4.7742x + 9.8387	26.4	y = 4.7742/0.655 + 9.8387	0.655 (0.075)	35.2	48
HRM	y = 6.5x + 8	30.6	y = 6.5/0.609 + 8	0.609 (0.041)	45.1	49
VHI	y = 5.0806x + 8.129	25.8	y = 5.0806/0.649 + 8.129	0.649 (0.034)	35.3	44

from 1997 to 1999, which is likely to be a function of time since a severe drought year in 1996. Although not statistically significant, it is interesting to note that the HRM treatment had the lowest cover of exotic species in 2 of 3 years of sampling.

Plant Community Responses

By employing a variety of analytical tools we have demonstrated that the plant communities in each treatment do not differ significantly. This may be due to minimal ecological impacts or it may be that the duration of this study is not yet sufficient to detect ongoing, gradual change. Community changes are difficult to document because they involve the combination of species richness and the abundance of each individual species. To address this issue we compared traditional diversity indices and employed more powerful ordination techniques. The standard diversity indices, Shannon-Wiener, Evenness, and Simpson's, suggest no consistent differences among treatments. Furthermore, the species overlap between the conventional treatment and each of the other three treatments is nearly 60 percent for each comparison.

The ordination of community data with NMDS also demonstrated no significant differences due to treatment; however, there were strongly significant differences between each year. This finding suggests that year-to-year variation has had a stronger effect on the plant community than the treatments themselves. Even though the EXC and VHI treatments represent strikingly different livestock treatments, and the CON and HRM treatments represent longer-term, but less extreme treatments, the plant communities have not shown a strong response to grazing intensity over a 3-year period. Based on findings from other studies (for example, Sala and others 1992), we suggest that the plant community of this water-limited ecosystem is often more sensitive to the high variability in precipitation and soil moisture than to the livestock management treatments compared in this study.

Improved Power with Nested Sampling

Stohlgren and others (1995, 1998) have demonstrated a number of benefits from conducting multiscale nested sampling in plant communities. Our results support their conclusions. For our semiarid grassland study site, standard species-area curves consistently underestimated species richness by about 40 percent, and correcting the species-area curve with Jaccard's Coefficient improved estimates to within 15 percent of the known species richness. If the goal of a particular monitoring program is to estimate the total number of species in a given area of grassland, then nested sampling designs such as the Modified-Whittaker plot should provide increased prediction accuracy.

Management Implications

Land managers appear to be increasingly placed in the catch-22 situation of doing more with less. Especially in the stewardship of public lands, the short-term future appears to include further cuts to monitoring budgets, even while pressure grows to make ecologically sound decisions in the face of increasing litigation. Our preliminary results allow us to make two management suggestions, with the caveat that our preliminary results are specific to the Reed Lake study site, and should influence broader interpretations only where they concur with other published works.

First, future monitoring efforts should heed the warning that small sampling plots may provide significantly less accurate information then multiscale, nested sampling plot designs. At our study site, the use of Modified-Whittaker plots and replicated treatments allowed us to improve our species richness prediction accuracy by 45 percent. Efforts are under way to apply this sampling design to other response variables, including the diversity of macro- and micro-arthropods, and the size and fluxes of nutrient pools.

Second, alternative livestock approaches, including exclosures and very high intensity treatments, did not have immediate or dramatic effects on the plant community of this semiarid grassland. In fact, the HRM treatment that has been in place for a dozen years shows no statistically significant differences from the plant community of the adjacent conventionally grazed grasslands. Therefore, if the goal of a grassland management program is to move plant communities in a particular direction with alternative grazing practices, planners and practitioners must recognize the necessity for a long-term commitment to treatments and to an efficient, well targeted monitoring program.

Acknowledgments _____

Reference _____

Belsky, A. J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54: 419–431.

Berendse, F. 1999. Implications of increased litter production for plant biodiversity. Trends in Ecology and Evolution 14: 4–5.

Brady, W. W., M. R. Stromberg, E. F. Aldon, C. D. Bonham, and S. H. Henry. 1989. Response of a semidesert grassland to 16 years of rest from grazing. Journal of Range Management 42: 284–288.

Brown, D. E. 1994. Biotic Communities: Southwestern United States and Northwestern Mexico. University of Utah Press, Salt Lake City.

Collins, S. L., and S. M. Glenn. 1997. Effects of organismal and distance scaling on analysis of species distribution and abundance. Ecological Applications 7: 543–551.

Collins, S. L., A. K. Knapp, J. M. Briggs, J. M. Blair, and E. M. Steinauer. 1998. Modulation of diversity by grazing and mowing in native tallgrass prairie. Science 280: 745–747.

Conley, W., M. R. Conley, and T. R. Karl. 1992. A computational study of episodic events and historical context in long-term ecological process: climate and grazing in the northern Chihuahuan desert. Coenoses 7: 55–60.

Covington, W. W., R. L. Everett, R. Steele, L. L. Irwin, T. A. Daer, and A. N. D. Auclair. 1994. Historical and anticipated changes in forest ecosystems in the inland West of the United States. Journal of Sustainable Forestry 2: 13–63.

Fleischner, T. L. 1994. Ecological costs of livestock grazing in Western North America. Conservation Biology 8: 629–644.

Fuhlendorf, S. D., and F. E. Smeins. 1997. Long-term vegetation dynamics mediated by herbivores, weather, and fire in a *Juniperus-Quercus* savanna. Journal of Vegetation Science 8: 819–828.

Gibbens, R. P., and R. F. Beck. 1988. Changes in grass basal area and forb densities over a 64-year period on grassland types of the Jornada Experimental Range. Journal of Range Management 41: 186–192.

Hastings, J. R., and R. M. Turner. 1965. The changing mile: An ecological study of vegetation change with time in the lower mile of an arid and semiarid region. University of Arizona Press, Tucson, AZ.

Herbel, C. H., F. N. Ares, and R. A. Wright. 1972. Drought effects on a semidesert grassland range. Ecology 53: 1084–1093.

Hobbs, R. J., and L. F. Huenneke. 1992. Disturbance, diversity, and invasion: implications for conservation. Conservation Biology 6: 324–337.

Hughes, L. E. 1996. What's in an exclosure? Rangelands 18: 201–203.

Krebs, C. J. 1989. Ecological Methodology. Harper and Row, New York, New York, USA.

Kruse, W. H., and R. Jemison. 2000. Grazing systems of the Southwest: p. 27–52 in R. Jemison and C. Raish, eds. Livestock Management in the American Southwest: Ecology, Society, and Economics. Elsevier, Amsterdam.

Mack, R. N. 1981. Invasion of *Bromus tectorum* L. into Western North America: An ecological chronicle. Agro-Ecosystems 7: 145–165.

Magurran, A. E. 1988. Ecological Diversity and Its Measurement. Princeton University Press, Princeton, NJ.

Mayes, H. M., and T. F. Archer. 1982. Arizona Cattle Ranches on Public Land. College of Agriculture, University of Arizona, Tucson, AZ.

McNaughton, S. J. 1983. Serengeti grassland ecology: the role of composite environmental factors and contingency in community organization. Ecological Monographs 53: 291–320.

Milchunas, D. G., and W. K. Lauenroth. 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. Ecological Monographs 63: 327–366.

Minchin, P. R. 1987. Simulation of multidimensional community patterns: towards a comprehensive model. Vegetatio 71: 145–156.

Mueller-Dombois, D., and H. Ellenberg. 1974. Aims and methods in vegetation ecology. John Wiley & Sons, New York.

Noss, R. F., E. T. LaRoe III, and J. M. Scott. 1995. Endangered Ecosystems of the United States: a preliminary assessment of loss and degradation. U.S. Department of the Interior, National Biological Service, Washington, DC.

Rambo, J. L., and S. H. Faeth. 1999. Effect of vertebrate grazing on plant and insect community structure. Conservation Biology 13: 1047–1054.

Saab, V. A., C. E. Bock, T. D. Rich, and D. S. Dobkin. 1995. Livestock grazing effects in western North America. Pages 311–353 in T. E. Martin and D. M. Finch, eds. Ecology and Management of Neotropical Migratory Birds. Oxford Press, New York, NY.

Sala, O. E., W. K. Lauenroth, and W. J. Parton. 1992. Long-term soil water dynamics in the shortgrass steppe. Ecology 73: 1175–1181.

Samson, F. B., and F. L. Knopf. 1996. Prairie conservation: preserving North America's most endangered ecosystem. Island Press, Washington, DC.

Savory, A. 1988. Holistic Resource Management. Island Press, Washington, DC, USA.

Sisk, T. D., T. E. Crews, R. T. Eisfeldt, M. King, and E. Stanley. 1999. Assessing impacts of alternative livestock management practices: raging debates and a role for science. Pages 89–103 in C. van Riper, III and M. A. Stuart, eds. Fourth Biennial Conference of Research on the Colorado Plateau. U.GS, Flagstaff, AZ.

Sokal, R. R., and F. J. Rohlf. 1995. Biometry. W.H. Freeman and Co., New York, NY.

Stohlgren, T. J., G. W. Chong, M. A. Kalkhan, and L. D. Schell. 1997. Rapid assessment of plant diversity patterns: a methodology for landscapes. Environmental Monitoring and Assessment 48: 25–43.

Stohlgren, T. J., M. B. Falkner, and L. D. Schell. 1995. A modified-Whittaker nested vegetation sampling method. Vegetatio 117: 113–121.

Stohlgren, T. J., L. D. Schell, and B. Vanden Heuvel. 1998. How grazing and soil quality affect native and exotic plant diversity in rocky mountain grasslands. Ecological Applications 9: 45–64.

Stuth, J. W. 1991. Foraging behavior in R. K. Heithschmidt and J. W. Stuth, eds. Grazing Management: An Ecological Perspective. Timber Press, Portland, OR.

Trimble, S. W., and A. C. Mendel. 1995. The cow as a geomorphic agent: a critical review. Geomorphology 13: 233–253.

Wildeman, G., and J. H. Brock. 2000. Grazing in the southwest: history of land-use and grazing since 1540. p. 1–26 in R. Jemison and C. Raish, eds. Livestock Management in the American Southwest: Ecology, Society, and Economics. Elsevier, Amsterdam.