Reframing the Grazing Debate: Evaluating Ecological Sustainability and Bioregional Food Production

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Abstract. The semi-arid grasslands of the Colorado Plateau are productive, diverse, and extensive ecosystems. The majority of these ecosystems have been altered by human land use, primarily through the grazing of domestic livestock, yielding a plethora of environmental and social consequences that are tightly interconnected. From an agroecological perspective, untangling these issues requires both an understanding of the role of livestock grazing in bioregional food production and the effect of that grazing on ecological sustainability. To address the former, we discuss the importance of cattle ranching as a bioregional food source, including estimates of meat production and water use in Arizona. To address the latter, we present data from a long-term project addressing changes in native plant community composition, under a range of alternative livestock management strategies. Our study site near Flagstaff, AZ includes four different management treatments: (1) conventional low-intensity, long-duration grazing rotations; (2) high-intensity, short-duration rotations; (3) very high-impact, very short-duration grazing (to simulate herd impact); and, (4) livestock exclosure. Preliminary results suggest belowground properties are responding more quickly to grazing treatments than aboveground properties. Particular

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response variables, such as cyanobacteria and diatoms, show a marked short-term response to very high-impact, short-duration grazing, but long-term implications are as yet unknown.

Key words: ecological sustainability, bioregional food production, livestock grazing, biological diversity, participatory research.

INTRODUCTION

Two years ago, we addressed the 4th Biennial Conference of Research on the Colorado Plateau on the issue of the ecological sustainability of cattle grazing on arid rangelands (Sisk et al. 1999). At that time, our focus was on the role of science in helping to resolve the contentious and often bitter social battle over grazing policy and practices, and the opportunities presented by public participation in the scientific process (Sclove 1998). We demonstrated that the current level of understanding of grazing impacts in the Southwest often lacked a rigorous scientific foundation, and we suggested an approach for designing research efforts to address scientific issues underlying environmental conflict. The centerpiece of our efforts has been an experiment designed to test a set of hypotheses derived from differing claims voiced by ranchers, resource managers, and environmentalists about the ecological impacts of livestock grazing on the Colorado Plateau. Here we provide an update, expanding on the scientific and policy themes that are so closely interwoven in the grazing debate.

Currently, consensus on the issue of livestock grazing in the Southwest does not appear to be on the near horizon of the socio-political landscape. This impression is particularly apparent in the mainstream media that tend to emphasize the contentiousness of environmental issues (e.g., Rotstein 1999). However, deeper investigation into the ecological literature provides some evidence of a broad agreement on livestock impacts. For example, Belsky et al. (1999) summarized roughly 100 papers from the scientific literature that measured the effects of cattle grazing on riparian zones in the western U.S. Their review found considerable evidence that cattle grazing often has negative effects on stream channel morphology, soils, vegetation, and wildlife. This review and others (e.g., Platts 1991, Kauffman and Krueger 1984, Armour et al. 1994), make a compelling case that livestock grazing should be carefully controlled, if not altogether eliminated, along riparian zones.

Riparian ecosystems, however, represent only a fraction of grazed lands in the Southwest, and information from this sensitive habitat-type does not necessarily pertain to other ecosystems. Upland grasslands, which constitute the majority of grazed lands, differ substantially from riparian ecosystems in structure, function, and evolutionary history, and the impacts of livestock grazing on these two ecosystems may be very different. Although we know of no rigorous scientific comparison of Southwestern riparian and upland responses to similar grazing systems, the literature suggests that the response of upland systems are more varied. Rambo and

Faeth (1999), studied semi-arid grasslands that had been excluded from grazing for over eight years, and showed that ungrazed grasslands had fewer plant species than adjacent, grazed plots. Insect species richness, however, showed no significant difference. In studies of ground-foraging birds, Bock et al. (1984) found that grazed areas and adjacent exclosures had similar abundances in years of average rainfall, but exclosures supported nearly 3 times as many birds as the grazed areas following two consecutive drought years (Bock and Bock 1999). This complexity of organismal responses to grazing, as well as an overall paucity of rigorous scientific information, has motivated our efforts to address relevant ecological questions through manipulative experiments conducted in concert with ranch management teams. We provide a brief retrospective on our involvement with two such groups that include environmental advocates and policy makers, and explain how this experience has provided a broader context for considering trade-offs associated with livestock grazing in the Southwest.

Ground Zero for Grazing Policy

For several decades, the center of conflict regarding grazing policy has focused on whether grazing degrades "the land." Fifty years of research provides clear, but equivocal evidence: it does in some places and at some times, and at other times and places it does not. In fact, there is also compelling evidence that livestock grazing can speed the recovery of certain degraded sites (van Wieren 1991), and that grazing may increase productivity in some ecosystems (McNaughton et al. 1997, Milchunas et al. 1989). Clearly, further efforts to characterize grazing as "good" or "bad" are overly simplistic and, we believe, problematic. Instead, two broad questions emerge: (1) how and where can grazing be practiced in an ecologically sustainable manner; and (2) how do we, the public, wish to manage our public grasslands in the Southwest? The answers to the former question will come from greater collaborative interaction among ranchers, research scientists, environmental groups, and the public who plan and apply on-the-ground management. We are optimistic that the collaborative groups, being founded with increasing frequency across the West, will be at the forefront of collaborative decision-making. The latter question however, is less tractable. Extreme, polarizing views are propagated daily through the media as demonstrated by the well-circulated jingle "cattle-free by '93" (now "2003") and the directly opposing political views espoused through the ranching industry. In fact, the contest has become so mythologized and self-referential that it is easy to lose sight of the real questions, such as whether regional agriculture is important to the four-corners states, what lands can be grazed sustainably and profitably, and what alternative land management should replace grazing in areas where it is unsustainable or not desired by the public.

Bioregional Perspective of Food Production

The scientific debate over livestock grazing has focused primarily on single species' responses (such as endangered species) and overall forage production. Ecosystems grazed by livestock have justifiably been compared to ungrazed areas to ascertain human impacts on natural systems. Interestingly, few comparisons are made between the biological diversity and ecosystem function of grazed ecosystems and other agroecosystems. In other words, if we assume that humans are going to impact natural ecosystems to produce food and fiber through agriculture, it seems appropriate to consider the relative ecological impacts of different agricultural practices in the arid Southwest.

Inherent to conventional agroecosystems dedicated to annual crop production is the nearly total replacement of native plant and animal communities. They generally consist of non-native plants (both crops and weeds), and fauna (especially birds, mammals, and arthropods) that can exist in communities that experience disturbance at high frequencies and intensities through actions such as plowing soils, which often increase erosion rates and contribute to a decline in soil organic matter (Davidson and Ackerman 1993). Rarely do modern agricultural systems generate sufficient nutrients internally to balance nutrients exported in crops, thus most farms depend on large inputs of synthetic fertilizers (Doerge et al. 1991). The crop uptake of these fertilizers however is fairly inefficient, often not higher than 50%, with residue nutrients often making their way into waterways, or the atmosphere (Matson et al. 1998). Inputs of pesticides including insecticides, herbicides and fungicides are also commonplace in conventional agroecosystems. While the pesticides applied today are less persistent in the environment than those used in previous decades, they are nonetheless highly toxic and relatively indiscriminate in the species that they affect. Finally, modern agroecosystems require substantial fossil fuel subsidies in the production process. The energy used to cultivate, harvest, synthesize and apply fertilizers, and irrigate, primarily comes from fossil fuels. The energy return on each fossil-fuel calorie invested in agriculture tends to be quite low (Pimentel and Pimentel 1996).

When compared with agro-ecosystems dedicated to annual agriculture, plant species diversity in grazed, upland agroecosystems in the Southwest appear relatively intact (Hughes 1996, Rambo and Faeth 1999). The specific ecological impacts of cattle grazing are often difficult to estimate, given the lack of non-grazed ecosystems that can be used as controls. However, this is not to say that livestock grazing is innocuous, because there is strong evidence that grazing can alter community composition of particular ecosystems through mechanisms such as selective biomass removal, alteration of soil properties, fire suppression, and transport of exotic species (Fleischner 1994). Indirect consequences of livestock grazing, such as the introduction of grasses for forage, especially Lehmann lovegrass (*Eragrostis lehmanniana*) and buffelgrass (*Pennisetum ciliare*), have had profound impacts on community dynamics in the Southwest (Bock and Bock 1998, Burquez and Martinez-Yrizar 1997). Where exotics have not been intentionally introduced, however, grazed ecosystems are generally dominated by native, perennial species (Rambo and Faeth 1999).

Estimating total costs of any agriculture is challenging given the gulf that exists between actual and perceived costs of natural resources. But without accurate cost estimates, the grazing debate remains awash with ambiguous statements. In 1990, crop agriculture in Arizona used approximately 5.2 million acre feet of water (Eden and Wallace 1992). Livestock in Arizona consume approximately 15 gallons animal-unit $^{-1}$ day $^{-1}$, which translates into an estimated annual water consumption by all range-fed Arizona livestock of only 8,384 acre-feet of water (1 acre-foot water = 1233.482 m^3; Table 1). When ranchers manage their livestock using horses, livestock grazing on Western rangelands may represent the only food production system in the United States that is based largely on solar energy rather than fossil fuel inputs. In other words, the work performed and inputs used to grow crops or raise animals in most agroecosystems involves a very significant reliance on commercial energy (Pimentel and Pimentel 1996). Producing livestock on western rangelands, however, relies heavily on native rates of net primary productivity, while using wind, gravity and/or solar panels to provide water.

Tradeoffs

Livestock grazing may have lower ecosystem impacts than annual agriculture, but it is also much less productive. A critical question, therefore, is whether the production of food from rangelands balances the tradeoffs in native ecosystem diversity and productivity that may occur with livestock grazing. To begin to address this question, it is important to develop a sense of arid rangeland food productivity. Following, we estimate levels of meat produced by cattle grazing on Arizona rangelands, excluding feedlot productivity. While these estimates are crude, we believe they provide a reasonable, approximate understanding of potential protein production.

Table 1. Estimated annual meat production and livestock water consumption according to ecosystem type in Arizona.

Ecosystem	Acres AUM^{-1} [1]	Area (ha)	% cover	Ha animal unit^{-1} year^{-1}	edible beef prod.[3,4] kg year^{-1}	protein[5] kg year^{-1}	water consumed[6] m^3 year^{-1}
Chaparral	12.5	1,303,452	4	61	1,871,842	411,805	442,808
Grassland	4.1	5,793,686	24	20	25,376,344	5,582,795	6,003,092
Pinon-Juniper	12.5	5,164,781	18	61	7,416,964	1,631,732	1,754,576
Ponderosa	19.8	885,079	3	96	807,634	177,679	191,056
Desert	20.0	9,143,387	31	97	8,257,226	1,816,611	1,953,374
Total		**22,290,385**	**80**		**43,730,113**	**9,620,625**	**10,344,905**

1 AUM = animal unit month = the area (in acres) required to feed one steer or cow/calf unit for 1 month. AUMs based on actual stocking rates for different Arizona ecosystems reported in USFW (1999)

2 D. Brown (pers. comm.)

3 In an animal's first year on the range, it will gain ~190 kg, and if it is left for a second year, it will gain ~330 kg in a good (wet) year and as low as 165 in a dry year. On average, therefore, an animal gains approximately 219 kg yr^{-1} (A. Kessler and D. Moroney, pers. comm.)

4 Edible meat constitutes ~40% of the total animal weight

5 Beef is ~22% protein (Ensminger et al. 1983) and the average yearly protein requirement for a person is ~23.7 kg

6 One cow or steer requires 15 gallons of water per day (Naeser and St. John 1998)

Stocking rates of livestock on lands in Arizona range between 4 and 20 acres AUM^{-1} (an animal unit month is either one steer or cow-calf pair) for desert, chaparral and woodland ecosystems (USFW 1999). This range in stocking rates reflects the variation in herbaceous, aboveground net primary productivity of the different ecosystems. By making conservative assumptions about stocking rates, we estimate that the current grazing of 80% of Arizona's land surface (Mayes and Archer 1982) results in sufficient protein production to supply one million people with 40% of their annual requirements (assuming 65g protein $capita^{-1}$ day^{-1}). Alternatively, if livestock numbers were decreased by 50% across all ecosystem types, then the 40% of Arizona land that would remain grazed could supply one million people with approximately one-fifth of their annual protein requirements. This latter level of food production is large enough that we believe the value of bioregional food production needs to be considered in the debate regarding livestock grazing in the arid Southwest. Elimination of livestock grazing in the Southwest would substantially impede any regional movement toward greater reliance on bioregional food production, and would shift agricultural activity, as well as the concomitant environmental impacts, to other regions. The potential socio-economic implications of such a proposal are beyond the scope of this paper, but undoubtedly warrant further consideration.

Reshaping the Debate

Native Habitats as the Endpoint

Although plant surveys have been a mainstay of the vast majority of grazing studies, the emphasis has often been placed on total forage, without regard for the particular species that make up the community (e.g., Holechek et al. 1999). Increasing public recognition of the value of native habitats and native species has made this an issue of contention in the current grazing debate. Dramatic declines in native habitats, such as the degradation or loss of 80% of Western riparian ecosystems (U.S. Department of Interior 1994), underscore the rapidity of change wrought by humans. Moreover, the list of nonnative plant species in Arizona has doubled in the past 50 years to roughly 330 and continues to grow (Burgess et al. 1991). Complicating this issue is the fact that the establishment of many nonnative plants in grasslands was aided in the early 1900s by government-subsidized seeding programs that intentionally (and unintentionally) included nonnative plants (Bahre 1995, Cox and Ruyle 1998). This trend in the loss of native habitats and native species is the product of multiple land-use actions, many of which are historically associated with, but not inherently necessary to livestock production (e.g., road building, erosion of streambanks, extensive fencing, chaining of trees, etc.).

Many examples of landscapes severely degraded by overgrazing exist and the mismanagement of rangeland has fueled a widespread anti-grazing sentiment. Many environmental groups have advocated the complete removal of cattle from large tracts of land, and this approach has been implemented on many National Park Service lands (Anderson 1993). The responses of arid and semi-arid grasslands to exclusion from cattle grazing have been mixed, with changing richness of native species ranging from dramatic increases (Brady et al. 1989) to slight decreases (Rambo and Faeth 1999). When interpreting vegetation responses to livestock removal, however, it is important to recognize that virtually all lands that are accessible to cattle or sheep in the Southwest have been grazed intensively at one time or another. Lands currently excluded from grazing do not necessarily represent the state of semi-arid grassland ecosystems prior to the introduction of domestic livestock (Milton et al. 1994), an ecological state that remains poorly understood and whose restoration is beyond current technical capacity. Instead, lands where grazing has been eliminated represent the likely endpoint of cattle removal from similar ecosystems that are currently being grazed. Thus, constructive approaches to resolving the present grazing debate will include the assessment of expected outcomes of different levels and styles of rangeland management (including livestock removal), rather than a restricted and largely theoretical choice between current conditions and those that predominated prior to the arrival of domestic grazers.

A Role for Research

Clearly, a broad range of land management options currently exists, and many are being implemented and evaluated across the Southwest. Science provides a framework for measuring and interpreting the environmental implications of each option. To assess some key elements of the ecological sustainability of grazing in semi-arid grasslands, we asked the following question: Do belowground and aboveground variables affecting grassland composition and function, respond in a predictable manner to increasing grazing intensity? For belowground properties, we measured soil compaction and specific members of the microbiotic community, whereas we measured plant cover and macro-arthropods as aboveground properties.

METHODS

Meaningful application of science to grazing issues will require comparisons of the effects of actual management practices, as well as experimental treatments designed to elucidate the relationships between grazing and ecosystem sustainability. In 1997 we initiated a study of grazing impacts in a semi-arid grassland in Arizona. Our experimental design, replicated in time and space, consists of four treatments in three blocks on the landscape (a total 12 study plots; see Sisk et al. 1999). The four treatments are as follows: (1) conventional low-density, long-duration grazing rotations (CON); (2) high-density, short-duration rotations inspired by Savory (1988) Holistic Resource Management (HRM); (3) very high-impact, 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 and land management agencies on adjoining pastures, while the latter two treatments are implemented on fenced 1-ha experimental plots created and managed by researchers. The timing of the graze event for each of the three cattle treatments falls within the months of May-October, but specific dates vary between years due to fluctuating environmental conditions and ranching logistics. Of the four treatments, only the VHI treatment does not represent a current grazing policy, but it does simulate herd behavior, and serves as a critical upper-end treatment to study the potential spectrum of responses. In all CON and HRM plots, we have carefully matched elevation, exposure, soil type, and vegetation type so that spatially and temporally extensive treatment effects can be complemented with the exclosure and VHI treatments implemented on 1-ha plots. For the purpose of this paper, we present data from one study site on the southern edge of the Colorado Plateau.

Site Description

Located at 2160m elevation in north-central Arizona, our primary field site is Reed Lake, characterized by Upper Great Basin grassland (Brown 1994) surrounded by Ponderosa Pine forest. Dominant perennial grasses are *Agropyron smithii* (western wheat grass) and *Elymus elymoides* (squirrel-tail grass). Soil type is fairly homogeneous among study plots and across our study site of approximately 25 ha, with a standard error of less than 10% for each soil particle size class. The top 8 cm of soil is, on average, comprised of 42% sand, 12% silt, and 44% clay (Fig. 1). Annual precipitation averages between 300 mm and 460 mm with the majority generally falling as monsoonal rains between June and September (Brown 1994).

Response Variables

Soil Compaction

As the intensity of cattle grazing increases, the amount of trampling increases. We measured soil compaction in the top 5 cm of the soil surface with a pocket soil penetrometer (Ben Meadows Company, Atlanta, GA 30341). In October of 1999,

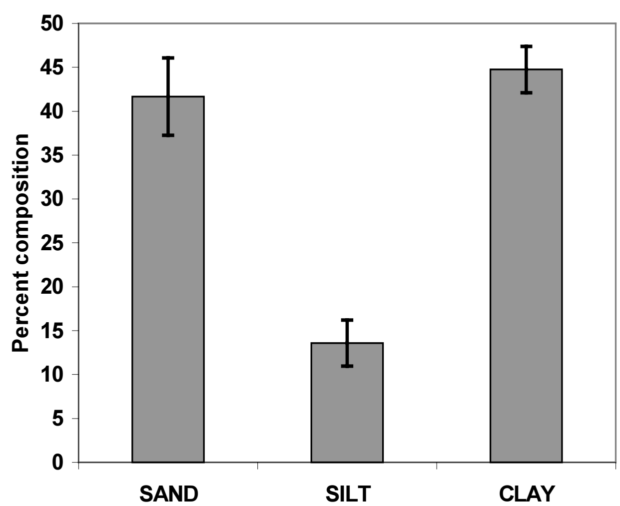

Figure 1. Soil composition for 12, 1-ha plots at the Reed Lake study site. Consistency in proportional representation of particle size classes suggests that results from the grazing experiment are not confounded by differences in soil type.

each plot was measured in three locations that were haphazardly selected (except for one of three plots in the conventional treatment that was missed due to a rainstorm). Within each of these locations the average of three readings was used as a final soil compaction measurement. This sampling event followed the conclusion of grazing for 1999, and was chosen to represent the cumulative compaction for that year. Data were analyzed with ANOVA.

Soil Microbiotic Community

Alterations of soil quality can have effects on cyanobacteria populations and, consequently, on their role as nitrogen fixers (Evans and Belnap 1999). In 1999, we employed a slide-incubation technique to assess cyanobacteria and diatoms (Rossi and Riccardo 1927, Rossi et al. 1935). Prior to the 1999 grazing season, five microscope slides were placed in each corner of EXC and VHI plots, which minimized potential disruption due to researchers in the plot. Slides remained in the ground for 26 days to incubate microbes and were subsequently transported to the laboratory. Cyanobacteria filaments and diatoms were then counted at 20X magnification with a phase contrast microscope. Data were analyzed with a nested ANOVA.

Plant Cover

Beginning in 1997, before the EXC and VHI treatments were initiated, we conducted annual ground cover (both basal and foliar cover) surveys using the modified-Whittaker plot design (Stohlgren et al. 1995). A modified-Whittaker plot was placed within each of the 12 plots and permanently marked, so that the researchers can return annually to conduct surveys. Data were analyzed for 1997-99 with a repeated measures ANOVA.

Arthropods

In 1998 we conducted sweep-net surveys of plots in the EXC and VHI treatments before and after the VHI grazing event. Total abundance of these vegetation-dwelling arthropods was calculated for each plot. Data were analyzed with a repeated measures ANOVA.

RESULTS

Soil Compaction

In comparison with the EXC and CON treatments, the HRM and VHI treatments showed greater soil compaction (df=3, F=15.308, P=0.006; Fig. 2). These increases are likely to have effects on other soil properties, including bulk density and infiltration rates, but the extent of these effects will depend on the persistence of these differences, which can only be determined through longer-term study.

Soil Microbiotic Community

Our VHI treatment had roughly 50% less colonization by cyanobacteria and diatoms, in comparison with the EXC plots (df=1, F=8.98, P=0.0047; Fig. 3). Because these organisms alter soil structure and fix nitrogen, these declines in abundance may portend further ecological consequences.

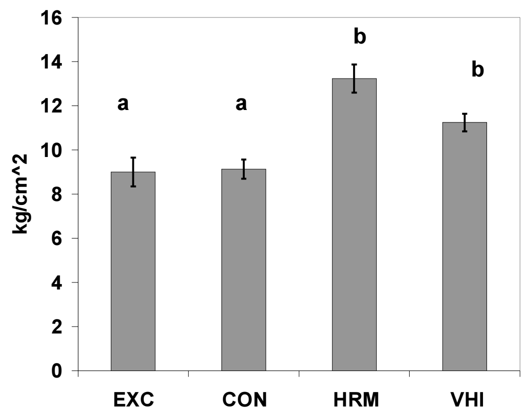

Figure 2. Soil compaction under four grazing treatments, following three years of treatment. EXC = livestock exclosure; CON = conventional, low-density, long-duration grazing; HRM = high-intensity, short-duration grazing; VHI = very high-intensity, short-duration grazing to simulate herd impact. Different letters denote significant differences in the degree of soil compaction (df=3, F=15.308, P=0.006).

Plant Cover

We found plant cover to be fairly similar among treatments, ranging from 78% to 88% (Fig. 4). Year-to-year variation in total plant cover was not significant, whereas treatment type was a significant factor (df=3, F=9.87, P<0.0001). At a finer scale of inspection, total plant cover measurements showed the HRM treatment to be consistently lower than the EXC and CON treatments by about 7-9%. Furthermore, the

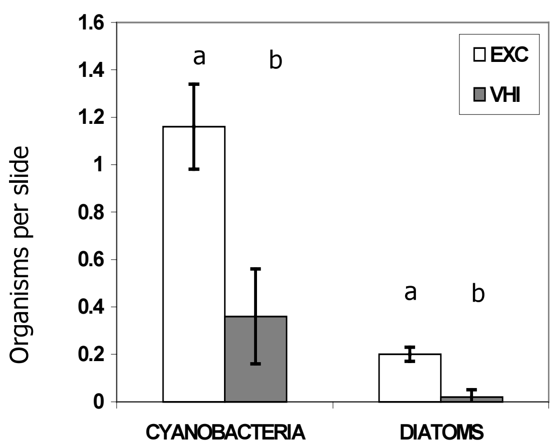

Figure 3. Abundance of cyanobacteria and diatoms in exclosures (EXC) and very high-intensity (VHI) plots following two years of treatment. Different letters denote significant differences in abundances (df=1, F=8.98, P=0.0047).

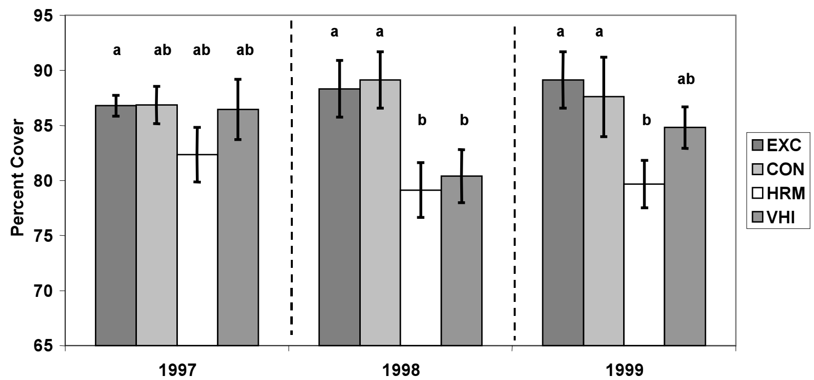

Figure 4. Percent plant cover for four grazing treatments (see Fig. 2 and text for treatment descriptions). EXC and VHI treatments were initiated following data collection in 1997, while CON and HRM treatments were put in place more than 12 years prior to initiation of this study. Between-year differences were not significant for any treatment; significant within-year differences among treatments are denoted by different letters (df=3, F=9.87, P<0.0001).

VHI treatment exhibited an 8% decline in plant cover after one year of treatment, but this difference did not persist into 1999. In general, short-term effects of treatments were measured, but their long-term implications remain unclear. Finer resolution measures, such as comparisons of community composition, are addressed in a separate paper (Loeser et al. in prep.).

Arthropods

Pre-graze and post-graze sampling of EXC and VHI plots showed a decline of greater than 50% in arthropod abundance following the VHI grazing event in 1998 (df=1, F=5.95, P=0.07; Fig. 5). In contrast to this short-term response, the pre-graze abundance, which is a measure of response since the 1997 grazing event, did not differ between treatments, suggesting that long-term effects may be negligible.

DISCUSSION

Although we are in the early stages of a long-term study, we have detected short-term differences among four treatments reflecting a gradient of grazing intensity. In general, it appears that soil properties and belowground processes are more sensitive, over the short-term, to differences in grazing treatments than are aboveground properties. This supports similar conclusions drawn by Anderson (1995) who argued that belowground organisms may be keenly susceptible to land-use change. Measurements of short-term changes in above- and belowground communities due to grazing were not unexpected, however, the more ecologically and policy relevant questions involve long-term shifts in biological diversity and ecosystem productivity.

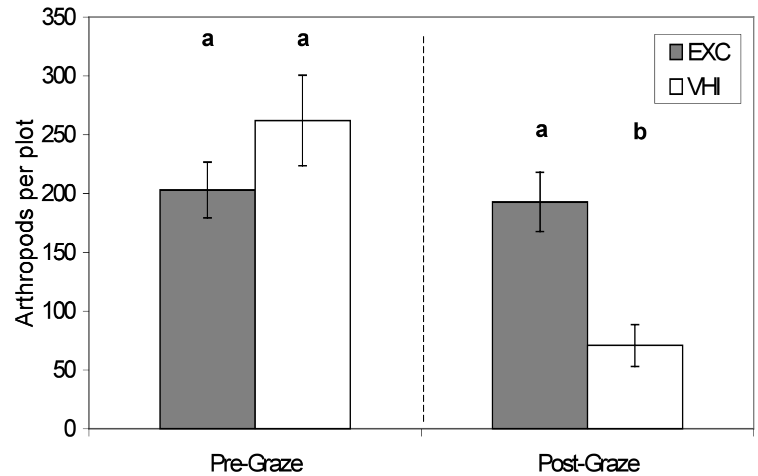

Figure 5. Arthropod abundance in exclosures (EXC) and very high-intensity (VHI) plots, before and after the 1998 grazing event, as determined by sweep net samples. Different letters denote a significant difference (df=1, F=5.95, P=0.07).

While these questions will only be answered with longer-term datasets, the short-term changes we have detected indicate that the experimental treatments have had significant, measurable effects that capture relevant impacts along a gradient of grazing intensities.

Belowground Properties

If the fundamental structure of the soil is being altered by the more intensive grazing treatments, as suggested by an increase in compaction in HRM and VHI plots, we would expect belowground soil organisms to respond. Furthermore, soil structural changes will likely affect other abiotic parameters, such as water penetration and retention. Preliminary results from our soil moisture measurements suggest that more heavily compacted sites have 1-5% less soil moisture (Loeser et al. unpub.). These alterations in soil abiotic parameters likely explain the nearly two-fold decrease in cyanobacteria and diatoms in the VHI compared to EXC treatments. Soil microorganisms in particular have limited mobility and are known to be sensitive to compaction (Whitford et al. 1995). Preliminary results from other ongoing studies at this site suggest that soil microarthropod abundance is roughly 40% lower in VHI plots than EXC plots (Loeser et al. unpub.).

Aboveground Properties

While belowground properties appear to be responding quickly to treatment effects, aboveground organisms, including plants and arthropods, have not yet demonstrated clear trends. Plots of the HRM treatment consistently showed lower ground cover than EXC plots, but because this was evident at the time that experiment began, it cannot be ascribed to the treatment itself. A treatment effect did occur in VHI plots after only one year, resulting in a loss of 10% of the live plant cover, but this difference did not persist into subsequent years. When we tested the possible relationship between arthropod samples and plant data, we did not find significant correlations (R^2=0.01, P=0.12). Arthropod samples collected shortly after the VHI grazing event showed a significant decline in total arthropod abundance, but samples from 1999, collected prior to grazing, did not differ significantly among treatments. While this suggests rapid recovery of the arthropod fauna, future collections over larger areas will be needed to determine long-term trends. Although our initial results are not conclusive, they indicate that alternative grazing treatments, such as the EXC and VHI treatments, have mixed effects on plants and arthropod communities.

Although aboveground measurements, such as plant cover and species richness, tend to dominate the grazing literature, we have demonstrated that measurements at multiple trophic levels offer additional information and provide a tractable approach for investigating grazing impacts on underlying ecosystem processes. A traditional animal- or forage-based approach would likely conclude that these treatment effects do not differ significantly, but clearly the impacts are more complicated, particularly within the soil. While additional data over an extended time period will be required to untangle grazing impacts and their ecological consequences, significant short-term differences in particular response variables between the two most extreme treatments indicate the methods that we employed to measure changes in this system are robust, and that long-term research efforts are justified.

Assessing the multi-faceted environmental implications of livestock grazing in the Southwest requires objective quantification of grazing impacts. We believe that an assessment of the environmental impacts of grazing should also examine grazing policy in the context of the increasing need for ecologically sustainable agriculture. Our research demonstrates short-term negative effects of very high grazing events on soil fauna and arthropods, but has not yet demonstrated long-term patterns in aboveground properties. As one of the very few bioregionally significant food production systems on the southern Colorado Plateau, grazing provides a significant source of edible protein that utilizes grassland communities comprised largely of native species. Efforts to generate more detailed and credible information on cattle and grassland community production levels might serve as common ground for opposing parties to discuss real-world compromises and the inclusive environmental impacts of livestock grazing versus increasing reliance on food, water, and energy imports to support the region's growing human population. We strongly believe that future research should move beyond the simplistic approach of grazed-versus-ungrazed comparisons to address a wider range of grazing practices, in order to more effectively determine whether an ecologically sustainable and socially acceptable level of grazing may exist for the publicly owned semi-arid grasslands of the Colorado Plateau.

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