

# Degraded rangeland dominated by unpalatable forbs exhibits large-scale spatial heterogeneity

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**Abstract** The spatial heterogeneity of vegetation and soil increases in response to land degradation caused by grazing mainly at a large spatial scale. This increase has been frequently associated with shrub invasion, but shrub invasion does not necessarily accompany land degradation. Instead, dominance by unpalatable forbs has been reported in some regions, but the spatial heterogeneity of such degraded rangeland has not been studied. We investigated the spatial heterogeneity of rangeland dominated by unpalatable forbs at a large spatial scale using Mongolian rangeland as an example. Spatial heterogeneity of the total vegetation cover and community heterogeneity were analyzed for three levels of land degradation. We found that the least-degraded site had homogeneous total vegetation cover and community, that the site with intermediate degradation exhibited low heterogeneity of the total vegetation cover but significant community type variation, and that the most degraded sites exhibited a periodic pattern of total vegetation cover as a result of a mixture of dense and sparse

patches of unpalatable forbs. These different responses can be used to assess land degradation levels and may have potential to monitor land degradation at a large scale by satellite images.

**Keywords** Land degradation · Geostatistics · Water runoff · Grazed grassland · Community heterogeneity · Mongolia

## Introduction

Land degradation in drylands causes an irreversible loss of productivity at a management time scale and is one of the most urgent global environmental threats (Adger et al. 2001). Shrub invasion, through which an initially homogeneous rangeland is invaded by shrubs and develops a mosaic of densely vegetated and bare patches (Schlesinger et al. 1996; Reynolds et al. 1997; Adler et al. 2001), is a typical example of how land degradation modifies the spatial heterogeneity of rangeland. This degradation often results from grazing (Rietkerk et al. 2000, 2002). Woody plants usually dominate the high-cover phases, whereas herbaceous elements may be either scattered in the low-cover phase or associated with the woody vegetation (Scholes and Archer 1997). Shrub patches form “islands of fertility” around which grass establishment occurs (Schlesinger et al. 1996; Bisigato and Bertiller 1999, 2004). The resulting change in spatial heterogeneity can have a marked effect on soil, water,

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and nutrient flows (Rietkerk et al. 2002); the nutrient distribution as a result of inputs of feces and urine (Augustine and Frank 2001); seed distribution (Bertiller and Bisigato 1998); and animal distribution as a result of the vegetation's structural defenses (Milchunas and Noy-Meir 2002).

The biotic feedbacks between animals and vegetation have also been considered to be major drivers that create spatial heterogeneity in grazed rangeland (Adler et al. 2001). Previously grazed patches are likely to be grazed again, leading to spatial heterogeneity in a rangeland as grazed and ungrazed patches develop ("patch grazing;" Hobbs and Swift 1988; Moen et al. 1998; Adler et al. 2001). A possible mechanism for the repeated grazing lies in the dietary preferences of the grazers (Mouissie et al. 2008) and regrowth of young, nutrient-rich shoots and leaves as part of the vegetation's recovery process. These processes create heterogeneity composed of a mixture of tall ("tussock") and short ("lawn") stands (Bakker et al. 1983; Posse et al. 2000).

However, the findings by these studies cannot be used to predict the spatial heterogeneity of degraded rangeland with the dominance of unpalatable forbs. In Mongolia, steppe and desert-steppe degradation is often characterized by an increase in the dominance of unpalatable forbs. Shrubs can mostly be ignored at such sites because of their low abundance (Sasaki et al. 2005, 2007; Hoshino et al. 2009). The spatial heterogeneity produced by shrubs has been explained based on their functional characteristics, such as deeper roots and greater stability against soil erosion (Reynolds et al. 1997), and neither factor applies to unpalatable forbs. In patch grazing, the current knowledge of processes that are based on grazing and high-palatability plants cannot be applied to the rangeland with the dominance by unpalatable forbs.

To obtain insights into the development of heterogeneity of rangeland with the dominance of unpalatable forbs, we studied the spatial heterogeneity of rangeland communities where land degradation is accompanied by increasing dominance by unpalatable forbs. Our primary aim was to identify whether the rangeland's spatial heterogeneity changed as a result of degradation. If spatial heterogeneity existed, this suggests that degradation may be occurring and can be used as an indicator of land degradation. Our primary aim is accompanied with the following hypothesis: because a combination of plant functional characteristics will be a

necessary compositional element for the process to cause spatial heterogeneity (Rietkerk et al. 2000), and because land degradation leads to the replacement of certain plant functional types (Sasaki et al. 2005, 2007; Hoshino et al. 2009), the processes that determine the spatial heterogeneity of vegetation should differ among different degrees of land degradation.

## Materials and methods

### Study site

Nomadic pastoralism is traditional and common in Mongolia. In 1992, when the socialist system was disrupted, a market economy was introduced (Fernandez-Gimenez and Batbuyan 2004). The number of livestock also increased because of the privatization of livestock (Fernandez-Gimenez and Batbuyan 2004). The infrastructure disruption caused nomadic pastoralists to concentrate in particular areas, such as water points, along roads, and in towns (Fernandez-Gimenez and Batbuyan 2004; Okayasu et al. 2007), in many cases leading to land degradation.

We selected an area near Mongolia's Hustai National Park (HNP) for our study. HNP is about 100 km from the capital city of Ulanbaatar. Near HNP, herders typically concentrate along the main road to Ulanbaatar so that they have access to various infrastructures. In contrast, grazing is strictly prohibited in HNP. This difference has produced a strong land degradation gradient around HNP, making the area suitable for studying land degradation caused by grazing and the associated changes in vegetation heterogeneity. Sheep comprise the highest proportion of the livestock community in the province (Tov) that includes HNP. They account for 48% of the total livestock number, versus 37% for goats, 9% for horses, 6% for cattle, and less than 0.1% for camels (National Statistical Office of Mongolia 2007).

### Field surveys

We selected study sites along the abovementioned land degradation gradient as follows. First, we excluded steep upslope areas to ensure similar environmental conditions at all sites. We established sampling plots at site 1 (105°49'00"E, 47°54'30"N; 3° slope) and site 2 (105°49'19"E, 47°55'48"N; 1°

slope) outside of the buffer zone surrounding HNP, in an area with high grazing intensity. We also established sampling plots at site 3 (106°04′04″E, 47°51′44″N; 0° slope) and site 4 (105°56′10″E, 47°47′16″N; 1° slope) inside the buffer zone around HNP, where grazing pressure was low to negligible. These four sites were chosen based on visual examination of the vegetation during our preliminary field trip to determine the community composition in terms of both plant palatability (i.e., the forage value) and the distance from the core area of HNP where grazing is strictly prohibited. Based on this inspection, we assumed that site 1 was most degraded, site 4 was least degraded, and sites 2 and 3 had intermediate levels of degradation. These judgments were preliminary, and we subsequently determined the actual degradation levels more precisely based on a field survey of land degradation indicators.

From 27 July to 15 August 2009, we established three 500-m-long transects at each site, separated by 50 m (i.e., at each site, the surveyed area covered 500 × 100 m). At intervals of 5 m along each transect, we established a 1 × 1 m plot, for a total of 300 plots per site and 1,200 plots in total. In each plot, we measured the cover by each species, excluding plants that covered less than 1% of the plot because our focus was on the dominant species. The position of each plot was established using a global positioning system (GPS) receiver. Soil sampling was also carried out in every fifth plot (i.e., at 25-m intervals) along each transect (i.e., 20 samples per transect, and a total of 60 samples per site). At each soil sampling plot, we obtained three composite samples at positions 25, 50, and 75% of the way along the diagonal of the plot, using a 100-mL sampling cylinder that was 5 cm in diameter by 5.1-cm deep. We used the samples to determine the soil organic matter content and soil texture class (clay, silt, and sand contents) according to the International Society of Soil Science (ISSS) standard.

We carried out an additional field survey on 8 August, 2011 to estimate the livestock grazing pressure. For each transect established on 2009, we established 1 × 1 m plots at intervals of 50 m (i.e., 10 plots × 3 transects = 30 plots per site). In each plot, we counted fecal pellets of sheep and goats. We excluded the dung of other animals because there were few examples. In addition, we interviewed two HNP rangers about the historical and current use of the pastures around our study sites.

## Analysis

To understand the spatial characteristics of the vegetation, we generally followed the method of Augustine (2003). That is, we quantified the spatial heterogeneity of the vegetation cover using semivariance analysis (Goovaerts 1997), and we performed cluster analysis and pattern diversity analysis to quantify the spatial heterogeneity of the plant community. Here, we defined the pattern diversity as 1 – [proportional similarity] (PS; Beals 1969); PS is defined later in this section. We then investigated the variation in soil attributes among the community types differentiated by cluster analysis using the classification and regression tree (CART) approach (Breiman et al. 1984).

### *Determination of the degradation level*

We evaluated land degradation using a combination of indicators based on the concept of ecological dysfunction, in which the mean residence time decreases for resources that enter the ecosystem or are produced by it; for example, soil erosion is a dysfunction because the soil produced by the ecosystem is lost (de Soyza et al. 1998, 2000). de Soyza et al. selected indicators to capture the process of shrub invasion and used the size and frequency of bare patches as a surrogate for soil erosion, shrub cover, and related factors. For the Mongolian plateau, Wiesmeier et al. (2009) and Lin et al. (2010) found that tussock grass facilitates the development of pools of nutrients such as soil organic matter at low levels of grazing disturbance, and that land degradation is initiated from the destruction and erosion of these pools. We therefore used the cover by tussock grass and perennial grass, the proportion of bare soil (1—total vegetation cover), and soil organic matter as key controls on the grassland nutrient pool in this region. In contrast, no ecological processes have been studied for degraded rangeland dominated by unpalatable forbs. Instead, we adapted some of the land degradation indicators based on vegetation composition that were used by Cheng et al. (2007) and Li et al. (2008). The balance between palatable and unpalatable plants can sensitively indicate the historical intensity of grazing (Bertiller et al. 2002). In this study, we classified palatability based on the criteria of Jigjidsuren and Johnson (2003). The relative cover by forbs

increases under heavy grazing in the study area (Sasaki et al. 2005). As indicators of grazing pressure, we used the number of fecal pellets of sheep and goats. We compared these indicators among the four study sites using pairwise *t* tests with Holm's adjustment. We used Spearman's rank correlation coefficient to test for a significant relationship between pellet count and the degree of degradation.

### Geostatistics

To quantify the spatial heterogeneity of the total vegetation cover for all plant species combined, we used semivariance analysis (Goovaerts 1997). We calculated the semivariance for each study site after explicitly merging the three transects for each site by accounting for their spatial coordinates. Linear, spherical, and dampened-hole models were assumed for fitting the semivariograms. These models were fitted using unweighted least-squares analyses, and the model with the smallest standard error was selected. The overall variation in the vegetation cover was determined using the sill (asymptote), and the mean spatial scale of the vegetation patchiness was found using the range, which represents the distance required to reach the asymptote. We tested for a significant relationship between the mean empirically derived semivariance of the vegetation cover (averaged as a function of distance) and the degree of degradation using Spearman's correlation coefficient ( $p < 0.05$ ).

### Community heterogeneity

To measure community heterogeneity, we first performed cluster analysis based on Euclidean distances and Ward's method for all the plots we surveyed. Clustering was terminated when 75% of the total information in the species  $\times$  plot matrix remained, and Wishart's (1969) objective function was used to define the information lost as agglomeration proceeded. We call the resultant clusters "community types" hereafter. The number of community transitions along a transect was calculated as the number of times the community type changed between two adjacent plots anywhere along the transect.

Next, we calculated the proportional similarity  $PS_{ij}$  between plots *i* and *j* as the sum of the relative species abundances (the proportion of total plants in a plot accounted for by the species) for species found in both

plots.  $PS_{ij}$  ranged from 0 (completely different species composition) to 1 (completely identical species composition). We then plotted a histogram of  $PS_{ij}$  for all combinations of *i* and *j* for each site and visually examined the differences in the community composition among the sites by examining the shape of the histogram. We also calculated  $PS_{ij}$  as a function of the distance between plots to measure the spatial scale of the community heterogeneity, and tested the differences between sites by pairwise *t* test with Holm's correction.

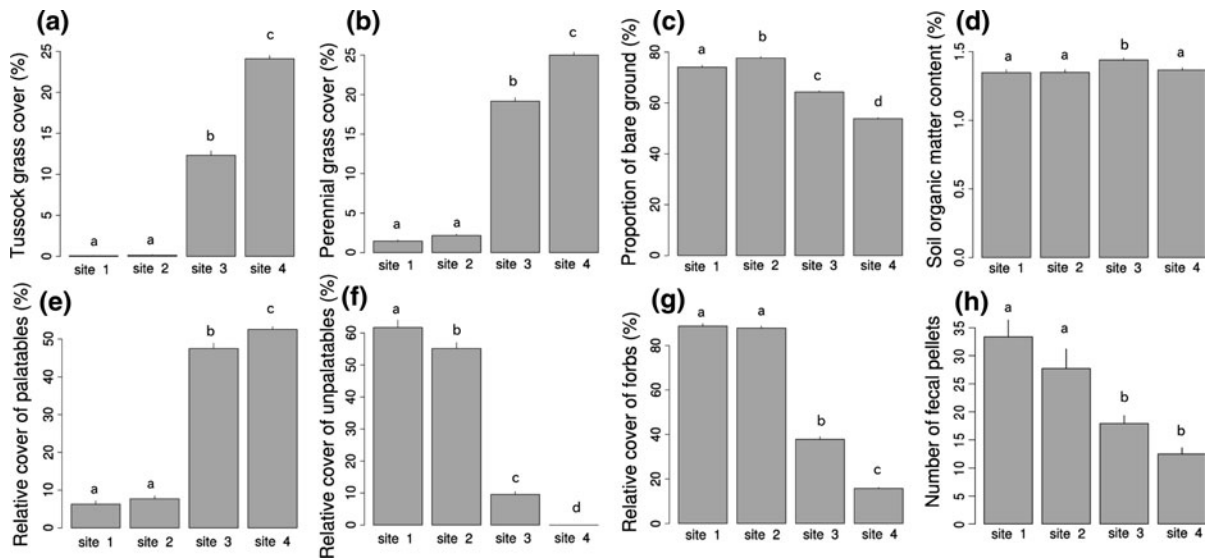
### Soil attributes

We used CART analysis (Breiman et al. 1984) to investigate the factors that controlled the soil attributes we sampled. For each target variable (i.e., soil organic matter, clay content, silt content, and sand content), we used the total vegetation cover and the site and community type derived by cluster analysis as the independent variables. We stopped the classification at the step in which the deviance calculated by 100 iterations of the *n*-fold cross-validation estimate was minimized.

## Results

Figure 1 shows the land degradation indicators for the four study sites. Significant differences between the study sites were tested by pairwise *t* test with Holm's correction at  $p < 0.05$  for all comparisons, unless otherwise indicated. Tussock grass cover and perennial grass cover, which can serve as surrogates of the site's ability to preserve soil nutrients, indicated that the two most degraded sites (1 and 2) did not differ significantly, but that the least-degraded sites (3 and 4) had significantly higher cover values. The proportion of bare ground differed significantly among all four sites. Soil organic matter was slightly but significantly higher at site 3 than at the other sites, which did not differ significantly. Vegetation palatability was significantly higher at sites 3 and 4 than at the other sites, with more abundant palatable plants at site 4 than at site 3. The proportion of unpalatable plants showed the opposite trend.

The number of fecal pellets decreased significantly from site 1 to site 4 (Spearman's rank correlation,  $p < 0.05$ ). A pairwise *t* test with Holm's correction



**Fig. 1** The means and standard errors of the indicators of land degradation: **a** tussock grass cover, **b** perennial grass cover, **c** the proportion of bare ground, **d** soil organic matter content, **e** relative cover of palatable plants, **f** relative cover of

unpalatable plants, **g** relative cover of forbs, and **h** the number of fecal pellets. Bars labeled with different letters differ significantly (*t* test,  $p < 0.05$ )

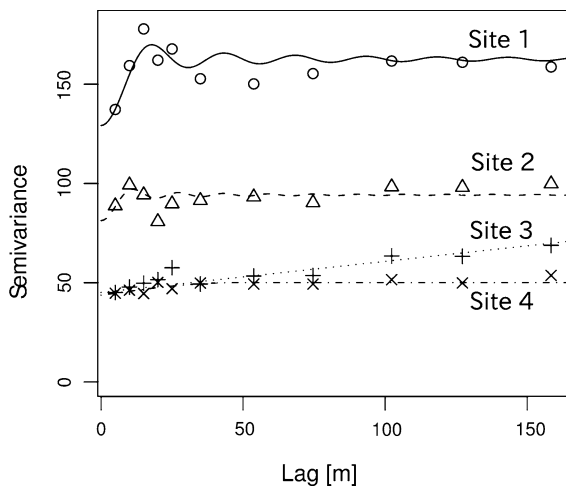
revealed that sites 3 and 4 did not differ significantly, whereas sites 1 and 2 differed significantly from each other, and both sites differed significantly from sites 3 and 4 ( $p < 0.05$ ). The rangers could not provide information on sites 1 and 2, which were far from the HNP core area they were responsible for. However, they informed us that site 4 (which was near HNP) was used as a winter camp because the narrow valley provided shelter, and that the users of the pasture (two herder families) had lived inside HNP until 2000, when they abandoned this pasture at the request of HNP; thus, at the time of our survey, this site had not been used for pasture for 9 years. The rangers told us that site 3 is still being used by several herders during both the winter and the summer, although the number of herders changes annually depending on rainfall. We found that most fecal pellets at site 4 were white, indicating that they were decomposed and that the site had not been used for grazing for some time, which is consistent with the information provided by the rangers. Based on this evidence, we believe that site 4 has sustained less grazing impact than site 3, despite the lack of a statistically significant difference in the numbers of fecal pellets.

Soil organic matter did not decrease significantly with increasing land degradation, possibly because the accumulation of fecal pellets at the degraded sites

counteracted any impacts of vegetation change and soil erosion; this may also explain why soil organic matter was significantly higher at site 3 than at site 4. Grazing-gradient studies have shown higher soil organic matter contents where animals concentrate because of the higher supplies of fecal pellets (Sasaki et al. 2007; Hoshino et al. 2009). On the other hand, a higher proportion of bare patches results in increased soil erosion, which decreases soil organic matter. Sasaki et al. (2007) also reported no correlation between soil organic matter content and grazing intensity, probably because of these counteracting effects.

In summary, we can divide the four sites into three degradation levels: sites 1 and 2 represent the highest level of degradation, site 3 represents intermediate disturbance, and site 4 represents the least disturbance. Sites 1 and 2 cannot be distinguished because there is little evidence of differences in their degradation levels. However, this does not affect the overall consequences of our study, as we will discuss below.

The mean empirically derived semivariance of the vegetation cover averaged through distance generally increased significantly with increasing degradation level (Fig. 2;  $p < 0.05$ , Spearman's rank correlation). For sites 1 and 2, our analysis suggested that the dampened-hole model was most appropriate



**Fig. 2** Empirical (data points) and fitted (lines) semivariograms for total vegetation cover at the four sites

**Table 1** Results of the fitted semivariogram for each site

Site	Optimal model	Nugget	Range	Sill
1	Dampened hole	129.21	23.23	142.17
2	Dampened hole	81.28	16.08	94.24
3	Linear	45.130	–	–
4	Spherical	43.74	47.08	50.624

(Table 1). This indicated that these sites exhibited a periodic spatial heterogeneity in the vegetation cover. The semivariances of sites 1 and 2 were distinctly larger than those of sites 3 and 4. For site 3, a linear model was most appropriate, whereas a spherical model was most appropriate for site 4. The wavelengths (ranges) of the periodicity at sites 1 and 2 were 23.2 and 16.1 m, respectively. We were not able to calculate the range at site 3, but the range at site 4 was much larger than the range at these sites, with a value of 47.1 m.

Cluster analysis distinguished eight community types. Table 2 shows the average frequency of the community types for each site for the three transects. At site 1, community types 1–4 were significantly more frequent than the other types (pairwise *t* test with Holm's correction;  $p < 0.05$ ). Community types 1 and 2 exhibited clear contrasts in their total vegetation cover (i.e., dense patches of *Artemisia adamsii* and sparse patches of *A. frigida*, respectively). Site 2 was dominated by community types 3 and 4, which are types with small amounts of palatable grasses and may

represent the invasion of community types 1 and 2 by these grasses or less-degraded versions of types 1 and 2. The contrast between *A. adamsii* and *A. frigida* is clear from the pattern diversity analysis described later in this section. Sites 1 and 2 generally exhibited similar combinations of community types, but site 2 included more plots with community types that contained small amounts of palatable grasses. Site 3 included some plots in all community types, but community type 6 was significantly more frequent than types 1, 2, 3, 7, and 8; this type was dominated by the most palatable grasses (*Stipa krylovii*), which was found at the least-grazed sites (Sasaki et al. 2005; Hoshino et al. 2009). In contrast, site 4 included only three community types (6, 7, and 8). Moreover, 78% of the plots were classified as community type 8, in which *S. krylovii* is dominant. The next most common community type at site 4 was type 7, whose species composition was similar to type 8 but with greater dominance by *S. krylovii*. Therefore, site 4 is considerably more homogeneous than the other sites in terms of the community types. The mean numbers of transitions among community types for each site were 62.3, 61.3, 59.0, and 28.3 for sites 1, 2, 3, and 4, respectively. The number of transitions at site 4 was significantly lower than at the other sites, which did not differ significantly. The cluster analysis and the community transition frequencies both suggested greater homogeneity at site 4.

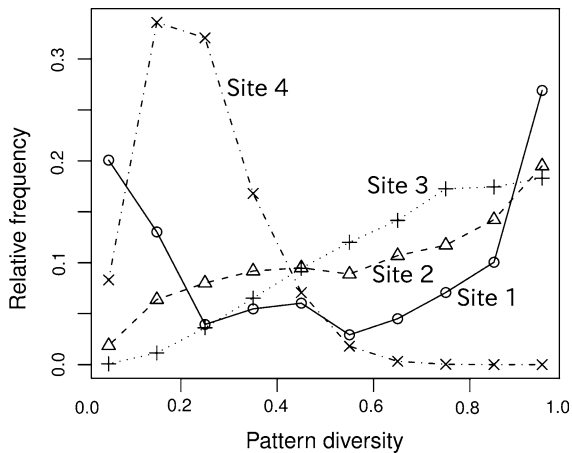
Figure 3 shows the frequency distribution for pattern diversity at each site. Site 1 exhibited a bimodal distribution, with peak values at pattern diversities of around 0 and 1. This bimodality means that in a comparison of any two arbitrary plots, the plots are either quite similar or quite different in terms of their species composition. Such a different species composition means that there were few common species, and that distinctly different plant communities coexisted at this site. Table 2 shows that the patches of *A. adamsii* and *A. frigida* at site 1 had fewer species in common, so the species composition must be similar within each patch and different between patches. In contrast, site 4 exhibited a unimodal distribution, with a peak at a pattern diversity of around 0.2. This indicated that in a comparison of any two arbitrary plots, the plots were likely to have a similar species composition. This finding was supported by the cluster analyses presented later in this section (i.e., most plots at site 4 were included in cluster 8; Table 2). Sites 2

**Table 2** Characteristics of the eight community types revealed by the cluster analysis

Community type	Mean cover (%)	Mean height (cm)	Mean number of plots that contained the specified community type				Palatability life cycle growth form (% of total)				Dominant species	Relative cover (% of total)			
			Site 1	Site 2	Site 3	Site 4	Pal	Mod	Unp	Per			Ann	Gra	For
1	40.46	18.09	32.3	11.3	1.3	0.0	2	4	94	99	1	2	98	<i>Artemisia adamsii</i>	93.5
2	18.76	20.78	20.3	16.0	8.7	0.0	9	82	9	95	5	13	87	<i>Artemisia frigida</i>	74.7
3	25.83	18.98	31.7	34.3	5.3	0.0	11	14	75	95	5	12	88	<i>Artemisia adamsii</i> <i>Elymus chinensis</i>	69.3
4	22.18	21.36	11.3	34.7	20.3	0.0	27	55	18	83	17	33	67	<i>Artemisia frigida</i> <i>Carex duriuscula</i>	17.7
5	28.89	25.74	4.3	3.3	19.0	0.0	53	29	18	95	5	66	34	<i>Artemisia adamsii</i> <i>Artemisia pectinata</i> <i>Poa attenuata</i> <i>Agropyron cristatum</i> <i>Cleistogenes squarrosa</i>	16.0
6	34.79	34.24	0	0.3	39.6	6.3	60	23	17	95	5	66	34	<i>Elymus chinensis</i> <i>Artemisia frigida</i> <i>Koehia prostrata</i> <i>Stipa krylovii</i> <i>Stipa krylovii</i>	9.5
7	44.80	40.49	0	0	4.3	15.3	72	14	14	92	8	85	15	<i>Artemisia adamsii</i> <i>Artemisia frigida</i>	8.7
8	46.69	39.87	0	0	1.3	78.3	48	17	35	83	17	77	23	<i>Stipa krylovii</i> <i>Stipa krylovii</i> <i>Koehia prostrata</i>	9.4

Only the most dominant species are listed (i.e., plants accounting for at least 70% of the total cover)

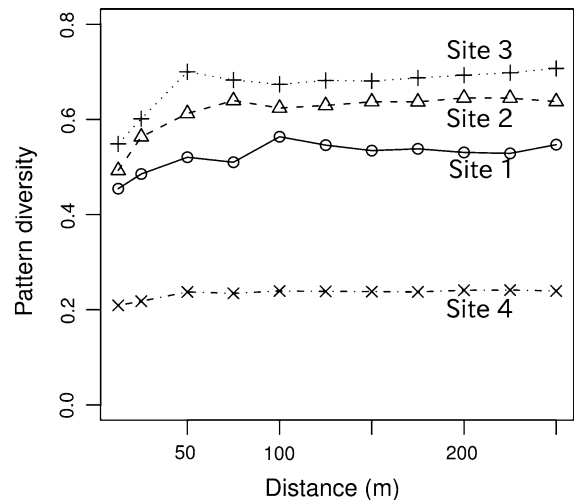
*Pal* palatable plants, *Mod* moderately palatable plants, *Unp* unpalatable plants, *Per* perennial plants, *Ann* annual plants, *Gra* grasses, *For* forbs



**Fig. 3** Frequency distribution for the pattern diversity of the four sites

and 3 had similar trends, without a prominent frequency peak. This indicated that any two arbitrary plots at these sites exhibited a wide variety of degrees of similarity in species composition. This is consistent with the clusters found at these sites. Table 2 shows that site 2 was dominated by moderately unpalatable and unpalatable species, but that the site also contained high-quality grasses. Site 3 had plots in all community types (Table 2). This high variety of species compositions provided various degrees of pattern diversity between any two arbitrary plots at sites 2 and 3, which resulted in the relatively flat shape of their frequency curves in Fig. 3.

Figure 4 shows the pattern diversity as a function of the distance between plots at each site. All sites differed significantly in the mean pattern diversity as a function of distance, but the mean was highest at site 3 because of the wide variety of communities at that site (Table 2). The pattern diversity reached an asymptote at a distance of 50–75 m at site 3, which indicated that most of the species could be found within a relatively small area. Sites 2 and 1 had the next-highest pattern diversities. Site 2 had higher values than site 1 because it included more high-quality grasses. More distance was required to reach an asymptote for these sites (around 100 m for both) than at site 3, because the high-quality grass was found less frequently at these more seriously degraded sites; this means that a larger area is needed to include these grasses. Pattern diversity was lowest at site 4 and did not increase markedly as the distance increased. This indicated that



**Fig. 4** Pattern diversity for each site as a function of the distance among plots

site 4 had relatively homogeneous vegetation, which is consistent with the results of our semivariogram and cluster analyses (Figs. 2, 3; Table 2) and is consistent with the fact that site 4 had the lowest value for community transitions (28.3 vs. more than 59 for the other three sites).

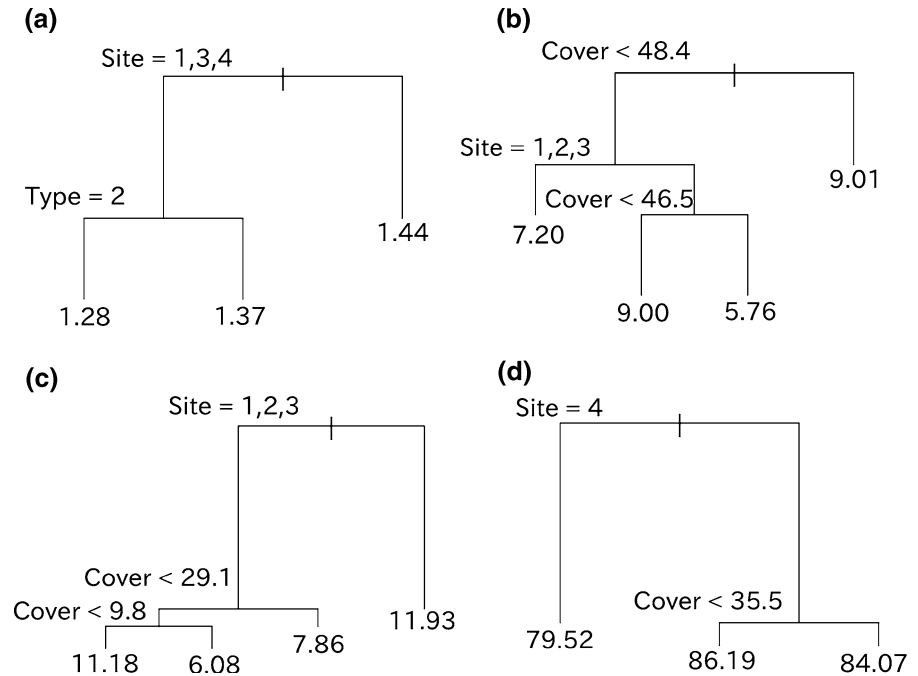
Figure 5 shows the results of the CART analysis. At sites 3 and 4, organic matter did not differ because site 4 had no plots containing community type 1. At sites 1 and 2, the *A. frigida* community (community type 2 in Table 2), with a low vegetation cover (19%), exhibited significantly lower organic matter content (Fig. 5a). The soil texture results show that site 4 was most homogeneous in terms of soil properties (Fig. 5b–d), and that the lower vegetation cover at the other sites increased the proportion of sandy soil; this suggests that higher vegetation cover was associated with increased clay and silt contents.

## Discussion

The spatial heterogeneity of the vegetation cover and community composition were lowest at site 4, which was the site with the lowest level of land degradation in our study area (Figs. 2, 3, 4). The spatial heterogeneity of the vegetation cover increased significantly from site 4 to site 1 ( $p < 0.05$ ). However, community heterogeneity responded differently to land degradation.



**Fig. 5** Results of the CART analyses for **a** soil organic matter, and for the soil **b** clay, **c** silt, and **d** sand contents. “Cover” represents the total vegetation cover (%) of each plot; “Type” represents the community type in Table 2. The number at the end of each branch represents the value of the soil attributes [e.g., soil organic matter content for (a)]



For the most degraded sites (sites 1 and 2), vegetation cover became more heterogeneous than at the other sites. Table 2 shows that the most degraded sites were generally dominated by patches of dense, unpalatable perennial forbs (e.g., *A. adamsii*; community types 1 and 3) and relatively sparse patches of moderately palatable perennial forbs (e.g., *A. frigida*; community types 2 and 4). Because the semivariograms for sites 1 and 2 showed periodicity (Table 1), these patches appear to periodically create spatial heterogeneity in the vegetation cover. These sites had a simple structure dominated by a periodic pattern of *A. adamsii* and *A. frigida* patches. The sparse *A. frigida* patches (community type 2) were associated with a lower soil organic matter content than in other patches, including dense *A. adamsii* patches (community type 1). This indicated that soil erosion from the sparse patches and accumulation in the dense patches maintained the spatial heterogeneity of the vegetation. Such positive feedback between vegetation abundance and soil erosion to maintain spatial heterogeneity is well known in shrublands, and it is related to land degradation accompanied by shrub invasion (Rietkerk et al. 2000, 2002; Bisigato et al. 2005). Our study found that even degraded herbaceous rangeland with few shrub patches can exhibit spatial heterogeneity. *A. frigida* in the sparse patches is more palatable than

*A. adamsii*, and its presence enhances the heterogeneity of the vegetation cover. As in the case of shrublands, differences in the physiological characteristics of *A. adamsii* and *A. frigida* may have contributed to the maintenance of spatial heterogeneity. *A. frigida* is competitively superior at sites with lower fertility (Chen et al. 2002; Gao et al. 2005), which explains its presence at sites 1 and 2 in our study, because these sites are likely to have lower fertility due to soil erosion, thereby favoring *A. frigida* over *A. adamsii*. Unfortunately, there have been no physiological studies for *A. adamsii*, probably because of the poor usefulness of this species for grazing.

The community types that contained tall tussock grass (community type 6, dominated by *S. krylovii*) and short grasses (community type 5, dominated by *Elymus chinensis*) were most frequently found at site 3: the combination of these two types of grass exhibited the typical spatial heterogeneity observed in grazed lawns (Posse et al. 2000; Adler et al. 2001; Mouissie et al. 2008). This type of lawn is produced by positive feedback between animal grazing and the vegetation as a result of patch grazing (Bakker et al. 1983; Posse et al. 2000; Adler et al. 2001). In our study, however, we found no clear evidence of this kind of feedback. To confirm the existence of such feedbacks, further detailed study will be needed to

collect data on more soil attributes, such as soil impedance and cations (Posse et al. 2000).

Soil texture did not show apparent trends as a function of the degree of land degradation or community types. Land degradation in arid rangeland had a marked impact on soil texture in some cases (Wang et al. 2006; Chen and Duan 2009), but many other studies showed no significant relationship (de Soyza et al. 1998, 2000; Rietkerk et al. 2000; Reynolds et al. 2007). The common features of the former cases are that the studies occurred in sandy land, which may indicate that soil erodibility or parent material may affect the response of soil texture to degradation. Abubakar (1997) pointed out that soil texture was most strongly affected by unsustainable cultivation. The change in soil texture by land degradation therefore depends on various conditions that did not appear to be significant in our study area.

In our study, we found that degraded rangeland dominated by unpalatable forbs showed spatial heterogeneity. In addition, our analysis showed that the spatial heterogeneities responded quantitatively differently (total vegetation cover) and qualitatively differently (community composition) to the level of land degradation. Our analysis also indicated that different processes may be responsible for spatial heterogeneity under the same geographical conditions as a result of the existence of a range of land degradation levels. Our findings suggest the possibility of monitoring land degradation accompanied by increased dominance of unpalatable forbs by means of remote sensing.

Spatial heterogeneity responds differently to grazing at different scales (Bertiller et al. 2002). For degraded rangeland dominated by unpalatable forbs, it is currently not known how the fine-scale spatial heterogeneity responds to grazing. We did not confirm how the patches are created by the land degradation processes, why these combinations of functional characteristics created a mixture of dense and sparse patches, and whether and how these patches contribute to the potential restoration processes.

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