

## ORIGINAL ARTICLE

# Rangeland management in highly variable environments: Resource variations across the landscape mediate the impact of grazing on vegetation in Mongolia

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## Keywords

Grazing strategy; key resources; landscape heterogeneity; non-equilibrium model; plant-herbivore interaction.

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## Abstract

Assessment of grazing-induced degradation of arid and semiarid rangelands with stochastic rainfall regimes is challenging. To assess grazing impacts on vegetation under highly variable environments, we focused on resource variations in relation to the scale of local rangeland use. Specifically, we tested the hypothesis that grazing impacts on resources were greater in areas heavily used during drought (i.e. key resource areas) than elsewhere. Near Mandalgobi, Mongolia, we established study sites in four typical vegetation communities (*Caragana*, *Allium*, *Achnatherum* and *Reaumuria*) according to landscape type. First, we interviewed key informants about the four community types to examine how local pastoralists used these communities during normal and drought summers. Second, we compared dung pellets numbers among communities to quantify the different grazing impacts on each community. Finally, to determine grazing impacts on vegetation in each community, we performed vegetation surveys in grazing and non-grazing plots at each site. According to respondents, during normal summers all types of community were used, but during drought, the main community used was *Achnatherum* community. The number of dung pellets was highest in the *Achnatherum* community. Species composition was significantly more affected by grazing in the *Achnatherum* and *Reaumuria* community than in the other communities. These results suggested that the *Achnatherum* community would have a role of key resource in study sites and grazing impacts affected key resource areas. If rangeland management is focused only on maintaining resource accessibility, then grazing-induced degradation of key resource areas could occur even on highly variable rangelands.

## Introduction

Assessment of grazing-induced degradation in arid and semiarid rangeland ecosystems with stochastic rainfall regimes is a challenge (Briske *et al.* 2003). Non-equilibrium rangeland modeling predicts that the occurrence of frequent multi-year droughts contributes to herbivore mortality and prevents herbivore numbers from increasing to carrying capacity in these variable environments (Ellis and Swift 1988). Therefore, plant composition and biomass on these rangelands are driven primarily by rain-

fall and not by grazing pressure, and under this model degradation of vegetation as a result of overgrazing is thus unlikely (Ellis and Swift 1988; Coughenour 1991; Behnke *et al.* 1993). Consequently, non-equilibrium modeling assumes that control of livestock numbers is less useful for rangeland management than is maintenance of resource accessibility (Ellis and Swift 1988; Behnke *et al.* 1993). Although many empirical studies have examined the non-equilibrium model (e.g. Fernandez-Gimenez and Allen-Diaz 1999; Fynn and O'Connor 2000; Oba *et al.* 2000; Hein 2006; Todd and Hoffman 2009; Dorji *et al.*

2010; Wesche *et al.* 2010), they have produced inconsistent results and there is no consensus as to whether livestock numbers should be controlled in highly variable rangeland systems.

Recent studies suggest that the relationship between non-equilibrium and equilibrium models is not exclusive, and that rangeland systems encompass both non-equilibrium and equilibrium environments at different scales (Briske *et al.* 2003; Vetter 2005; Derry and Boone 2010). For example, non-equilibrium models were accepted at the coarse scale while equilibrium models were accepted at the fine scale (Oba *et al.* 2003; Zemlich 2007). Thus, grazing impact on vegetation may appear differently by scales, and therefore to develop rangeland management we need to focus on the scale that related to local rangeland use when we assess grazing impact (Vetter 2005).

Our aim here was to assess the impacts of grazing on vegetation in highly variable environments by considering resource variations at a landscape scale in relation to the scale of local rangeland use. Specifically, we focus on key resource areas that pastoralists use for livestock survival during drought. Key resources are defined in terms of the key factor determining livestock populations – usually survival through the season of plant dormancy – and thus the availability of forage during drought (Illius and O'Connor 2000; Vetter 2005). Illius and O'Connor (1999) predicted that key resource areas, were more prone to deterioration than non-key resource areas from the impacts of concentrated grazing. Based on Illius and O'Connor's (1999) prediction, we established the hypothesis that on Mongolian rangelands, grazing impacts would be greater on key resource areas than in other areas. To test the hypothesis, we performed research in two steps: (i) we examined how local pastoralists use various resources by interviewing key informants; and (ii) we then used vegetation surveys to examine how grazing affected the vegetation species composition of each resource.

## Materials and methods

### Study area

Our study area was in the Saintsagaan *sum* (district), which includes the city of Mandalgobi (45°46'N, 106°16'E) in Mongolia's Dundgobi province. Mandalgobi is located in the desert-steppe ecological zone. The main livestock are sheep, goats, horses and camels. The climate is arid and cold, with a short summer. From 2000 to 2010, the mean annual air temperature was  $-0.29^{\circ}\text{C}$  and the average annual precipitation was 110.8 mm (min: 64 mm, max: 212 mm), with a coefficient of variance (CV) of 45.8% (data from the National Climatic Data Center). Our study sites were considered to be highly variable environments

according to Ellis's (1994) prediction that non-equilibrium dynamics predominate in areas where mean annual precipitation is less than 250 mm and precipitation CV exceeds 33%. Our survey was conducted in July to August 2010, and total precipitation in 2010 is 101.8 mm, and total precipitation in the previous year of research was 81.0 mm; therefore, we considered 2010 as normal year.

### Study design

We set up four ecological sites, each located at a different landscape position (Table 1). The ecological sites were located on the land catena (i.e. sequence) found on denudation planes, namely hillslope, tableland, pediment and depression; these land units are typical of the study area (Sasaki *et al.* 2008b). We called these sites the *Caragana* community, *Allium* community, *Achnatherum* community and *Reaumuria* community, respectively, in accordance with the plant species that dominated at each site. The *Caragana* community is dominated by the legume shrubs *Caragana microphylla* (Pall.) Lam. and perennial grass *Cleistogenes squarrosa* (Trin.) Keng and forbs *Allium mongolicum* Rgl. The *Allium* community is dominated by perennial forbs *Allium polyrrhizum* Turcz. ex Rgl. and largely distributed in the study area. The *Achnatherum* community is dominated by large, dense bushy grass *Achnatherum splendens* (Trin.) Nevski and forb *Salsolla collina* Pall. and *Reaumuria* community is located at a depression with prominent salinized soils (with higher pH and EC) (Sasaki *et al.* 2008b), and dominated by resistance to salinity, mesophytic semi-shrubs *Reaumuria soongorica* (Pall) Maxim., *Salsolla passerina* Bge. and *Kalidium foliatum* (Pall.) Moq. (Table 1). The taxonomic nomenclature follows that of Grubov (1982).

Each site had grazing and non-grazing plots. The grazing plots had three replicates and the non-grazing plots had one or two replicates (Table 1). Grazing plots were in different patches of each vegetation type and were located at least 300 m apart. However, replicate plots within the sites were located in closely similar topo-

**Table 1** Land units, dominant vegetation types, size and period of enclosure for the four community types

Community type	Land unit	Dominant vegetation type	Size of enclosure	No. years enclosed
<i>Caragana</i>	Hill slope	Shrubland	500 × 500 m	17
			50 × 50 m	6
<i>Allium</i>	Tableland	Grassland	3 × 0.5 km	29
<i>Achnatherum</i>	Pediment	Grassland	3 × 0.5 km	29
<i>Reaumuria</i>	Depression	Halophytic shrubland	50 × 50 m	6

graphic position, because potential correlations between grazing impact and other environmental factors (especially micro-topography) often create confounding effects due to sampling across different vegetation types and soil textures (Landsberg *et al.* 2002; Sasaki *et al.* 2005; Todd 2006). We therefore selected plots where these confounding effects appeared to be smallest. Moreover, we did not locate plots within 500 m of livestock camps and water sources, because livestock tend to concentrate in these areas. Thus, replicate grazing plots within sites were located in distinct vegetation patches in closely similar topographic positions, avoiding areas close to grazing sources where there was heavy grazing concentration. Each grazing plot was 20 × 20 m.

For the non-grazing plots we used five exclosures (one each for the *Allium*, *Achnatherum* and *Reaumuria* communities and two for the *Caragana* community). The sizes and durations of the grazing exclosures are given in Table 1. It is difficult to find ungrazed sites in Mongolia, because the vast majority of the rangeland is freely accessible (Sasaki *et al.* 2009a); the durations of grazing exclusion therefore differed among sites. In our search for exclosure sites we chose areas that had been enclosed for as long as possible. Near the town of Mandalgobi, an airport fence created a contrast in grazing pressure along approximately 3 km of the fence. Along the fence, a gentle slope (generally less than 5°) contributed to the existence of an *Allium* community and an *Achnatherum* community (Sasaki *et al.* 2009a); we therefore used these as non-grazing plots. For one non-grazing plot in the *Caragana* community we used a fenced area around a TV tower. For the other non-grazing plot in the *Caragana* community and for the plot in the *Reaumuria* community we used two exclosures that we had established on a hilltop and in a lowland, respectively, in 2004.

### Interview survey

In Saintsagaan *sum* there are nine *bags* (villages). *Bags* 1–4 are located on rangelands and *bags* 5–9 are located in urban areas. Our study sites were located in *bags* 1–4. According to the local system of government, each *bag* consists of at least 100 pastoralist households. Each *bag* has one head, who liaises between government and local pastoralists. The head gathers information on the number of livestock belonging to each household and reports it to government. The head also visits local pastoralists and informs them about government decisions related to the rules of pasture use. These heads of the *bags* therefore knew the pasture conditions in the study area, and through frequent communication with the local pastoralists they recognized the movements and numbers of livestock belonging to each household in their *bag*. We

therefore interviewed the four heads of *bags* 1–4 on the rangeland of the Saintsagaan *sum*. Pastoralists in the Saintsagaan *sum* are often dispersed over wide areas, with some households moving more than 200 km from the study sites. It was therefore difficult to interview those pastoralists who had moved so far away. Although not many key informants were interviewed, these people knew the many movements of the local pastoralists within each *bag*. Moreover, to avoid data bias associated with the individual experiences of pastoralists, we considered that it was better to ask a “key person” instead of a selection of pastoralists.

The key informants were males aged from 45 to 60 years. We interviewed them individually in July and August 2010, using a semi-structured interview process to gather information on how local people used the different types of community during normal summers and drought summers. Informants were interviewed only once, but in the interview they were asked about pastoralists’ behavior in both types of summer. We asked each key informant to evaluate which communities were used for grazing in normal and drought summers. They allocated usage points to the communities, as follows: 0, not used; 1, rarely used; 2, often used; 3, mainly used. Communities used with the same frequency were given the same rank. We also asked the key informants to explain the reasons for their evaluations.

### Dung pellets counting

We set out five circles (diameter 1 m) within each grazing plot (20 × 20 m). The circles were placed at 5-m intervals diagonally across the plot. We counted the number of dung pellets (from goats and sheep) present within each circle in July to August 2010. We counted both old and fresh dung pellets. As each community had three grazing plots, the total number of circles was 60 (5 circles × 3 plots × 4 communities). Decomposition of dung pellets is very slow in Mongolia. According to Abaturov *et al.* (2008), annual loss of dung pellets’ weight was no more than 9–12%. Hence we supposed that the high number of dung pellets represents a long term, such as 10-year, measure of grazing impact.

### Vegetation sampling

We set out 20 quadrats (each 1 × 1 m) within each grazing plot and non-grazing plot for vegetation sampling in July and August 2010. The quadrats were placed at 3-m intervals in the plots. We identified all species present in the quadrats and estimated their cover. As each community had three grazing plots and one or two non-grazing plots (Table 1), the total number of plots was 17. Thus, a

total of 340 quadrats (20 quadrats  $\times$  17 plots) were examined.

## Data analysis

To examine how the key informants' evaluations for normal and drought summers differed by community, we used the Kruskal–Wallis test, because the data were not normally distributed. For both evaluations, the four key informants were treated as four replicates in the evaluation of each community.

To examine whether grazing impacts differed among communities, we compared the numbers of dung pellets among communities at the plot scale by using the Kruskal–Wallis test, because the data were not normally distributed. When the number of dung pellets differed significantly among communities, the Kruskal–Wallis test was followed by a *post hoc* contrast test (*t*-test with Bonferroni's correction).

To determine the response to grazing within each community, we used species composition data to calculate the degree of dissimilarity between grazing plots ( $n = 3$ ) and non-grazing plots ( $n = 1$  or  $2$ ) in each community by using quantitative Sørensen's index:

$$C_N = 2jN/(aN + bN)$$

where  $aN$  is the total cover in treatment *a* (grazing),  $bN$  is the total cover in treatment *b* (non-grazing) and  $jN$  is the sum of the lower of the two abundances, for each species present in both treatments (see Jost *et al.* 2011). This index takes into account both qualitative and quantitative changes in species composition. All dissimilarity values were calculated at the plot scale. Dissimilarity values between treatments were calculated for all pairs of plot between treatments.

We then compared these dissimilarities among four communities by using one-way analysis of variance (ANOVA) after confirming the assumption of homogeneity of variance. When there were significant differences among communities in the degree of dissimilarity, we applied one-way ANOVA followed by a *post hoc* contrast test (*t*-test with Bonferroni's correction) to examine which communities are different.

We compared each species' cover between grazing plot and non-grazing plot in each community at the quadrats scale by using Mann–Whitney *U*-test in order to examine which species increased or decreased by treatment. Species that had average cover values of at least 1% were included in the statistic.

The significance level for all tests was  $P < 0.05$ , and all statistical analyses were performed with R software (version 2.9.2, R Foundation for Statistical Computing, 2009, <http://www.R-project.org>, The R Foundation for Statistical Computing, Vienna, Austria).

## Results

### Key informants' evaluations of each community

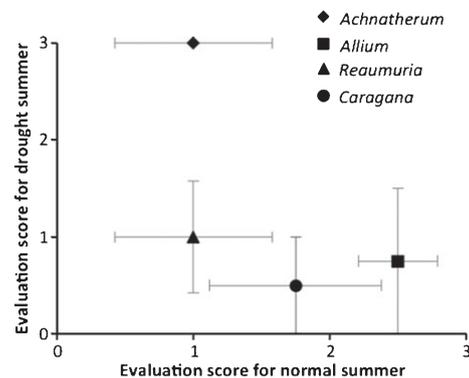
The key informants' evaluations of the four communities differed between normal and drought summers (Figure 1). Informants' evaluation scores did not differ significantly by community in the case of normal summers (Kruskal–Wallis  $\chi^2 = 5.19$ , d.f. = 3,  $P = 0.15$ ), but they did differ by community for drought summers (Kruskal–Wallis  $\chi^2 = 8.67$ , d.f. = 3,  $P < 0.05$ ; Figure 1). All informants gave the highest score to the *Achnatherum* community for drought (Figure 1). They explained that at this time *A. splendens* persisted but there was little vegetation in the other communities.

### Numbers of dung pellets in each community

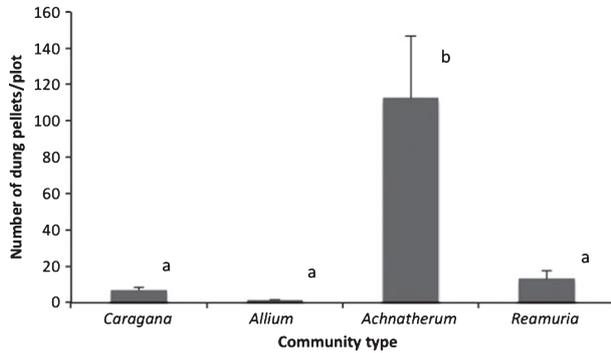
The numbers of dung pellets per plot differed significantly among communities (Kruskal–Wallis  $\chi^2 = 9.15$ , d.f. = 3,  $P < 0.05$ ; Figure 2). *Post hoc* contrasts indicated that the number of dung pellets was significantly higher in the *Achnatherum* community than in the other three (*Caragana*, *Allium* and *Reaumuria*) communities ( $P < 0.05$ ; Figure 2), but it did not differ significantly among the *Caragana*, *Allium* and *Reaumuria* communities.

### Impact of grazing on species composition in each community

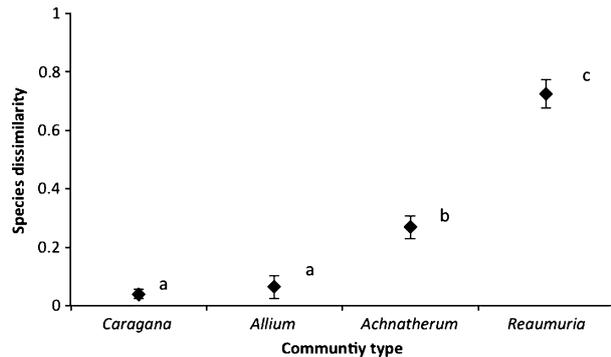
The degree of dissimilarity in species composition between non-grazing and grazing plots differed significantly among communities ( $F = 94.4$ , d.f. = 3,  $P < 0.001$ ; Figure 3). *Post hoc* contrasts indicated that the dissimilarity in species composition was significantly higher in the *Reaumuria* community than other three communities and significantly higher in the *Achnatherum* community than in the *Allium* and *Caragana* communities ( $P < 0.01$ ;



**Figure 1** Key informants' evaluation scores for communities during normal and drought summers. Error bars represent means  $\pm$  standard error (SE).



**Figure 2** Comparison of numbers of dung pellets among communities. Error bars represent means  $\pm$  standard error (SE). Letters indicate differences among communities at the  $P < 0.05$  level ( $t$ -tests with Bonferroni's correction).



**Figure 3** Comparison of species composition dissimilarity (Sørensen's index) among communities. Error bars represent means  $\pm$  standard error (SE). Letters indicate differences among communities at the  $P < 0.05$  level ( $t$ -tests with Bonferroni's correction).

Figure 3) but did not differ significantly among the *Caragana* and *Allium* communities.

The *Caragana* community and *Allium* community have species that decreased at grazing plots and both communities did not have species that increased at grazing plots (Table 2). *Achnatherum* community and *Reaumuria* community have species that increased or decreased their coverage at grazing plots. *Chenopodium album* L. increased their cover at grazing plots in the *Achnatherum* community and *Kalidium foliatum* (Pall.) Moq. increased their cover at grazing plot in the *Reaumuria* community (Table 2).

## Discussion

### Local pastoralists exploit resource variations

Our results showed that local pastoralists used different resources between normal summers and droughts (Figure 1). According to the key informants, during normal summers the pastoralists used all four communities,

which contained mainly small graminoids with highly palatable to livestock (Jigjidsuren and Johnson 2003), but during drought there were few small graminoids (e.g. Sasaki *et al.* 2009a). In drought the pastoralists therefore used the *Achnatherum* community (Figure 1), which retained high levels of vegetation cover (Sasaki *et al.* 2009a). *A. splendens* is highly tolerant of dry conditions, and indeed the cover of this species remains high during drought in the study area (Sasaki *et al.* 2009a). The *Achnatherum* community is usually found at the margins of saltmarshes in the bottoms of valleys where the groundwater is close to the surface (Grubov 1982), and its distribution range would be smaller than those of the other types of community. The results therefore suggest that, during drought, the grazing impact in the study area was concentrated on small areas such as those of the *Achnatherum* community.

### Grazing impact on each resource

Our results showed that grazing affected the vegetation differently in each community: that is, the species composition in each community responded differently to grazing. The number of dung pellets in the *Achnatherum* community was significantly higher than in the other three types of community (Figure 2). Thus, the result indicated that grazing pressure on the *Achnatherum* community was higher than on the others. The species composition dissimilarity between grazing and non-grazing plots of the *Reaumuria* community was highest in those of the other communities and those of the *Achnatherum* community was higher than those of *Allium* and *Caragana* communities (Figure 3). The main change between the grazing and non-grazing plots in the *Achnatherum* community was the appearance of *C. album* in the grazing plots (Table 2). *C. album* is an annual forb and weedy species that increases in abundance with grazing on Mongolian rangelands (Fernandez-Gimenez and Allen-Diaz 2001). *C. album* did not appear in the non-grazing plot but was significantly more abundant in the grazing plots ( $P < 0.01$ ; Table 2); its average cover was second highest after that of *A. splendens* among grazed plots of the *Achnatherum* community (average cover of *C. album* on grazing plots was 2.18%; Table 2). In addition, the cover of *A. splendens* was significantly lower in the grazing plots than in the non-grazing plot ( $P < 0.01$ ; Table 2). The covers of weedy annuals such as *C. album* are increased abruptly along grazing gradient in Mongolia (Sasaki *et al.* 2008a). The shift of plant composition from perennial grasses to annual forb along grazing gradients in Mongolia significantly decreased the metabolizable energy (a main index of nutritive value, calculated as the sum of digestible neutral detergent fiber, crude protein, fatty acids and non-

**Table 2** Results of Mann–Whitney *U*-test for the effect of grazing on the cover (%) of different species in the four different community types

Community and plant	Grazing	Non-grazing	Mann–Whitney <i>U</i>	<i>P</i>	Increased or decreased at grazing plots
<i>Caragana</i> community					
Perennial grass					
<i>Agropyron cristatum</i>	0.45	2.13	898	<0.05	Decreased
<i>Cleistogenes squarrosa</i>	1.60	1.97	1471	<0.05	Decreased
<i>Stipa glareosa</i>	0.45	1.42	1168	0.81	–
<i>Stipa kryrovii</i>	0.96	1.61	956	0.81	–
Perennial forb					
<i>Allium polyrrhizum</i>	1.29	0.03	1340	0.124	–
Annual forb					
<i>Chenopodium acuminatum</i>	1.31	4.00	457	<0.01	Decreased
<i>Salsolla collina</i>	2.22	5.87	822	<0.01	Decreased
<i>Allium</i> community					
Shrub					
<i>Caragana leucophloea</i>	0.08	1.27	378	<0.01	Decreased
Perennial forb					
<i>Allium polyrrhizum</i>	4.32	11.40	144	<0.01	Decreased
Annual forb					
<i>Bassia dasypylla</i>	0.17	1.22	35	<0.01	Decreased
<i>Salsolla collina</i>	0.24	3.55	14	<0.01	Decreased
<i>Chenopodium album</i>	1.95	0.09	705	0.201	–
<i>Achnatherum</i> community					
Perennial grass					
<i>Achnatherum splendens</i>	6.34	14.25	143	<0.01	Decreased
Perennial forb					
<i>Scorzonella divavicats</i>	0.01	1.78	–	–	–
Annual forb					
<i>Bassia dasypylla</i>	1.09	7.06	214	<0.01	Decreased
<i>Salsolla collina</i>	0.59	3.69	102	<0.01	Decreased
<i>Chenopodium album</i>	2.18	0.00	1020	<0.01	Increased
<i>Reaumuria</i> community					
Shrub					
<i>Kalidium foliatum</i>	6.23	0.00	1100	<0.01	Increased
<i>Reaumuria soongorica</i>	1.08	5.90	404	<0.01	Decreased
<i>Salsolla passerina</i>	0.09	9.85	1200	<0.01	Decreased
Perennial forb					
<i>Allium polyrrhizum</i>	1.51	4.33	16	<0.01	Decreased
Annual forb					
<i>Salsolla collina</i>	0.00	3.80	227	<0.01	Decreased

*n* = 60 for all communities under grazing. *n* = 40 (*Caragana* community) or 20 (other communities) for non-grazing plots.

fiber carbohydrates) (Sasaki *et al.* 2012). This means the types of vegetation changes seen at the *Achnatherum* community would lead to reduction of productivity.

On the other hand, the main change between the grazing and non-grazing plots in the *Reaumuria* community was the appearance of *K. foliatum* in the grazing plots (Table 2). In the case of the *Reaumuria* community, which was located in a depression landscape type in a halophytic shrubland, there were different dominant species within the same landscape type. One of the reasons was that the soils of this landscape type were salinized, with a high pH and electrical conductivity (Sasaki *et al.*

2008b), and differences in salinity may have influenced which vegetation species were dominant. Notably, *K. foliatum* and *S. passerine*, which dominated at the grazing plot and non-grazing plot, respectively, had high percentage covers in different places within the same landscape type (i.e. depression). Moreover, they were both halophytic shrubs belonging to the Chenopodiaceae, and they have the same functional traits based on Sasaki *et al.* (2009b). The number of dung pellets in the *Reaumuria* community were significantly less than the *Achnatherum* community (Figure 2), grazing pressure on *Reaumuria* community was less than the *Achnatherum* community.

Therefore we consider the species composition difference between grazing and non-grazing plots in the *Reaumuria* community would be affected by environmental factors such as salinity rather than grazing impact.

We showed that the number of dung pellets in the *Achnatherum* community and the species composition change with grazing were significantly higher in the *Achnatherum* and *Reaumuria* community than in the other communities (Figure 2). Although the dissimilarity of species composition at the *Reaumuria* community was highest in four community types, grazing pressure on the *Reaumuria* community was less than the *Achnatherum* community. Following these results, *Achnatherum* communities that were used by pastoralists during drought, had the highest number of dung pellets and their species composition was affected by grazing impact. Our study thus provided empirical support for the model of Illius and O'Connor (1999), namely that grazing affects key resource areas.

Numerous previous reports support the non-equilibrium model, suggesting that rainfall is a primary driver of vegetation dynamics (e.g. Dorji *et al.* 2010; Wesche *et al.* 2010; Cheng *et al.* 2011). However, these studies did not consider resource variations at a landscape scale in relation to the scale of local rangeland use, and the validity of scaling up the results of these studies to rangelands management is therefore questionable. According to Illius and O'Connor (1999), small key resource areas were affected heavily by grazing and large resource areas were less affected. Therefore, if resource variations are not taken into account, the impacts of grazing on vegetation are likely to be underestimated.

### Rangeland management in highly variable systems

Our experiment lacked replicated non-grazing treatments, except in the case of the the *Caragana* community, and the duration of grazing exclusion differed among communities, thereby limiting our ability to discern the effects of livestock exclusion on vegetation. For example, the low degree of species dissimilarity between grazing and non-grazing plots in the *Caragana* community would be influenced by the short duration of grazing exclusion. However, we investigated the impact of grazing on vegetation from several perspectives (interviews with key informants, and comparison of numbers of dung pellets and species composition dissimilarity among communities). We provided empirical evidence that resource variations, when related to the scale of local rangeland use, affects the impacts of grazing on vegetation.

Our results showed that the cover of *A. splendens*, which was present in highly productive areas (i.e. along drainage lines) (e.g. Ludwig *et al.* 2005), declined with

grazing. Thus, in order to develop rangeland management systems, future studies could be aimed at detecting the degradation threshold of *A. splendens*, for example, in terms of reduction in growth rate. At the same time, grazing controls, such as limitation of the number of livestock in key resource areas, are needed for preventive management, even on highly variable rangelands. According to the non-equilibrium model, management based on constant and conservative stocking rates is inappropriate and costly to pastoralists in such variable systems, as carrying capacities vary with precipitation (Behnke *et al.* 1993). However, we showed that grazing impact on the *Achnatherum* community was high, and because these areas are small their degradation can reduce livestock populations by diminishing the ability of an ecosystem to provide spatial buffers during climatic disturbance (Illius and O'Connor 2000). These previous results, together with our findings, suggest that if rangeland management is focused only on maintaining resource accessibility, then grazing-induced degradation of key resource areas can occur. Therefore, rangeland management needs to control livestock numbers, even in highly variable environments. Particularly in light of the fact that livestock numbers in the study area increased after privatization occurred in the 1990s (Saizen *et al.* 2010), numbers during normal summers need to be controlled to reduce grazing impacts on key resource areas during drought.

To our knowledge, our study provides the first empirical evidence that key resource areas are affected by grazing, even in a highly variable rangeland ecosystem. For management purposes, future studies are needed to detect the threshold degradation point of key resource areas in terms of not only species composition but also productivity. The linkage between grazing strategies and resource variations needs to be understood better if we are to develop sustainable rangeland management in semiarid regions.

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