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Spatial patterns of vegetation and soil fertility along a grazing gradient in a desert steppe in Inner Mongolia, China

by

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ABSTRACT

Spatial heterogeneities of vegetation and soil can strongly affect ecological processes in arid and semi-arid ecosystems. However, little is known about how those spatial patterns respond to grazing intensity in such systems. I studied how grazing intensity affect the spatial patterns of vegetation and soil nutrients at scales ranging from 0.1 to 18.7 m in a desert steppe in Inner Mongolia, China. Vegetation patches were more fragmented and homogeneous under higher grazing pressure. Heavy grazing also destroyed the spatial aggregation of plant species richness. Spatial heterogeneity of soil water and organic matter contents decreased along the gradient of increasing grazing intensity, while that of soil mineral N was first increased and then decreased along the grazing gradient. Both percent plant cover and power-law modeling could be used to indicate the risk of desertification associated with increasing grazing pressure.

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1. Introduction

Degradation and desertification has been a global threat to conservation, management and development of arid and semi-arid ecosystems (Schlesinger et al. 1990, Dodd 1994, Reynolds et al. 2007). Up to 73% of rangelands in these ecosystems have experienced some forms of degradation, including long-term reduction of productivity and biodiversity, loss of soil fertility through wind and water erosion, reduced water resources and, in some cases, salinization (Lund 2007). Due to the fragility of arid and semi-arid ecosystems, human utilization can easily surpass the natural carrying capacity of those ecosystems and lead to land degradation. To address the challenge of degradation and desertification, the interaction between human activities and ecosystem processes should be better understood.

Spatial patterns of vegetation and soil resources have been emphasized in recent studies because of their strong relationship with ecosystem functions (Schlesinger et al. 1996, Schade and Hobbie 2005, Maestre et al. 2006, Coppedge et al. 2008). Spatial vegetation patterns have been suggested to indicate desertification processes (Kefi et al. 2007). However, it is still unclear how human disturbances would change those spatial patterns. In this chapter, I will review the current understanding about the formation and functions of spatial patterns of distribution of vegetation and soil nutrients and their responses to grazing, which is the major land use type in arid and semi-arid ecosystems. I will also describe the research objectives and hypotheses of this M.Sc. thesis.

1.1 Spatial patterns in arid and semi-arid ecosystems

The distribution of plants, soil nutrients and animals commonly forms spatial patterns in natural ecosystems. For example, regularly isolated clusters of trees and shrubs are typical traits of savanna ecosystems, and tree lines and snow deposition can form striped patterns in sub-alpine forests (Rietkerk and van de Koppel 2008). Grazed arid and semi-arid ecosystems cover nearly 30% of the earth's terrestrial surface (HilleRisLambers et al. 2001). In these systems, plant communities can often be considered as two-phased mosaics consisting of a high-cover phase and a low-cover phase (bare land) (Aguiar and Sala 1999). Thus, the vegetation will be characterized by different patches at multiple scales. Vegetation patches vary in form, ranging from irregular mosaics to regular shape (such as "stripes", "labyrinths" and "spots"), and in sizes, changing from less than a meter to several hundred meters (see HilleRisLambers et al. 2001 for a review, Rietkerk et al. 2002). In these ecosystems, soil nutrients could also form spatial patterns at scales varying from the individual plant to landscape feature (Afzal and Adams 1992, Jaramillo and Detling 1992, Frank and Groffman 1998, Augustine and Frank 2001).

There are several major hypotheses explaining the formation of vegetation patterns in arid and semi-arid ecosystems. Some mechanisms emphasize the impact of disturbances such as fire, ungulate grazing and termite. For example, Bromley et al. (1997) suggested that vegetation mosaics started from complete cover and were formed by disturbances, which changed vegetation cover and created bare land. Other scientists attributed vegetation patterns to patterns of soil nutrients and properties (Boaler and Hodge 1962, Belsky 1986). All the above mechanisms are primarily based on conceptual models. In recent years, more and more studies prefer to use the so-called “scale-dependent feedbacks” mechanism to explain vegetation pattern formation based on simulation models (Lejeune et al. 1999, HilleRisLambers et al. 2001, Rietkerk et al. 2004, Rietkerk and van de Koppel 2008). This mechanism claims that localized facilitation among plants and competition for resources over a long range are able to generate regular or random vegetation patterns in arid and semi-arid ecosystems.

Vegetation patterns can significantly affect the spatial distribution of soil nutrients and other soil properties. Plants, especially shrubs and trees, have a strong positive effect on soil nutrients to form islands of fertility where nutrient availabilities are higher than in the surrounding bare land (Schlesinger et al. 1996, Reynolds et al. 1999). Multiple mechanisms explain the positive impact of plant coverage on soil properties, including deposition of litter in local area, root sequestration, unbalanced distribution of water caused by roots and symbiotic nitrogen-fixing bacteria in roots (Schlesinger and Pilmanis 1998). As a result of positive plant impacts, soil properties can show spatial patterns similar to that of vegetation (Schlesinger et al. 1996, Rietkerk et al. 2000, Schade and Hobbie 2005). Augustine and Frank (2001) found that ungrazed grasslands in Yellowstone National Park exhibited a high degree of patchiness in the distribution of soil N and N-mineralization rates (94 and 77% of total variation could be explained spatially, respectively) at small spatial scales (0.1-2 m). Rietkerk et al. (2000) proved that the variations of soil moisture content could be spatially explained (86%) in a savanna in West Africa. Overall, these studies suggest that the impact of vegetation on soil nutrient distribution mainly takes place at small scale (several meters or less), although other abiotic factors such as landscape position, precipitation and parent material also affect the spatial distribution of soil properties at larger scales (Belsky 1994).

Geostatistics have been widely applied to study the spatial variation of vegetation and soil variables (Schlesinger et al. 1996, Augustine and Frank 2001, Gallardo et al. 2006). Spatial patchiness of single variable could be studied by semivariance analysis, which examines the variance between measurements taken at increasing distance from each other and provides useful information with regard to the average size of patches and the degree of heterogeneity (Augustine and Frank 2001, Lane and BassiriRad 2005). Cross-correlogram that study the spatial correlation between two dependent variables provides an useful tool to examine how vegetation patches affect the spatial distribution of soil variables (Rietkerk et al. 2000). Near-surface aerial photos with high resolution have been recently used to construct maps of vegetation patches to study the spatial pattern of vegetation (Bar Massada et al. 2008). Compared to geostatistical methods, which require extensive sampling in a regular or fixed sampling matrix, photography method is more time-efficient in studying the vegetation patches.

Given that ecological processes are closely related to spatial patterns in arid and semi-arid ecosystems, the functions of natural spatial patterns have been recently emphasized in ecological research. Spatial vegetation patterns can influence seedling establishment (Tirado and Pugnaire 2003), pollination (Aguilar et al. 2006) and plant community composition (Facelli and Brock 2000, Joshi et al. 2006, Lopez et al. 2009). Vegetation patches serve as important habitats for animals such that changes in vegetation patchiness can greatly alter the community structure of soil biota and birds (Housman et al. 2007, Coppedge et al. 2008). Vegetation patches are more effective in trapping water and sediments than bare ground, protecting ecosystems from wind and water erosion (Reid et al. 1999, Puigdefábregas 2005, Descheemaeker et al. 2006).

It is well documented that spatial heterogeneity of soil nutrients can affect the coexistence of plant species, plant community structure and productivity (Day et al. 2003, Hutchings et al. 2003, Lundholm 2009). Therefore, localized nutrient accumulation under vegetation patches may provide important feedback to plant populations and communities. Spatial heterogeneity of soil nutrients can change soil microbial composition (Herman et al. 1995) and functions (Bennett and Adams 1999, Schade and Hobbie 2005, Gonzalez-Polo and Austin 2009). For example, Schade and Hobbie (2005) found higher net N mineralization, net nitrification and microbial biomass in islands of fertility than those in surrounding bare ground in Sonoran Desert, Arizona, USA. Spatial patterns of soil nutrients and microbial activities can have strong impacts on the C stock and greenhouse gas emission and influence the feedback of arid and semi-arid environment on climate change (Maestre and Reynolds 2006).

1.2 Grazing and spatial patterns of vegetation and soil

Due to the importance of the biotic and abiotic spatial patterns on ecological functions, there is an increasing body of literature documenting how disturbances, such as grazing, fire and climate change, affect those spatial patterns (Reynolds et al. 1999, Golodets and Boeken 2006, Coppedge et al. 2008). Livestock grazing is the main land use in arid and semi-arid rangelands and strongly modifies the properties and functions of ecosystems, such as forage production, diversity, community composition and soil fertility (Hobbs 1996, Marìa and Martìn 2001, Han et al. 2008). Contradicting results are usually reported with respect to the effects of grazing on the above properties and functions of ecosystems. For example, grazing has been shown to increase (Loeser et al. 2007, Olofsson et al. 2008), maintain (Stohlgren et al. 1999), or decrease rangeland plant diversity (Kruess and Tscharntke 2002). Grazing has also been shown to increase (Pineiro et al. 2009), maintain (Tracy and Frank 1998, Cui et al. 2005), or decrease soil organic carbon concentration (Su et al. 2005, Steffens et al. 2008).

Similarly, grazing has been found to present contrasting effects on the spatial patterns of vegetation in arid and semi-arid ecosystems (Berg et al. 1997, Adler et al. 2001, Olofsson et al. 2008). Grazing impacts can have great variation spatially based on the regime of grazing and properties of ecosystems (Hobbs 1996), and it can strongly affect the direction of change of spatial vegetation pattern (Adler et al. 2001). Rietkerk (2000) suggested that grazing intensity affected the spatial distribution of grazing and subsequently changed the spatial patterns of vegetation. However, few studies have examined the responses of spatial vegetation pattern to grazing intensity.

Grazing has been shown to increase (Afzal and Adams 1992, Rietkerk et al. 2000) or decrease (Augustine and Frank 2001, Wiesmeier et al. 2009) spatial heterogeneity of soil properties in arid and semi-arid ecosystems. Grazers have direct impacts on soil properties by tramping and wallowing, which can cause soil compaction (Knapp et al. 1999), and by adding excreta to change nutrient availabilities (Tolsma et al. 1987). Grazers can also indirectly affect spatial distribution of soil properties through changing substrate input through plant roots and litter, altering the spatial distribution of plant species composition and/or affecting the spatial patterns of other organisms such as insects (Augustine and Frank 2001). The interaction between spatial patterns of vegetation and soil is important in understanding the role of grazing on arid and semi-arid ecosystems; however, most previous studies only focused on one aspect of the interaction. Moreover, few studies have attempted to understand how grazing intensity affects this interaction.

Spatial patterns of vegetation and soil can also affect grazing behavior. For example, Hester et al. (1999) found that sheep prefers small-size grass patches in a heather moorland in Scotland. Grazers can create patches that maintain a higher forage quality and plant growth rate. These patches, known as “grazing lawns”, have a higher possibility to attract grazers to feed on them again (Dutoit 1990). Forming grazing lawns can provide positive feedback between grazing and re-grazing. In contrast, this feedback can become negative when grazers destroy the spatial patterns of vegetation and soil. If grazing intensities induce different patterns of vegetation and soil distribution, the feedback between grazing and re-grazing is likely to change accordingly. Therefore, understanding the relationship between grazing intensity and spatial patterns is essential for sustainable management of arid and semi-arid ecosystems.

Recent studies have suggested that changes in vegetation pattern can have direct application in rangeland management to indicate suitable grazing intensities and warning signals for desertification (Schlesinger and Reynolds 1990, Kefi et al. 2007). Kefi et al. (2007) found that the sizes and numbers of vegetation patches follow a power-law distribution under low grazing pressures in three different Mediterranean arid and semi-arid ecosystems, but the patch-size distribution would deviate from standard power law under higher grazing pressures. Based on this phenomenon and model simulation results, they suggested that the deviation from power-law distribution could serve as a predictor of early desertification. However, Maestre and Escudero (2009) did not observe such deviation along a gradient of increasing desertification; instead they found that percent plant cover could be used as an indicator for desertification. Studying the impacts of grazing intensity on spatial vegetation pattern can provide an opportunity to test the suitability of these predictors.

The Inner Mongolia Steppe is the main part of the Central Eurasian Steppe region and the biggest continuous grassland in China. Overgrazing has caused severe land degradation and eventually desertification in this region (Su et al. 2005, Zhao et al. 2005, Zheng et al. 2005). Compared to other types of grasslands, desert steppe accounts for 39% of total native grassland in Inner Mongolia and has a higher susceptibility to overgrazing (Li et al. 2000). Thus, desert steppe is a great model ecosystem to study the grazing-induced desertification. Choosing a suitable grazing intensity and identifying early signals for desertification are extremely important for sustainable rangeland management in this type of desert steppe.

1.3 Research objectives

As discussed above, there is still a knowledge gap regarding how disturbances (such as grazing) affect spatial distributions of vegetation and soil resources in arid and semi-arid ecosystems. Little is known about the role of grazing intensity in modifying the influences of livestock grazers on those spatial patterns. The performance of spatial vegetation patterns as a desertification indicator has not been tested in a wide range of ecosystems.

The central research question of this M.Sc. thesis is how grazing intensity affects the spatial patterns of vegetation and soil fertility in a desert steppe in Inner Mongolia, China. The specific research objectives are:

1) To determine how vegetation characteristics (such as aboveground biomass and height) and soil nutrient availability (such as soil organic carbon and mineral nitrogen) change along a grazing gradient (Chapter 2)

2) To determine how the characteristics of vegetation and soil patchiness change along the grazing gradient (Chapter 2 & 3)

3) To determine the spatial correlations between vegetation metrics and soil nutrient availabilities and their responses to grazing intensities (Chapter 2), and

4) To determine whether percent plant cover and the patch-area distribution of vegetation can indicate the desertification process associated with increasing grazing pressure (Chapter 3)

For the above objectives, I hypothesized that:

1) Increasing grazing intensity decreases biomass and height of the vegetation and soil nutrient availability

2) Grazing intensities have different impacts on patchiness of the vegetation and soil properties. Compared to ungrazed exclosures, heavy grazing decreases spatial heterogeneity of the studied variables, while light grazing would maintain those spatial heterogeneity

3) Spatial distributions of soil nutrients are correlated with those of vegetation metrics under no or light grazing pressure and heavy grazing weakens these correlations, and

4) Patch-area distribution of vegetation is suitable for detecting signs of desertification

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2 Grazing intensity affected spatial patterns of vegetation and soil nutrients in a desert steppe, Inner Mongolia, China

2.1 Introduction

The distribution of plants, soil nutrients and animals commonly forms spatial patterns in natural ecosystems. Since ecological processes are tightly related to the spatial patterns of biotic and abiotic factors at different scales, the functions of spatial patterns are recently emphasized in ecological research. Spatially structured vegetation patterns can influence the coexistence of plant species (Kneitel and Chase 2004, Lopez et al. 2009), community stability (Dayton et al. 1992, Frelich and Reich 1995) and ecosystem function (Lovett et al. 2005, Maestre et al. 2005). Soil spatial heterogeneity can also affect the distribution and coexistence of plant species (Fransen et al. 2001, John et al. 2007) and interact with global change factors to further influence ecosystem properties and processes (Maestre and Reynolds 2006).

Due to the important ecological function of spatial patterns, there is an increasing interest on the processes that can alter spatial patterns of biotic and abiotic factors. As a major land use type, grazing strongly modifies ecosystem processes and presents two contrasting effects on the vegetation and soil in arid and semi-arid environments (Milchunas and Lauenroth 1993, Augustine and McNaughton 1998, Frank and Groffman 1998). On one hand, grazing offers a management tool to maintain primary production, biodiversity and habitat structure (McNaughton 1979, Hobbs 1996, Collins et al. 1998). On the other hand, improper use of grazing, such as overgrazing, can dampen productivity, cause severe loss of soil nutrients and further induce deterioration of ecosystems, including desertification (Rietkirk and van de Koppel 1997, Su et al. 2005, Han et al. 2008). Therefore, different grazing regimes (e.g., different grazing intensities) should reveal divergent impacts on spatial patterns of vegetation and soil. However, there have been few studies to examine the impacts of grazing intensity on spatial patterns of vegetation and soil in arid and semi-arid ecosystems (except Rietkerk et al. 2000, Bisigato et al. 2005).

In arid and semi-arid ecosystems, the spatial distribution of plant communities can often be considered as a two-phased mosaic consisting of a high-cover (vegetation patch) phase and a low-cover phase (bare land) (Aguiar and Sala 1999). This patchy pattern of vegetation distribution indicates strong spatial dependence; in other words, the vegetation variable at any point can be reasonably predicted by the variable at an adjacent point. Significant spatial dependence has been linked to spatial heterogeneity and weak or non-significant dependence indicates spatial homogeneity; spatial homogeneity describes spatially random, not a spatially uniform data points (Adler et al. 2001). Grazing can either increase or decrease the spatial heterogeneity of vegetation (Glenn et al. 1992, Rietkerk et al. 2000, van de Koppel et al. 2002, Olofsson et al. 2008). Studying the effects of grazing intensity on vegetation spatial heterogeneity can help to understand the above results and improve rangeland management due to the important functions of vegetation spatial pattern.

Grazing may also affect the spatial patterns of soil properties in arid ecosystems. Grazers have direct impacts on soil properties through tramping and wallowing that can increase soil compaction (Knapp et al. 1999), and change nutrient distribution via excreta input (Augustine and Frank 2001). In addition to direct impacts, grazers can have indirect influences on spatial distribution of soil properties through changing vegetation patterns. Some studies found that soil nutrients accumulated under vegetation patches, forming the so-called “islands of fertility” (Schlesinger et al. 1996, Reynolds et al. 1999). Schlesinger et al. (1996) suggested that overlapping of vegetation and soil nutrient patterns was due to the positive effect of plant on soil nutrients and Olofson et al. (2008) found that the grazers’ impact on soil nutrient heterogeneity can be consistent with their influence on vegetation patterns. However, in other studies the response of vegetation patterns to grazing did not correlate with changes in soil nutrient pattern (Rietkerk et al. 2000, Augustine and Frank 2001). Little is known about how grazing intensity affect the relationship between spatial patterns of vegetation and soil nutrients.

In this study I examined the impact of sheep grazing intensity on the vegetation, soil nutrients and their spatial heterogeneity in a desert steppe by combining a spatial sampling design with geostatistical analyses. I also examined the relationship between spatial patterns of vegetation parameters and soil nutrients and the response of this relationship to the grazing gradient. The studied scale (0.1-18.7 m) was fine enough to provide more detailed information about the local interaction between plant and soil than most previous research in Inner

Mongolia steppe, where a minimum of 1 m sample distances were used (Bai et al. 2002, Chen and Zeng 2004, Su et al. 2006).

I hypothesized that 1) vegetation characteristics and dynamic soil variables (such as soil mineral nitrogen and soil water content) are more likely to be influenced by grazing intensity than soil organic carbon (SOC), total nitrogen (TN) and C/N ratio, 2) light and heavy grazing have different impacts on spatial patterns of the studied parameters; and 3) spatial distributions of soil nutrients are correlated with that of vegetation and such correlations are affected by grazing.

2.2 Materials and Methods

2.2.1 Study Site

The study site (41°47’17”N, 111°53’46”E) is located in Siziwang Banner, Inner Mongolia Autonomous Region (IMAR), China. This site is around 140 km north of Huhhot, the capital city of IMAR. The study area has a mean annual temperature of 3.4 °C with the highest monthly mean temperature in July (24.0 °C). Long-term mean annual precipitation is 280 mm. In recent years, the annual precipitation (2006, 161 mm; 2007, 162 mm; 2008, 230 mm) was relatively low at this site. The climate is arid, with windy and dry springs (March to June, with 49 mm average rainfall), and warm and comparatively rain-rich summers (July to September, with 220 mm average rainfall). Average moisture index (the ratio between precipitation and water loss) ranges from 0.15 to 0.30. The vegetation is of typical desert short-grass steppe, dominated by *Stipa breviflora* Griseb., *Artemisia frigida* Willd. and *Cleistogenes songorica* Roshev..

Other species that are common include *Convolvulus ammannii* Desr., *Artemisia pectinata* Pall., *Kochina prostrata* (L.) Schrad, *Caragana stenophylla* Pojark., *Leymus chinensis* (Trin.) Tzvel. and *Salsola collina* Pall.. Vegetation cover averages 20%. The soil is a Brown Chernozemic (the Canadian System of Soil Classification) or a Kastanozemic soil (the FAO System of Soil Classification).

2.2.2 Experiment Design

In July 2002, ~ 35 ha steppe, which used to be grazed (stocking rate: 6-10 sheep-unit-month ha^{-1}) year round under collective sheep grazing for over 30 years, was fenced for this experiment. In 2004, this site was divided into two blocks and an experiment with a randomized complete block design was established so that each block contained four pastures (Fig. 2.1). In total, there were 8 pastures, each with an average area of 4.4 ha. Four treatments (CK: control; LG: lightly grazed; MG: moderately grazed; HG: heavily grazed) were randomly applied within each block. The stocking rates were 0 (CK), 5.46 (LG), 10.92 (MG), 16.26 (HG) sheep-unit-month ha^{-1}. Grazing period was around six months per year, from early April to early October. The grazing gradient covered from grazer exclusion to heavy grazing and represented the most common grazing pressures in this region (Wei et al. 2000, Jiao et al. 2006).

To capture the spatial structures of soil and vegetation parameters, I applied a sampling matrix consisting of 177 points in each pasture in 2008 (Fig. 2.1). All of the sampling matrices were located on a relatively level ground. The matrix contained two parts. The first part was a 4×4 m grid where sampling points were established every 1 m (resulting in 25 points). Then, three 1×1 m grids within the 4×4 m grid were randomly chosen and five additional sampling points were located with a 0.5 m interval in each chosen 1×1 m grid. Then one 0.5×0.5 m grid was randomly chosen in the each selected 1×1 m grids and another 24 points were established by every 10 cm interval within each 0.5×0.5 m grid (Fig. 2.1). This defines the fine scale sampling. At the coarser scale, 65 points were evenly distributed within a 30×10 m grid by a 2.5 m interval (Fig. 2.1). The 30×10 m grid was established 2 m south of the 4×4 m grid with the 30 m border paralleling the long edge of the pasture. The coarse scale samples consisted of the 65 samples in the 30×10 m grid and the 25 samples in the 4×4 m grid with 1 m sampling interval as described above.

2.2.3 Sampling and Sample Analysis

Sampling was done in mid-August 2008 that corresponded to peak biomass in the growing season. Timing of sampling allowed me to compare my results with some previous research in Inner Mongolia (Bai et al. 2002, Su et al. 2005). Within a 5 cm radius of each sampling point, canopy height and species richness were measured. Then AGB was harvested and later oven-dried at 65 °C for two days. In-situ volumetric SWC in the top 6 cm was measured with a ThetaProbe ML2X (Delta-T Devices Ltd, Cambridge, UK; ±1% accuracy) at each sampling point. After the SWC measurement, the 0-6 cm soil was sampled by a 4 cm (ID) auger. Each sample was temporarily stored in coolers and passed through a 2 mm sieve in the same day of sampling. A set of sub-samples was kept frozen after sieving for measuring soil mineral N.

The NH_4^+ and NO_3^- concentrations in the frozen soil sub-samples were extracted by 2 mol L^{-1} KCl at a 1:3 ratio (w:v) and analyzed by a flow injection analyzer (FIAstar 5000, FOSS Analytical, Höganäs, Sweden). Another set of sub-samples were air-dried, ground to powder for total C and N analyses. Soil organic C was measured by the dichromate oxidation method (Nelson and Sommers 1982). Soil total N was determined using micro-Kjeldahl digestion (Nelson and Sommers 1980). Soil C/N ratio was calculated by the ratio between SOC and TN.

2.2.4 Data Analyses

Mean values and coefficient of variance (CV) of AGB, height and soil properties were compared among grazing treatments by one-way ANOVA. The CV was also calculated separately for the fine scale and coarse scale samplings. If the grazing effect was significant, a post hoc comparison of means was done with a Tukey's HSD test. The Kruskal-Wallis test was used to examine the grazing effects on species richness. A significance level of $\alpha = 0.10$ was used for all analyses. All the above analyses were carried out in SPSS software version 11.5.

A principal component analysis (PCA in SPSS software, version 11.5) was used to explore the correlation among vegetation and soil variables. To improve the resolution of PCA diagram, I randomly chose 50 sampling points from each pasture to form the data set (consisting data points at both fine and coarse scales) for PCA analysis. This data set showed a similar separation among treatments as the whole data did. The solution was rotated with the varimax method to maximize the variance of factor loading.

The spatial pattern of soil and vegetation distribution was evaluated by semivariance analysis, which assesses the variance between measurements taken at increasing distance from each other (known as lag distance) (Schlesinger et al. 1996). Before the analysis, each data set was tested for normality and logarithmic transformation was applied if necessary. When this transformation did not improve normality, a Box-Cox transformation was carried out (Box and Cox 1964). Semivariance γ for a certain lag distance h is calculated as:

$$\gamma(h) = \frac{1}{2N(h)} \sum_{i=1}^{N(h)} \left[y(x_i) - y(x_i + h) \right]^2$$

where $N(h)$ is the total number of data pairs that are separated by h, $y\ (x_i)$ is the value of studied variable at position x_i, and $y\ (x_i + h)$ is the value of variable at a distance h from x_i.

In each pasture, random, linear, spherical, exponential or Gaussian models were fitted to the semivariance-lag distance distribution by using an un-weighted least-squares analysis in GS+ version 7.0 software (Robertson 2000). The model that minimized the reduced sum of squares was determined as the best-fit model. Data exhibiting no spatial patterns (where the semivariance forms a flat line) were fitted into a random model. For data with patterned distribution, the semivariance is relatively small at short lag distances, suggesting neighboring samples are more similar and autocorrelated, and increased when paired samples become less similar. Semivariance then reaches an asymptote at range (A), the distance over which sample variances are spatially autocorrelated. Nugget variance (C_0), which is defined as the variance at zero lag distance, reflects either the error associated with measurement and/or analysis or the variances that can be spatially explained at a scale smaller than the minimal lag distance. The difference between total variance (also known as sill variance, $C+C_0$) and nugget variance is the spatial variance (C). Magnitude of spatial heterogeneity (MSH) can be measured by the proportion of total variance that could be spatially explained ($C/[C+C_0]$) (Lane and BassiriRad 2005). A fitted model with a range larger than the active lag distance suggests that there is a large scale trend in the sampled space. To remove the trend, trend surface analysis was used by treating x and y coordinates, their interaction and higher order terms (up to 4th order) in a multiple regression analysis with the studied parameter as the dependent variable (Gallardo et al. 2006). Residuals from the significant multiple regression analysis were used for semivariance analyses and the isotropic semivariogram was chosen.

For the fine scale sampling design, the active lag distance was 2.7 m and the lag interval was 0.1 m, which was the minimum distance between sample points. The minimum and maximum number of pairs in a lag interval was 51 and 296, respectively, with a mean of 166 pairs. For the coarse scale sampling design, the active lag distance was 18.7 m and the lag interval was 1.0 m. The number of pairs per lag interval ranged from 40 to 319 (mean = 182). Range and MSH were compared among grazing intensities by one-way ANOVA (SPSS, version 11.5). Linear regression analysis (SPSS, version 11.5) was also used to determine whether grazing intensity was a significant predictor for the geostatistical metrics.

ANOVA and regression analysis were done separately for the fine and coarse scale data.

Cross-correlogram (GS+ software, version 7.0) plotted the cross-correlation $r_{ab}(h)$ between two dependant variables (a and b) as a function of increasing lag distance h and was used to compare the spatial patterns of a and b. Value of $r_{ab}(h)$ was calculated as:

$$r_{ab}(h) = \frac{\sum_{i=1}^{N(h)} a(x_i)b(x_i + h) - m_a m_b}{s_a s_b}$$

where $a(x_i)$ represents the value of studied variable a at position x_i, $b(x_i + h)$ is the value of variable b at position $x_i + h$, and m_a and s_a denote the sample mean and standard deviation of a, respectively. Identical spatial patterns of a and b equate $r_{ab}(h)$ with 1, opposite patterns equate $r_{ab}(h)$ with -1 and total unrelated patterns make $r_{ab}(h) = 0$. Critical values for Pearson correlation coefficient r were used to evaluate the significance of $r_{ab}(h)$ at each lag interval.

Spatial Analysis by Distance Indices (SADIE) was designed to study count data (Perry et al. 1996) and was done to examine the spatial arrangement of plant species richness in this study. SADIE measures the distance to regularity (D), which represents the minimum total distance in space that the count of studied parameter would have to move spatially so that all the sampling points have the same count. By randomly permuting count data in space, D can be recalculated and an average D can be obtained if the permutation repeats. The index of aggregation, I_a, was defined as the ratio between observed D and average D obtained from permutations. The probability of rejecting the null hypothesis that the spatial distribution of studied parameter is random can be calculated by the percentage of permutations, in which the recalculated *D* is higher than the observed *D*. In this study, I_a of plant species richness and its significance was calculated separately at the fine and coarse scale by using SADIEShell v1.2.2 software with 2340 permutations (http://www.rothamsted.ac.uk/pie/sadie/). A significant I_a with value above one suggests that plant species richness is spatially aggregated. A random pattern of species richness has an I_a around one and a regular pattern has an I_a less than one.

2.3 Results

2.3.1 Mean Values and CV

Mean AGB was affected by grazing intensity; biomass decreased from 207±32 (mean±SE) in CK to 70±32 g m^{-2} in HG (Fig. 2.2a). Grazing intensity also affected vegetation height, which was 57 and 54% lower in HG than in CK and LG, respectively (Fig. 2.2b). The LG treatment had the highest C/N ratio in the soil among all grazing intensities (Fig. 2.2c). Average soil NH_4^+ concentration in LG (3.71±0.11) was higher than those in MG (3.35±0.11) and HG (3.33±0.11) (Fig. 2.2e). Grazing intensity did not affect SWC, soil NO_3^- (Fig. 2.2d, 2.2f) or species richness (data not shown).

When CV was calculated with the whole data set (including both fine- and coarse-scale samples) in each experimental plot, AGB and height presented higher variances than any other variables and their CV values were not affected by grazing intensity (Fig. 2.3). In MG plots, CV of soil NO_3^- (0.241±0.011) was lower than that in CK (0.311±0.011) and LG (0.323±0.011) (Fig. 2.3f). Grazing effect was also significant for soil NH_4^+ (*P* = 0.074) so that CV in HG was lower than that in LG (*P* = 0.076 and Fig. 2.3e). Grazing did not affect CV of any other soil variables (Fig. 2.3). For most variables, CVs at the fine and coarse scale were similar in magnitude except that TN had a larger CV at the coarse scale (*P* = 0.075, data not shown).

2.3.2 PCA

Component 1 in the PCA explained 32.7% of the variance and component 2 explained 14.1%, for a total of 46.8% variance explained. Component 1 mainly described the differences among samples related to aboveground biomass and height and component 2 separated samples by SOC and TN (Table 2.1). While Component 3 recognized variations in NH_4^+ and NO_3^-, it was not kept because of its low eigenvalue (0.972). Visualization of PCA results showed that most samples from HG had negative scores for components 1 and 2 and were separated from samples of both CK and LG (Fig. 2.4). The MG samples had a similar range of the component 1 score as HG, while it was not separated from HG because of its large range of component 2 score (Fig. 2.4).

2.3.3 Geostatistics

At the fine scale. There was a significant negative linear relationship between range for plant AGB and grazing intensity (*P* = 0.020, R^2 = 0.562, Fig. 2.5a). Spherical models were the best-fit for AGB in two CK pastures with a mean range of 1.02 m (Table 2.2 and Fig. 2.6). In contrast, AGB in HG either fitted a random model, indicating no spatial pattern in the studied scale, or an exponential model with a 0.18 m range, suggesting spatial patterns happened in a small area (Table 2.2 and Fig. 2.6). In CK pastures, I_a for species richness was larger than 1 (Table 2.4), indicating spatial aggregation of richness. This spatial pattern was not found in one LG pasture and one HG pasture (Table 2.4).

Grazing significantly affected MSH for SWC: MSH for MG (P = 0.034) and HG (P = 0.098, Table 2.2) were both higher than MSH for CK. The MSH for SOC declined with increasing grazing intensity from 94.9 (mean) in CK to 58.1% in HG (Fig. 2.5b), while the range for SOC did not show directional response to grazing (Table 2.2). For soil NH_4^+ and NO_3^-, spatial pattern were only found in low or intermediate grazing intensities (Table 2.2).

At the coarse scale. AGB presented spatial pattern at coarse scale in some of pastures, while the range did not change along grazing gradient (linear regression: P = 0.700), neither did vegetation height (Table 2.3). In HG1, I_a for species richness was 2.493 (P < 0.001), while I_a in other plots were not significantly different from 1 (Table 2.4). For plots with random or linear model as best fit for SWC data at the fine scale, exponential models became the best fit at the coarse scale (Table 2.3). Changing from the fine scale to coarse scale for these plots, range (Fig. 2.5a) and MSH declined (Fig. 2.5b) with increasing stocking rates (Fig. 2.5a). There was no directional response of spatial patterns of NH_4^+, NO_3^-, TN or C/N ratio to increasing grazing intensities (Table 2.3).

2.3.4 Correlations and Cross-correlograms

In each sampling plot, AGB had a positive relationship with vegetation height (Table 2.5). There was a positive relationship between AGB and soil NO_3^- in CK and LG; however, this relationship was non-significant in one of the MG plots and both HG plots (Table 2.5). The cross-correlograms revealed that there was significant cross-correlation between AGB and soil NO_3^- at lag intervals of 0.1-0.14 and 0-0.14 m in CK1 and CK2, respectively (Fig. 2.7). At a similar interval, the cross-correlation was not significant in other plots (Fig. 2.7).

2.4 Discussion

The decrease in AGB and vegetation height by HG and the similarity in AGB and vegetation height between CK and LG (Fig. 2.2) are consistent with a previous study that was conducted in a similar desert steppe in China (Wei et al. 2000). This reduction of AGB by intensive grazing led to the dramatic decline of vegetation patch size, suggesting increased vegetation fragmentation with the increasing grazing intensity (Table 2.2 and Fig. 2.6). The breakup from relatively larger vegetation patches to smaller ones along the grazing gradient further indicated that the spatial distribution of vegetation became more homogeneous with increasing grazing intensity. In arid and semiarid environments, vegetation patches (especially the large ones) provide favorable habitats for maintaining species richness and improving seedling establishment (Callaway 1997, Maestre et al. 2001, Maestre et al. 2003). With the loss of large vegetation patches under grazing, fragmented vegetation could lead to loss of spatial aggregation of species richness (Table 2.3). This change of spatial vegetation pattern could also negatively affect pollination and reproduction (Aguilar et al. 2006), and increase the risk of plant species loss (Joshi et al. 2006). Previous studies have also found that fragmented vegetation induced loss of rare animal species and altered animal community composition (Golden and Crist 1999). Therefore, range managers should avoid the use of high intensity grazing to maintain vegetation spatial pattern.

Even though plant height was significantly correlated with AGB (Table 2.5), its spatial heterogeneity was not decreased by heavy grazing (Table 2.2), suggesting that spatial pattern of height distribution was different from that of AGB. This might have resulted from the changed plant community by grazing. Palatability of individual plant species affects the behavior of sheep so that more palatable species would likely to be consumed first. Therefore, a less palatable species with a relatively large size, such as *A. frigid*, could increase its abundance under high grazing pressure in desert steppes (Li et al. 2008a), causing the disparity of grazing effects on the spatial heterogeneity between AGB and height. Though plant height has been successfully used to study the spatial pattern of vegetation (Olofsson et al. 2008), these results suggest that plant height only represents one aspect of spatial vegetation structure and it should be used with other parameters (e.g. AGB) for interpretation of data.

Loss of soil C and N by overgrazing has been found in different types of steppes in Inner Mongolia (Su et al. 2005, Han et al. 2008, Steffens et al. 2008).

For example, Li et al. (2008a) found that heavy grazing (10-15 sheep-unit-month ha^{-1}) decreased SOC and TN by 16.4 and 11.4% in an Inner Mongolian desert steppe. These studies suggested that depletion of plant litter input and wind and water erosion were the major mechanisms for the soil total C and N loss under heavy grazing. However, neither SOC nor TN responded to grazing intensities in this study (Fig. 2.2). Steffens *et al.* (2008) found that SOC loss caused by grazing was not recovered after five years of grazer exclusion in a semi-arid steppe in China. In this study, the length of treatment time (4 years) and small sample size could limit the ability of detecting responses of soil total C and N to grazing intensities.

The altered soil C/N ratio by grazing (Fig. 2.2) could be explained by the grazing-induced changes in community composition and litter input. It was found that *S. collina*, a C4 annual weed species with higher tissue C/N ratio (Wang et al. in preparation), greatly increased its abundance and productivity in grazed plots after 2004 (Wang XL and Han GD, personal communication). The replacement of C3 species with a C4 annual was likely to contribute to higher soil C/N ratio in LG plots, because of high litter accumulation in LG and the loss of litter cover in MG and HG (Li et al. 2008; Wang XL and Han GD, personal communication). Spread of *S. collina* should be controlled to prevent further loss of soil fertility in range management practices for the studied ecosystem type.

This study found that the heterogeneity of SOC and SWC decreased with increasing grazing intensity (Fig. 2.5). Augustine and Frank (2001) proposed four mechanisms to explain how grazing might affect soil spatial heterogeneity: (1)

urine and dung input by grazers, (2) changes in root turnover/exudation and input of leaf litter, (3) changes in spatial distribution of plant species composition, and (4) interactions of grazing with organisms such as insects. Dung and urine input could increase soil nutrient availability (Tolsma et al. 1987) and create patches with high nutrient availability (Jaramillo and Detling 1992); however, this study found that heavy grazing tended to lower soil mineral N availability and variability (Fig. 2.2 and Fig. 2.3). Therefore, impacts of grazer excreta on the studied steppe might be rather limited. It is interesting that heterogeneity of AGB also decreased with increasing grazing intensity. Compared to bare ground, vegetation patches could provide the habitat for various organisms and had higher root turnover/exudation, litter input and microbial activities (Schade and Hobbie 2005, Joshi et al. 2006). Vegetation patches are also less likely to be affected by wind erosion and water runoff (Li et al. 2008b, Throop and Archer 2008). Thus, the change of spatial vegetation patterns could alter the input to the soil of substrates such as root material and litter, the soil loss through erosion, and species composition, leading to different spatial patterns of soil nutrients. Change of vegetation heterogeneity provides another important mechanism through which grazing can affect the spatial heterogeneity of soil nutrients.

The finding that increased grazing intensity decreased soil mineral N availability and its overall variability (Fig. 2.2 and Fig. 2.3) is the opposite of some previous studies (Willms et al. 1986, Tolsma et al. 1987). The indirect effect of sheep grazing on litter decomposition and mineral N availability might be more important than the direct effect of excreta addition (Hobbs 1996). The strong decline of plant AGB with heavy grazing (Fig. 2.2) could lead to less plant root biomass. With a similar trend of litter cover, the lower root biomass under heavy grazing suggested the decreased availability of primary substrate for decomposition and consequently less N release from mineralization. The loss of large vegetation patches, where decomposition rates were usually higher, under high grazing intensity could lead to the lower overall variability of soil mineral N and consequently reduced productivity. This mechanism suggests that high intensity grazing will severely damage forage production in the desert steppe ecosystems and increase the risk of desertification.

As a more active part of soil nutrient pool, spatial pattern of soil mineral N appeared to be less affected by grazing compared to those of other parameters such as SOC and TN (Rietkerk et al. 2000, Augustine and Frank 2001). Previous research reported that NH_4^+ and NO_3^- had spatial patterns with a range below or around 1 m at fine scale (Jackson and Caldwell 1993, Ryel et al. 1996, Rietkerk et al. 2000) and a range from 2 to 15 m at coarse scale (Augustine and Frank 2001, Gallardo et al. 2006); those are similar to what was found in this study. In the current study, NH_4^+ and NO_3^- only showed fine scale spatial patterns in LG or MG plots, indicating that increased grazing intensity first created heterogeneity of soil mineral N and then decreased its heterogeneity at the highest grazing intensity. Although herbivores can have idiosyncratic impacts on soil nutrient supply (Bardgett and Wardle 2003), higher NH_4^+ in LG than that in CK suggested that low grazing pressure was likely to positively influence soil NH_4^+ availability by increasing N mineralization and plant root exudation, especially under vegetation patches (Holland et al. 1996, Frank and Groffman 1998, Tracy and Frank 1998). Similar to the effects of excreta addition, these indirect grazing effects could create some area with high mineral N availability and further formed spatial patterns of mineral N. The reduced heterogeneity of mineral N by heavy grazing was consistent with the responses of SOC and SWC and such changes likely follow similar mechanisms. These results indicate that low grazing intensity will preserve soil fertility and potentially improve forage production.

In this study, there was no significant correlation between the size of AGB spatial autocorrelation and that of any other variables at either the fine or coarse scale (data not shown). Spatial cross-correlation between AGB and soil NO_3^- at a small distance was only significant in CK (Fig. 2.7), suggesting grazing could directly affect the spatial relationship between vegetation and soil nutrients. However, similar phenomenon was not found between AGB and other soil variables. Schlesinger et al. (1996) found that islands of fertility were similar in size as the individual shrub and suggested this phenomenon would be more prominent in arid shrubland than in grassland ecosystems. In this perennial grass and forb species dominated steppe, it is possible that these species were less effective in forming islands of fertility than shrubs, causing the lack of overlap between the spatial patterns of soil nutrients and AGB. However, there were still connections between vegetation and soil nutrients that allow grazing to indirectly impact the spatial structure of soil nutrients through changing biotic processes such as plant community composition, litter input and plant root exudation, as discussed above. My results revealed the importance of considering vegetation-soil feedback in the study of the spatial heterogeneity of soil nutrients even in grassland ecosystems.

2.5 Conclusions

Results from this study support the hypothesis that different grazing intensities had divergent effects on spatial patterns of vegetation and soil nutrients in the studied desert steppe. In general, heavy grazing increased the homogeneity of the spatial distribution of AGB, SOC and SWC, demonstrating the strong relationship between spatial heterogeneity of vegetation and soil nutrients. Heavy grazing also led to loss of plant species aggregation and decrease of soil NH_4^+ availability. In contrast, light grazing intensity created the spatial heterogeneity of soil mineral N and maintained aboveground biomass production and soil nutrient availability. These results suggest that light grazing is essential in range management to preserve forage production, soil fertility and biodiversity. Loss of cross-correlation between AGB and soil NO_3^- in grazed plots indicates that grazing can directly alter the relationship between spatial patterns of vegetation and soil nutrient. This study demonstrates that sheep grazing plays a strong role in generating, maintaining and changing the spatial patterns of vegetation and soil nutrients in the studied desert steppe ecosystem.

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Table 2. 1 Component loading for PCA diagram.

	Componet	
	1	2
AGB	0.589	-0.332
Vegetation height	0.75	-0.413
Plant species richness	0.384	-0.121
SWC	-0.273	0.054
NH_4^+	0.198	0.084
NO_3^-	0.378	0.085
SOC	0.653	0.483
TN	0.599	0.337

Table 2.2 Summary of semivariogram model parameters for aboveground biomass (AGB), vegetation height, soil water content (SWC), soil NH_4^+, NO_3^-, soil organic C (SOC), total N (TN) and C/N ratio at the fine scale.

	Block	Model	A (m)	MSH	Block	Model	A (m)	MSH
AGB (g m^{-2})	CK1	S	0.88	0.631	CK2	S	1.15	0.786
	LG1	S	0.55	0.750	LG2	S	0.33	0.877
	MG1	E	0.83	0.502	MG2	S	0.14	0.936
	HG1	E	0.18	0.914	HG2	R	<0.01	<0.001
Vegetation height (cm)	CK1	R	<0.01	<0.001	CK2	S	0.73	0.739
	LG1	E	0.01	0.938	LG2	S	0.10	0.952
	MG1	S[a]	0.69	0.500	MG2	S	0.11	0.998
	HG1	R	<0.01	<0.001	HG2	E	0.78	0.500
SWC (vol, %)	CK1	L	>2.85	0.116	CK2	R	<0.01	<0.001
	LG1	L	>2.85	0.168	LG2	S	0.48	0.553
	MG1	S	0.83	0.676	MG2	S	1.16	0.808
	HG1	E	0.32	0.545	HG2	S	0.55	0.547
NH_4^+ (mg kg^{-1})	CK1	R	<0.01	<0.001	CK2	R	<0.01	<0.001
	LG1	S	0.79	0.763	LG2	E	0.16	0.875
	MG1	R	<0.01	<0.001	MG2	E	0.21	0.797
	HG1	R	<0.01	<0.001	HG2	R	<0.01	<0.001
NO_3^- (mg kg^{-1})	CK1	R[a]	<0.01	<0.001	CK2	R	<0.01	<0.001
	LG1	R	<0.01	<0.001	LG2	S[a]	0.12	0.992
	MG1	R[a]	<0.01	<0.001	MG2	R	<0.01	<0.001
	HG1	R	<0.01	<0.001	HG2	R	<0.01	<0.001
SOC (g kg^{-1})	CK1	S	0.15	0.998	CK2	G	0.82	0.900
	LG1	S	0.23	0.892	LG2	S	0.98	0.587
	MG1	S	1.07	0.615	MG2	S	1.09	0.629
	HG1	S	0.56	0.500	HG2	S	0.57	0.661
TN (g kg^{-1})	CK1	E	0.27	0.730	CK2	S	1.40	0.500
	LG1	S	0.81	0.520	LG2	L[a]	>2.85	0.302
	MG1	S	0.17	0.690	MG2	E	0.13	0.869
	HG1	S	0.72	0.860	HG2	E	0.33	0.780
C/N ratio	CK1	R	<0.01	<0.001	CK2	E	1.14	0.501
	LG1	S	1.53	0.502	LG2	E	0.19	0.938
	MG1	R	<0.01	<0.001	MG2	S	1.08	0.693
	HG1	S	0.62	0.698	HG2	R	<0.01	<0.001

A, range; MSH, magnitude of spatial heterogeneity; R, random; L, linear; S, spherical; E, exponential; G, Gaussian. [a] indicates that the data was detrended.

Table 2.3 Summary of semivariogram model parameters for AGB, vegetation height, SWC, soil NH_4^+, NO_3^-, SOC, TN and C/N ratio at the coarse scale.

	Block	Model	A (m)	MSH	Block	Model	A (m)	MSH
AGB (g m^{-2})	CK1	S	3.4	0.820	CK2	E	0.9	0.839
	LG1	R	<0.01	<0.001	LG2	S	4.6	0.800
	MG1	E	8.3	0.850	MG2	G	4.4	0.990
	HG1	G	4.2	0.790	HG2	R	<0.01	<0.001
Vegetation height (cm)	CK1	S	12.7	0.902	CK2	R	<0.01	<0.001
	LG1	S	15.8	0.500	LG2	G	2.0	0.883
	MG1	L	> 18.7	0.082	MG2	G	2.6	0.998
	HG1	E	8.0	0.906	HG2	R	<0.01	<0.001
SWC (vol, %)	CK1	E	2.1	0.889	CK2	E	2.0	0.873
	LG1	E	3.1	0.840	LG2	E	1.3	0.903
	MG1	S^a	5.0	0.552	MG2	R^a	<0.01	<0.001
	HG1	R	<0.01	<0.001	HG2	R	<0.01	<0.001
NH_4^+ (mg kg^{-1})	CK1	S	15.6	0.953	CK2	R	<0.01	<0.001
	LG1	G	3.2	0.842	LG2	G	2.4	0.999
	MG1	E	9.1	0.921	MG2	G	3.1	0.934
	HG1	S	3.9	0.874	HG2	S	1.8	0.999
NO_3^- (mg kg^{-1})	CK1	G	2.2	0.876	CK2	R	<0.01	<0.001
	LG1	R	<0.01	<0.001	LG2	R^a	<0.01	<0.001
	MG1	E	11.5	0.819	MG2	S	6.7	0.695
	HG1	G	2.5	0.952	HG2	E	3.1	0.802
SOC (g kg^{-1})	CK1	E	3.1	0.600	CK2	S	1.2	0.994
	LG1	G	36.2	0.702	LG2	R	<0.01	<0.001
	MG1	E^a	2.2	0.788	MG2	R	<0.01	<0.001
	HG1	G	10.0	0.821	HG2	L	>18.7	0.265
TN (g kg^{-1})	CK1	E	5.9	0.844	CK2	R	<0.01	<0.001
	LG1	E	3.8	0.831	LG2	E	1.8	0.879
	MG1	S^a	4.8	0.587	MG2	R^a	<0.01	<0.001
	HG1	R	<0.01	<0.001	HG2	E	2.9	0.802
C/N ratio	CK1	E	1.6	0.847	CK2	S	10.0	0.546
	LG1	G	29.1	0.629	LG2	R	<0.01	<0.001
	MG1	R	<0.01	<0.001	MG2	S	1.2	0.997
	HG1	S	14.6	0.553	HG2	E	3.0	0.857

Refer to Table 2.2 for explanation of abbreviations.

Table 2.4 Index of aggregation, I_a, and probability of having $I_a \leq 1$ for plant species richness at the fine and coarse scales.

	Block 1	I_a	P	Block 2	I_a	P
Fine scale	CK1	1.532	0.030	CK2	1.481	0.034
	LG1	0.889	0.633	LG2	1.639	0.015
	MG1	1.717	0.011	MG2	1.592	0.017
	HG1	1.278	0.128	HG2	1.310	0.084
Coarse scale	CK1	0.718	0.923	CK2	0.843	0.645
	LG1	0.898	0.553	LG2	0.588	0.999
	MG1	1.529	0.058	MG2	1.341	0.118
	HG1	2.493	<0.001	HG2	0.890	0.547

Table 2.5 Pearson correlation coefficients between AGB (g m^{-2}) and vegetation height (cm), SWC (vol, %), soil NH_4^+ (mg kg^{-1}), NO_3^- (mg kg^{-1}), SOC (g kg^{-1}), TN (g kg^{-1}) and C/N ratio.

	Height	SWC	NH_4^+	NO_3^-	SOC	TN	C/N
CK1	0.290	/	0.205	0.310	0.205	/	0.211
CK2	0.539	/	/	0.173	/	/	/
LG1	0.299	/	/	0.212	/	/	/
LG2	0.302	/	/	0.277	/	0.234	/
MG1	0.425	0.170	/	/	/	/	/
MG2	0.359	/	/	0.163	/	/	/
HG1	0.455	/	/	/	-0.203	/	/
HG2	0.617	/	/	/	/	/	/

The correlation analyses were performed independently for each plot and only significant correlations are shown. Refer to Table 2.2 for explanation of abbreviations.

Figure legends

Fig. 2.1 (a) A schematic diagram of the experiment design. CK: control; LG: lightly grazed; MG: moderately grazed; HG: heavily grazed. Numbers indicate the length (m) of each edge. (b) A schematic diagram of the 30×10 m and 4×4 m sampling matrix. Each square represents a sampling point.

Fig. 2.2. Grazing effects on (a) AGB, (b) vegetation height, (c) soil C/N ratio, (d) SWC, (e) NH_4^+ and (f) NO_3^-. Error bars denote S.E. Bars with different letters are significantly different at $P \leq 0.05$.

Fig. 2.3. Grazing effects on the coefficient of variances (CV) for (a) AGB, (b) vegetation height, (c) soil C/N ratio, (d) SWC, (e) NH_4^+ and (f) NO_3^-. Error bars denote S.E. Bars with different letters are significantly different at $P \leq 0.05$.

Fig. 2.4. PCA diagram showing components 1 and 2, separating samples from different grazing treatments.

Fig. 2.5 (a) Relationships between the range for AGB (filled circle, solid line, P = 0.020, R^2 = 0.562) and SWC (empty circle, dash line, P = 0.063, R^2 = 0.362) and stocking rates at the fine scale. (b) Relationships between the MSH for SWC (filled circle, solid line, P = 0.033, R^2 = 0.487) and SOC (empty circle, dash line, P = 0.015, R^2 = 0.598) and stocking rates at the fine scale. For plots in which random or linear model were the best-fit for SWC semivariances, the range at the coarse scale was used instead.

Fig. 2.6 Semivarigram for AGB in each pasture. CK: control; LG: lightly grazed; MG: moderately grazed; HG: heavily grazed.

Fig. 2.7 Cross-correlogram between AGB and soil NO_3^- at the fine scale under different grazing treatments. Dotted lines indicate critical values of the Pearson correlation coefficient r for $\alpha = 0.05$ with the number of pairs at each lag class.

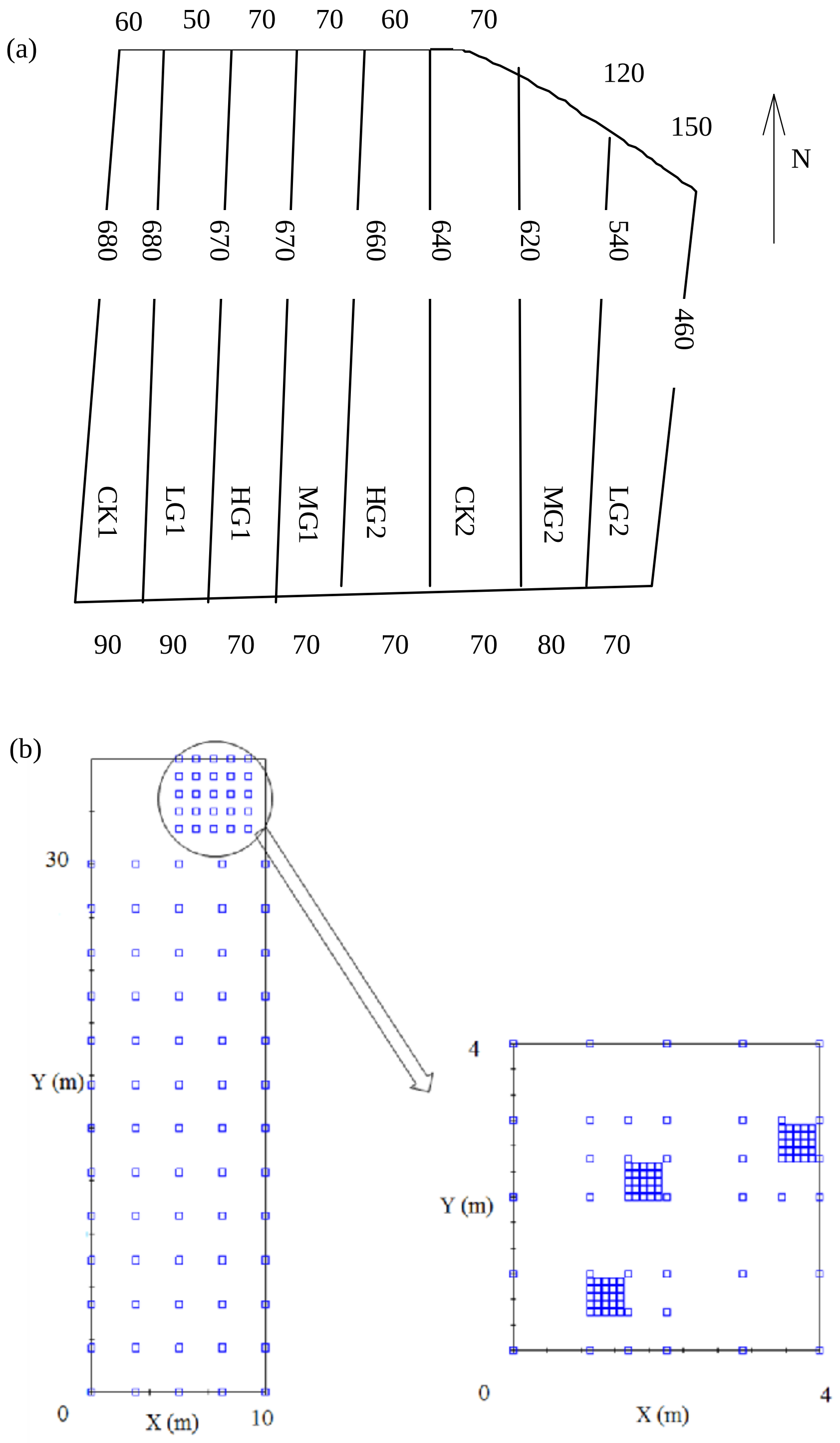

Fig. 2.1 A schematic diagram of the experiment design. Numbers indicate the length (m) of each edge. (b) A schematic diagram of the 30×10 m and 4×4 m sampling matrix.

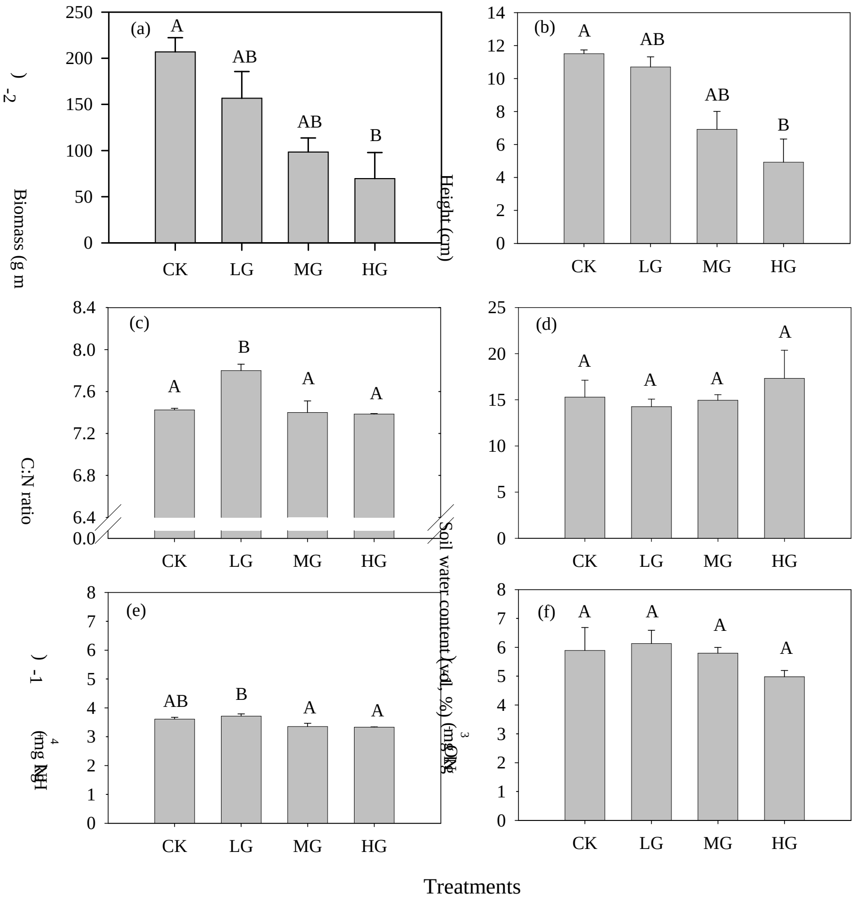

Fig. 2.2 Grazing effects on (a) AGB, (b) vegetation height, (c) soil C/N ratio, (d) SWC, (e) NH_4^+ and (f) NO_3^-.

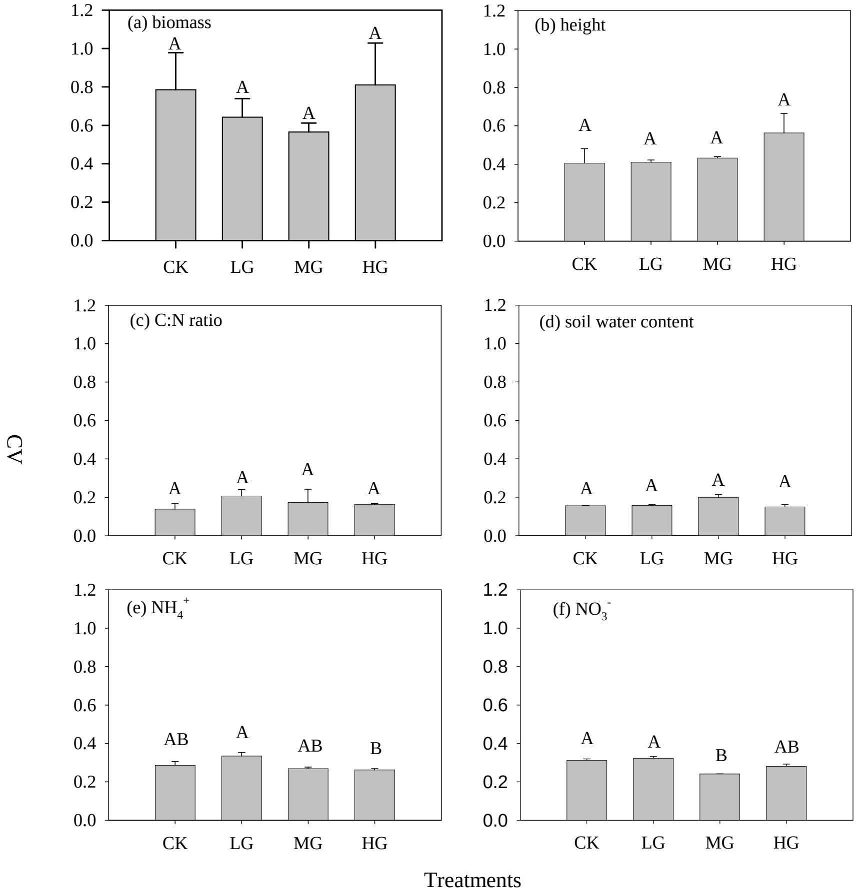

Fig. 2.3 Grazing effects on the coefficient of variances (CV) for (a) AGB, (b) vegetation height, (c) soil C/N ratio, (d) SWC, (e) NH_4^+ and (f) NO_3^-.

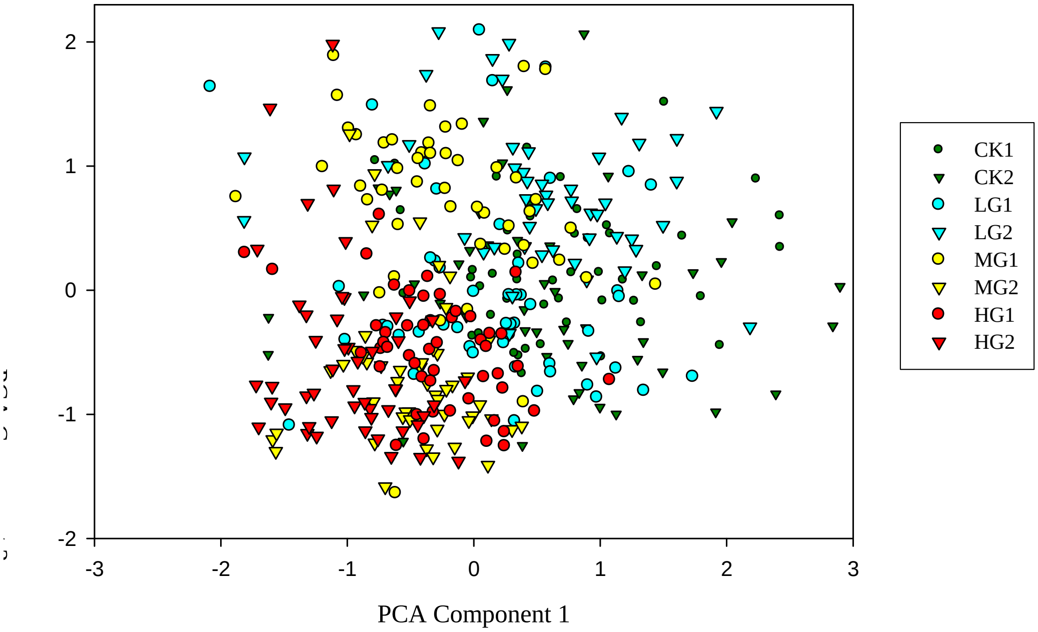

Fig. 2.4 PCA diagram showing components 1 and 2, separating samples from different grazing treatments.

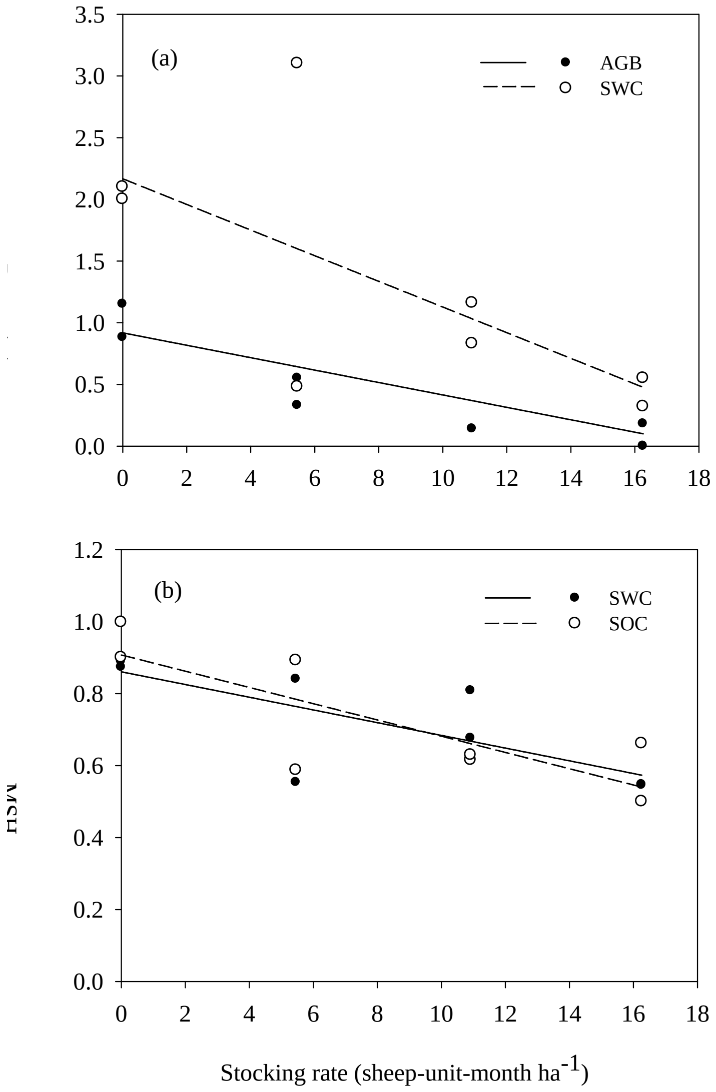

Fig. 2.5 (a) Relationships between the range for AGB and SWC and stocking rates at the fine scale. (b) Relationships between the MSH for SWC and SOC and stocking rates at the fine scale.

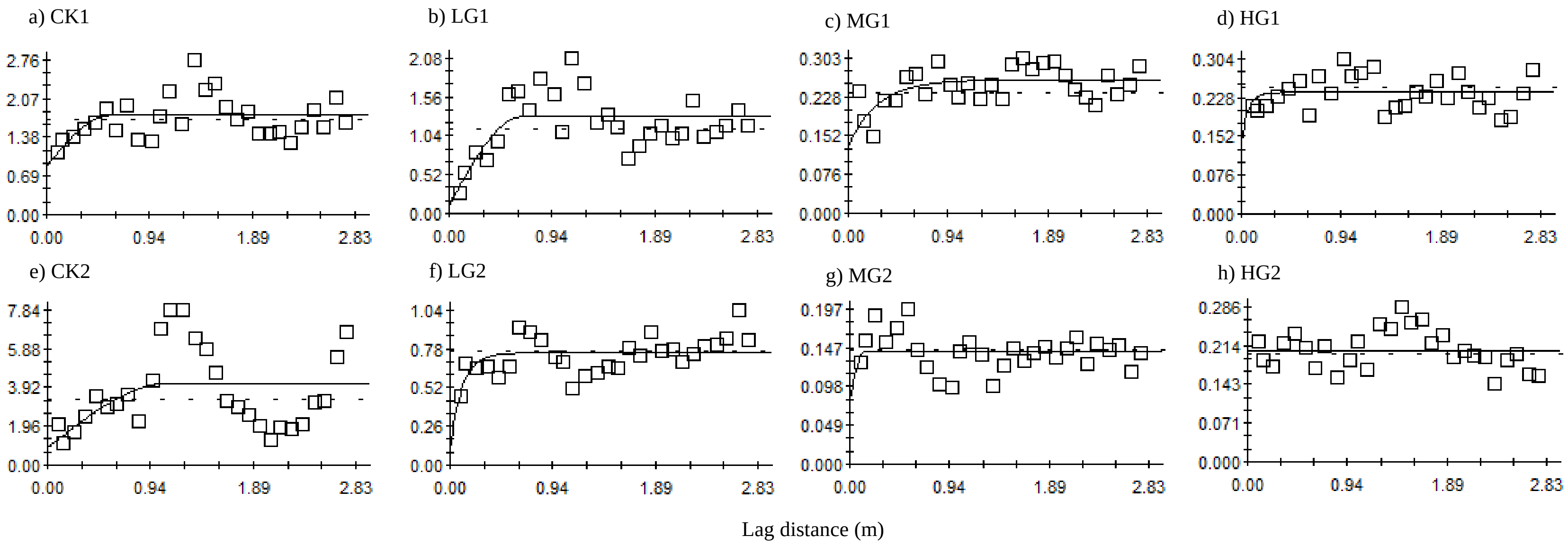

Fig. 2.6 Semivarigram for AGB in each pasture.

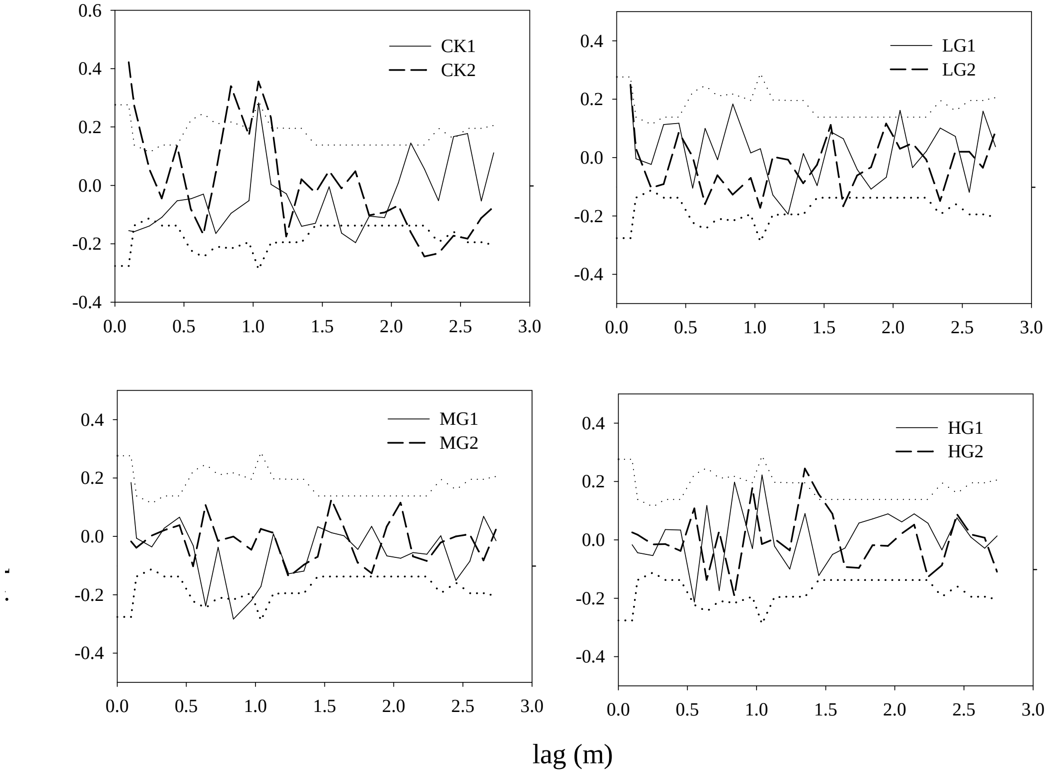

Fig. 2.7 Cross-correlogram between AGB and soil NO_3^- at the fine scale under different grazing treatments.

Spatial vegetation patterns as early signs of desertification: a case study of a desert steppe in Inner Mongolia, China

3.1 Introduction

Grazed arid and semi-arid ecosystems occupy around 30% of the earth's terrestrial surface (HilleRisLambers et al. 2001). In these ecosystems, vegetation can be considered as a two-phased mosaic consisting of a high-cover phase and a low-cover phase (bare land) (Aguiar and Sala 1999). The spatial pattern of vegetation varied greatly among ecosystems, ranging from irregular mosaics to regular patterns such as "stripes", "labyrinths" and "spots" (HilleRisLambers et al. 2001, Rietkerk et al. 2002, Rietkerk et al. 2004). Even within an ecosystem, spatial distribution of vegetation can show many different patterns (Bautista et al. 2007).

In arid and semi-arid ecosystems, the effects of spatial vegetation pattern on ecological processes have been emphasized in the recent literature (Golden and Crist 1999, Ludwig et al. 2007, Collins et al. 2009). Vegetation patch has strong effects on the soil beneath them with respect to water infiltration, plant debris input, and sediment deposition (Puigdefábregas 2005), and can thus form islands of fertility where nutrient availabilities are higher than those in surrounding bare soil (Schlesinger et al. 1996, Reynolds et al. 1999). Therefore, vegetation patchiness causes the heterogeneous distribution of soil nutrients (Rietkerk et al. 2000, Schade and Hobbie 2005), affects hydrological processes such as runoff and sediment yield, and influences the rate and direction of soil development (degradation or aggradation) in arid and semi-arid ecosystems (Reid et al. 1999, Ludwig et al. 2000, Descheemaeker et al. 2006). Spatial vegetation pattern also has a strong relationship with the diversity of various organisms, including plants, insects and birds (Golden and Crist 1999, Crist et al. 2006, Joshi et al. 2006, Coppedge et al. 2008).

Recent research has attempted to link spatial vegetation patterns to early desertification due to their important ecological functions. The pioneering study of Kefi et al. (2007) showed that patch-size distribution would deviate from the standard power law under high grazing pressure and they suggested that this deviation could serve as an indicator of higher risk of desertification based on both field and model simulation data. However, Maestre and Escudero (2009) did not observe such deviation along an increasing desertification gradient, which was surrogated by a set of soil variables; instead they found that percent plant cover could indicate higher risk of desertification. The above two studies, both conducted in Mediterranean ecosystems, were the only ones that used vegetation spatial pattern to predict desertification processes. In addition, they both applied the line-intercept method to estimate individual vegetation patch size; such a one-dimensional method might lead to biased patch size distribution data. Therefore, the relationship between the proposed desertification indictors and degradation processes should be tested in other types of ecosystems, and a more robust method for measuring patch size also needs to be developed.

Inner Mongolia Steppe is the main part of the central Eurasian Steppe region and the largest continuous grassland in China. Overgrazing has caused severe land degradation and eventually desertification in this region (Su et al. 2005, Zhao et al. 2005, Zheng et al. 2005). Compared to other types of grasslands, desert steppe accounts for 39% of total native grassland in Inner Mongolia and has a higher susceptibility to overgrazing (Li et al. 2000). Therefore, well-controlled experiments with a gradient of grazing intensity can represent desert steppe ecosystems with different desertification potentials, given the lack of a common method for assessing desertification (Verón et al. 2006). I examined the change of spatial vegetation pattern along a grazing gradient in a desert steppe in Inner Mongolia, China. Vegetation distribution was studied by a series of land photographs to produce maps that show vegetation patches and bare ground. Spatial metrics (such as patch density, mean patch area and clumpiness index) and patch-area distribution were then determined based on these maps. In this study, I specifically asked two questions: First, how will grazing intensities modify the patchiness of vegetation? Second, will percent plant cover or the change of patch-area distribution (from a power law to a truncated power law) indicate higher risk of desertification associated with increasing stocking rate?

3.2 Materials and Methods

3.2.1 Study site and experiment design

The study site (41°47’17”N, 111°53’46”E) is located in Siziwang Banner, Inner Mongolia Autonomous Region (IMAR), China. This site is around 140 km north of Huhhot, the capital city of IMAR and the research ranch at the site is managed by Inner Mongolia Agricultural University. The study area has a mean annual temperature of 3.4 °C with the highest monthly mean temperature in July (24.0 °C). Long-term mean annual precipitation is 280 mm. The climate is arid, with windy and dry springs (with 49 mm average rainfall), and warm and comparatively rain-rich summers (with 220 mm average rainfall). The vegetation is of typical desert short-grass steppe, dominated by *Stipa breviflora* Griseb., *Artemisia frigid* Willd. and *Cleistogenes songorica* Roshev.. Other species include *Convolvulus ammannii* Desr., *Artemisia pectinata* Pall., *Kochina prostrata* (L.) Schrad, *Caragana stenophylla* Pojark., *Leymus chinensis* (Trin.) Tzvel. and *Salsola collina* Pall.. Vegetation cover averages 20%. The soil is a Brown Chernozemic (the Canadian System of Soil Classification) or a Kastanozemic soil (the FAO System of Soil Classification).

In July 2002, ~ 35 ha steppe, which used to be grazed (stocking rate: 6-10 sheep-unit-month ha^{-1}) year round under collective sheep grazing for over 30 years, was fenced for this experiment. In 2004, this site was divided into two blocks and an experiment with a randomized complete block design was established so that each block contained four pastures (Fig. 2.1). In total, there were 8 pastures, each with an average area of 4.4 ha. Four treatments (CK: control; LG: lightly grazed; MG: moderately grazed; HG: heavily grazed) were randomly applied within each block. The stocking rates were 0 (CK), 5.46 (LG), 10.92 (MG), 16.26 (HG) sheep-unit-month ha^{-1}. Grazing period was around six months per year, from early April to early October. The grazing gradient covered from grazer exclusion to heavy grazing and represented the most common grazing pressures in this region (Wei et al. 2000, Jiao et al. 2006). One 30×10 m grid was set up in each pasture to study the spatial distribution of selected vegetation and soil nutrient variables, which has been discussed in Chapter 2. One 10×10 m area in each 30×10 m grid was randomly chosen to examine the spatial distribution of vegetation patches.

3.2.2 Vegetation photography and image processing

The 10×10 m area was divided into 8×8 matrix by 1.25 m interval. To each quadrat, 1 mm-GSD (ground sampled distance) image was taken using a Canon PowerShot A570 IS camera at peak growing season (mid-August). Four corners of each quadrat were marked in the field as control points for geo-correction of images.

A linear rubber sheeting method (Saalfeld 1985, White and Griffin 1985) was used to geo-correct the images based on the four control points at each quadrat (ERDAS IMAGINE 9.1 software). The geo-correction also allocated coordinates for corners of images so that all images from the same experimental plot could be merged by the Mosaic tool in the IMAGINE software. The merged images were then re-measured to a 2 cm resolution by nearest neighbor interpolation in the Mosaic tool. The smallest vegetation patch size (e.g. a very small bunch of *S. breviflora*) approximated 2×2 cm at this site. In each image, pixels were classified into 15 classes by using an iterative self-organizing clustering algorithm in the Isodata tool in the IMAGINE software (ERDAS 2006). The 15 classes generated for each image were then grouped into two classes, vegetation and bare ground, based on the visual interpretation of the image. An 8-

pixel neighborhood method was used to construct the map of individual patches. The above classification process produced many artifacts that usually were small patches (Bar Massada et al. 2008). To smooth the image, I used focal analysis that used a moving window (consisting of 3×3 pixels) to replace the value of the centre cell by the mean of all 9 pixels in the window (ERDAS 2006).

3.2.3 Data analyses

A series of spatial metrics were calculated by the FRAGSTATS v3.3 software (McGarigal et al. 2002) to study the spatial vegetation pattern in each experiment plot: percent cover (COVER), patch density (PD), mean patch area (AREA_MN), coefficient of variation (CV) of patch area (AREA_CV) and clumpiness index (CLUMP) (explanation see Table 3.1). Linear regression was used to examine the relationship between metrics and stocking rates. The significance level of α = 0.05 were used for all analyses that were carried out in SPSS version 11.5.

Area (A) for each vegetation patch was measured using FRAGSTATS. For each experimental plot, I examined the relationship between the number of patches, $N(A)$, and their area. Recent studies used a binning method to fit vegetation patch data into power-law type models (Kefi et al. 2007, Maestre and Escudero 2009). Even though binning method may produce biased estimation for exponents in power-law type models (White et al. 2008), it allowed direct visual interpretation of the patch-area distribution. To test the ideas proposed in both Kefi et al. (2007) and Maestre and Escudero (2009), I chose to follow their method. Two different models were used to fit the patch-area distribution:

a power law model, $N(A) = CA^{-\gamma}$

a truncated power law model, $N(A) = CA^{-\gamma} e^{-(S/S_x)}$

where γ was the estimated scaling exponent of the model, S_x (called truncation value) was the patch area above which N decreases faster than in a power law, and C was a constant (Jordano et al. 2003, Kefi et al. 2007). A sum of squares reduction test examined whether the scaling exponent was different between truncated power law and power law and was used to compare the fit of the two models (Schabenberger and Gotway 2005).

3.3 Results

3.3.1 Spatial metrics

Vegetation maps from different pastures demonstrated that vegetation became more fragmented with increasing stocking rates (Fig. 3.1). There was a negative relationship between COVER and stocking rate (Fig 3.2a). COVER was strongly correlated with all the other variables (Table 3.2). Patch density (PD) also decreased with increasing stocking rate (Fig. 3.2b). Although there was no significant relationship between stocking rate and AREA_MN (Fig. 3.2c, P = 0.107), AREA_CV declined with higher grazing pressure (Fig. 3.2d). There was also a significant linear relationship between CLUMP and stocking rate (Fig. 3.2e), suggesting that heavy grazing decreased patch aggregation.

3.3.2 Patch-area distribution

Power laws were the best fit for patch-area distribution of vegetation in the two control experimental plots, while the truncated power law was the best fit for patch-area distribution in one of the LG plots and in all MG and HG plots (Fig. 3.3). There was a significant linear relationship between γ and stocking rate (Fig. 3.4). In addition, γ had a significant exponential relationship with plant cover (Fig. 3.4). Similar to plant cover, γ was correlated with most spatial metrics, expect for AREA_MN (Table 3.2). Patch-area distribution also supported that the size of largest patch decreased and the patch counts increased with increasing grazing intensity.

3.4 Discussion

My findings that vegetation patches became more fragmented and homogeneous under higher grazing intensity (Fig. 3.1 and 3.2) are consistent with the assumption that grazing intensities represent different desertification potentials in this desert steppe. In this study, high intensity grazing markedly removed aboveground biomass and altered those relatively large patches into smaller ones. Joshi et al. (2006) found that fragmented vegetation with small patch size (0.25 m^2) had decreased colonization and increased extinction rates of plant species. A recent meta-analysis concluded that fragmentation of habitats can have large negative effects on pollination and reproduction at a broad range of scales (Aguilar et al. 2006). Most likely the fragmented vegetation under overgrazing will affect plant species colonization and extinction. In fact, aggregation of plant species richness at the 0.1-2 m scale was disrupted by grazing (Chapter 2), which may be explained (at least partially) by decreased plant species colonization. Therefore, to prevent potential diversity loss by fragmentation of vegetation, continuous high intensity grazing should be avoided.

In arid and semi-arid ecosystems, vegetation patches are usually considered as sinks for water, sediments and nutrients that would be lost under bare ground conditions, which therefore would act as the source of water, sediments and nutrients (Schlesinger et al. 1990, Cerdà 1997, Dunkerley and Brown 1999). In this study, the results of COVER (Fig. 3.2a) suggest that this desert steppe could be less efficient in trapping water and sediments in runoff under higher grazing pressure (Wu et al. 2000). Even though soil erosion was not measured, the decreased availability of soil NH_4^+ under heavy grazing (Chapter 2) indicates that overgrazing could increase the susceptibility of soil to wind and water erosion in the studied desert steppe.

Under heavy grazing, vegetation patches became more homogeneous in terms of patch extent (measured by AREA_CV). Heavy grazing might also change the diversity and heterogeneity of physical and biological conditions of patches. For example, *S. collina* was a key species to form some large vegetation patches (> 1 m^2) in CK plots that constituted a unique type of habitat. However, this type of patch was not found in HG plots, and most patches in HG were similar in size, and likely also in habitat condition. Due to the strong connection between vegetation and soil, heavy grazing also reduced the spatial heterogeneity of soil nutrients (Olofsson et al. 2008, Chapter 2). The decreased heterogeneity of vegetation patch and nutrients could lead to loss of habitat diversity and potentially affect diversity of various organisms (Wiens 1997, Adler et al. 2001, Coppedge et al. 2008).

The significant linear relationship between most metrics and stocking rate (Table 3.2) suggests that low intensity grazing impacted vegetation patches in a way different from the treatments with higher grazing intensities. Low intensity grazing preserved the relatively large patches and patch heterogeneity. Although this study only examined the vegetation pattern after four years of applying the grazing intensity treatment, the results indicate that low stocking rate is highly preferable to maintaining the spatial structure of vegetation in rangeland management at the studied desert steppe.

This study showed that power-law type models can be used to describe the vegetation patch-area distribution in Inner Mongolian desert steppe. Recent studies have found that vegetation in some arid and semi-arid ecosystems can be fitted into power-law models and the positive short-distance feedback (such as facilitation) can be responsible for this power-law distribution of vegetation (Kefi et al. 2007, Scanlon et al. 2007, Rietkerk and van de Koppel 2008). Kefi et al. (2007) suggested that decreasing local positive feedback could cause the deviation of power-law to truncated power-law distribution, so did the increasing grazing pressure. It is outside the scope of this study to examine the generation of vegetation patterns; however, in Chapter 2, I found a positive correlation between plant aboveground biomass and soil NO_3^- concentrations in CK plots at the fine scale but not at coarse scales, suggesting that scale-dependent feedback could be one of the reasons for the formation of power-law distribution in the desert steppe.

To my knowledge, this is the first study to confirm the hypothesis of Kefi et al. (2007) that power-law distribution could deviate to truncated power-law distribution with increasing grazing pressure. Considering improper grazing is the main cause of desertification in this desert steppe, the deviation of patch-area distribution could indicate higher risk of desertification. Patch-area distribution revealed similar responses to increasing grazing intensity as spatial metrics, including the disappearance of large patches and increased patch numbers (density) (Fig. 3.3), both of which resulted in the deviation from power law. Although this study is among a limited but growing number of empirical experiments that examined the patch-area (size) relationship in grazed grasslands, previous research has found similar negative effects of grazing on spatial vegetation pattern (Ares et al. 2003, Bar Massada et al. 2008, de Knegt et al. 2008). Therefore, deviation from the power law is likely to be found in other arid and semi-arid ecosystems. It is interesting that γ strongly correlated with the spatial metrics that indicate patch heterogeneity (such as AREA_CV) and aggregation (CLUMP). If γ can be linked to similar metrics in other arid and semi-arid ecosystems, the use of power-law models will have a broader application to assess other characteristics of vegetation patches including heterogeneity and aggregation.

The two LG experimental plots had different best fit models for the patch-area relationship (Fig. 3.3) and spatial metrics in LG usually had high variation (Fig. 3.1), suggesting that these two LG pastures had quite different vegetation patterns. If the deviation from power-law relationship represented a change of degree of degradation in desert steppe, then this change might happen at a stocking rate close to 0.9 sheep ha^{-1}. In addition to grazing, differences in historical management practices and landscape position between blocks might be other factors in affecting vegetation pattern. These factors should be considered in deciding the proper stocking rate. Combined with low stocking rate, rotational grazing offers a possibility to better preserve sustainable vegetation pattern (Teague and Dowhower 2003).

This study links percent plant cover to the characteristics of the spatial distribution of vegetation, such as γ and other spatial metrics (Fig. 3.4 & Table 3.3), in desert steppe ecosystems, suggesting that vegetation cover can indicate higher possibility of grazing-induced desertification, at least in this ecosystem. Since cover is easier to measure than patch-area distribution and spatial metrics, it could potentially serve as a predictor for spatial vegetation pattern (Imeson and Prinsen 2004). However, the reliability of using cover to monitor vegetation pattern and degradation should be evaluated for more ecosystem types. For instance, in a semiarid Mediterranean landscape where vegetation cover was relatively constant throughout the whole site, there was a great variation of vegetation spatial pattern within this site (Bautista et al. 2007).

In terms of the validity of using power-law models to indicate desertification, Kefi et al. (2007) and Maestre and Escudero (2009) provided contradictory results. My results suggested that both percent plant cover, proposed by Maestre and Escudero (2009), and the deviation from power-law relationship, proposed by Kefi et al. (2007) can indicate the changes of spatial vegetation pattern and potentially serve as an indicator for desertification and degradation processes in the Inner Mongolia desert steppe. I noticed that the above two studies used different numbers of transects during data collection so that they had quite different numbers of patches for data analysis. For each bin, the largest number of patches was around 10^1 in Maestre and Escudero (2009), while this parameter reached or surpassed 10^2 in Kefi et al. (2007). In fact, sub-image (2×2 m) of two CK experimental plots in this study could show a patch-area distribution that was best fitted by a truncated power law (data not shown). Therefore, differences in the size of datasets could affect the fitting of data to a power law vs truncated power law. In this study, the vegetation patches were usually much smaller than 1 m^2 so that the two dimensional method could collect a relatively large number of patches in a small area. Moreover, two dimensional methods could give better estimations for patch area and they should be tested in future studies for evaluating the patch-area relationship in arid and semi-arid ecosystems.

3.5 Conclusions

This study indicates that heavy grazing strongly fragmented vegetation patches, decreased patch heterogeneity, and caused the patch-area distribution to deviate from the power-law model in a desert steppe in Inner Mongolia. Due to its severe negative impacts on vegetation growth and spatial distribution, heavy grazing should be avoided to sustain biomass production. Light intensity grazing maintained a vegetation spatial pattern similar to the treatment with complete exclusion of livestock, so this is the preferable management regime for the studied desert steppe. Due to its strong relationship with patch dynamics, percent plant cover can be used to effectively evaluate the impact of grazing on vegetation pattern in a desert steppe. This study also supported that deviation from power-law distribution could reflect the degree of disturbances, such as disturbances caused by different degrees of grazing intensity, and potentially indicate the risk of desertification, which should be examined by future studies especially in other types of ecosystems.

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Table 3.1 A list of spatial metrics used in this paper. Definitions of those metrics follow McGarigal et al. (2002)

Metric name	Description	Range
Percent cover (COVER)	Percent of total land surface area with vegetation (%)	0 ≤ COVER ≤ 100
Patch density (PD)	Number of patches per unit area	PD ≥ 0
Mean patch area (AREA_MN)	Mean area (cm^2) of patches on the landscape	AREA_MN ≥ 0
Coefficient of variation (CV) of patch area (AREA_CV)	Standard deviation (SD) divided by the mean, multiplied by 100, for patch area	AREA_CV ≥ 0
Clumpiness index (CLUMPY)	A measure of patch aggregation	-1 ≤ CLUMPY ≤ 1, approaches -1 when the focal patch type is maximally disaggregated; equal to 0 when the patch distribution is random; equals to 1 when the patch is maximally aggregated

Table 3. 2 Pearson correlation between plant cover and scaling exponent, γ, of the best-fit model of the patch-area distribution and other spatial metrics of vegetation. ($n = 8$)

	Cover		γ	
	Correlation coefficient	P value	Correlation coefficient	P value
Patch density (PD)	-0.971	<0.001	-0.868	0.005
Mean patch area (AREA_MN)	0.912	0.002	0.553	0.156
Coefficient of variation (CV) of patch area (AREA_CV)	0.909	0.002	0.821	0.012
Clumpiness index (CLUMPY)	0.939	<0.001	0.866	0.005

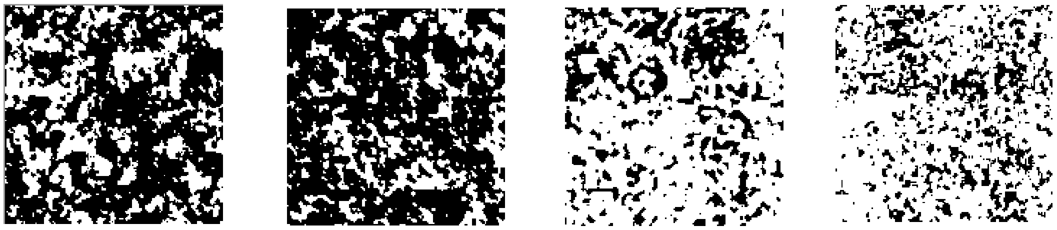
Figure captions

Fig. 3.1 Vegetation maps (3×3 m), which were produced from vegetation photography, from four pastures within block 2. Maps shown were all randomly chosen from the 10×10 m map. Black color indicates vegetation and white color shows bare ground. CK, control; LG, lightly grazed; MG, moderately grazed; HG, heavily grazed.

Fig. 3.2 Linear regression between stocking rate and (a) percentage cover (COVER, $P = 0.015$, $R^2 = 0.597$), (b) patch density (PD, $P = 0.002$, $R^2 = 0.793$), (c) mean patch area (AREA_MN, $P = 0.107$, $R^2 = 0.269$), (d) coefficient of variation (CV) of patch area (AREA_CV, $P = <0.001$, $R^2 = 0.874$) and (e) clumpiness index (CLUMP, $P = 0.004$, $R^2 = 0.740$).

Fig. 3.3 Effects of grazing on the patch-area (cm^2) distribution of vegetation in a desert steppe ecosystem, Inner Mongolia, China. CK, control; LG, lightly grazed; MG, moderately grazed; HG, heavily grazed. (a), block 1; (b), block 2. The best-fit model was provided (either power law, PL, or truncated power law, TPL) with the P-value of the sum of squares reduction test. The R^2 and scaling exponent, γ, of the best-fit model were also shown.

Fig. 3.4 Relationship between the scaling exponent, γ, of the best-fit model of the patch-area distribution and the stocking rate (filled circle, $P = 0.011$, $R^2 = 0.684$) and cover (empty circle, $P = 0.017$, $R^2 = 0.728$), respectively.



CK LG MG HG

Fig. 3.1 Vegetation maps, which were produced from vegetation photography, (3×3 m) from four pastures within block 2.

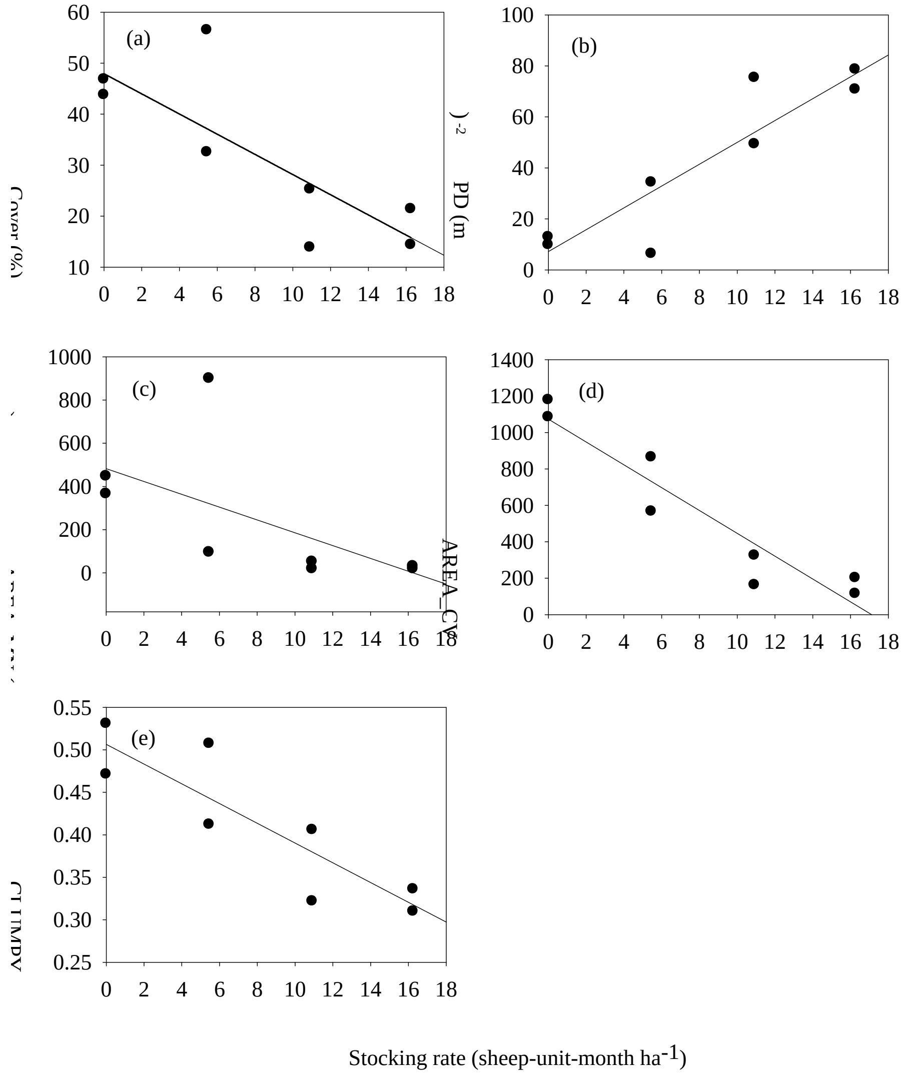


Fig. 3.2 Linear regression between stocking rate and (a) percentage cover, (b) patch density, (c) mean patch area, (d) coefficient of variation (CV) of patch area and (e) clumpiness index.

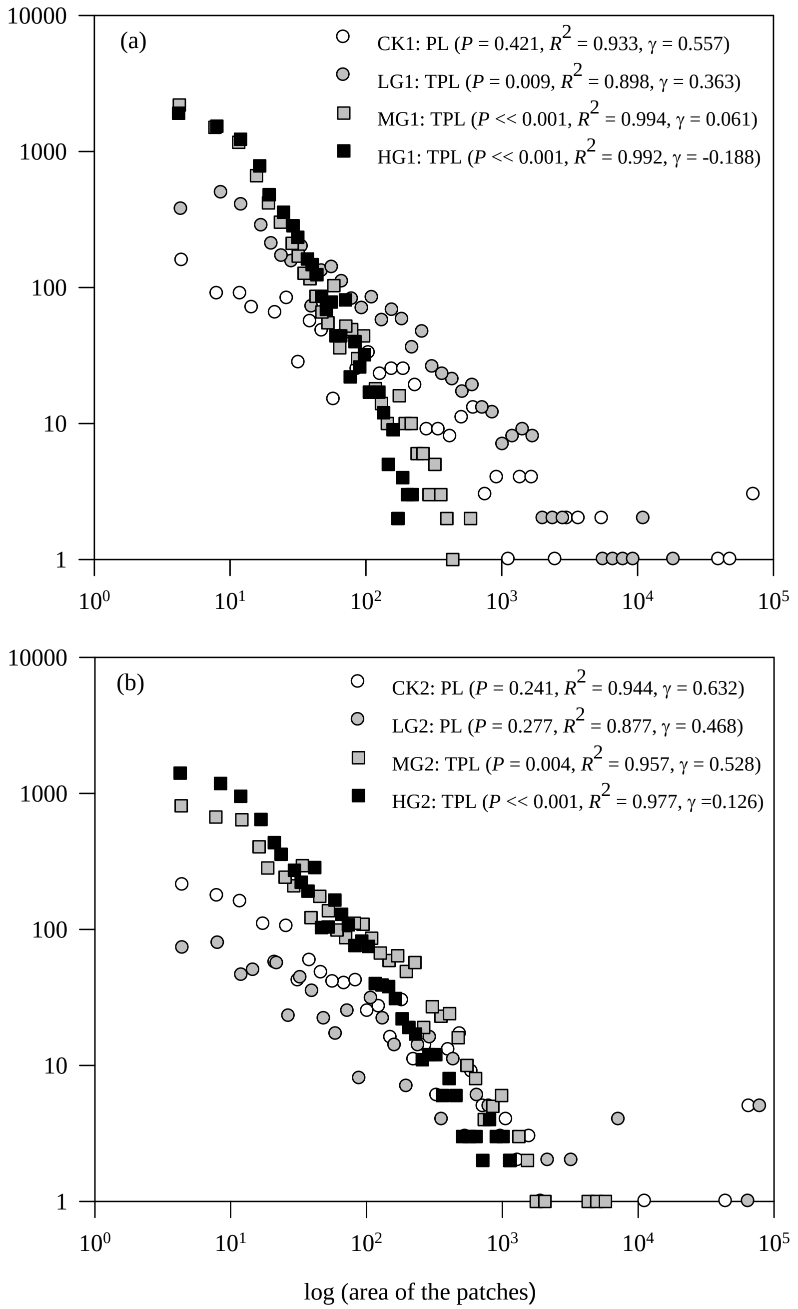


Fig. 3.3 Effects of grazing on the patch-area (cm^2) distribution of vegetation in a desert steppe ecosystem, Inner Mongolia, China.

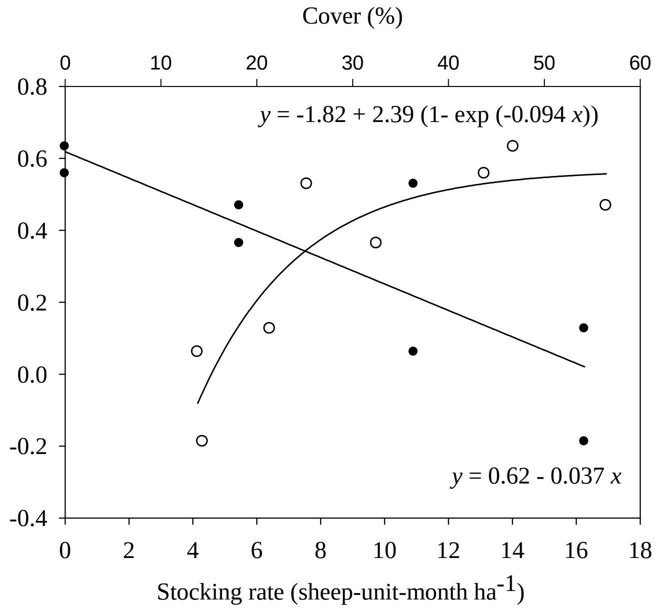


Fig. 3.4 Relationship between the scaling exponent, γ, of the best-fit model of the patch-area distribution and the stocking rate and cover, respectively.

4. General discussion and conclusions

4.1 Summary

Spatial patterns of vegetation and soil can have strong impacts on ecological functions of arid and semi-arid ecosystems. However, grazing has been variously shown to create, maintain or destroy those patterns. Grazing intensity can be important in interpreting the contradictory results, while few studies have linked grazing intensity to the changes of spatial patterns of vegetation and soil fertility. The recent literature has proposed to use the characteristics of spatial vegetation pattern to indicate early desertification processes. By now, few studies have attempted to test the performance of these indicators. In this dissertation, I examined the spatial patterns of vegetation and soil nutrients under different grazing intensities in a desert steppe in Inner Mongolia, China. The grazing gradient is also treated as an analog of a desertification gradient to evaluate the suitability of two predictors of desertification: percent plant cover and power-law modeling of vegetation patch-area distribution.

I found lightly grazed plots had similar percent plant cover, aboveground biomass production and soil nutrient availabilities as un-grazed plots, while heavy grazing strongly decreased percent plant cover, biomass production and soil NH_4^+ availability, suggesting grazing intensities have different effects on vegetation characteristics and soil fertility.

In terms of spatial vegetation pattern, most spatial metrics used in this study, such as patch density, coefficient of variation of mean patch area and clumpiness index, had linear relationships with stocking rate, suggesting that vegetation patches became smaller, more fragmented and homogeneous under higher grazing pressure. Geostatistical analyses of aboveground biomass also supported the decreasing heterogeneity of vegetation patches along the grazing gradient.

Heavy grazing decreased the heterogeneity of the spatial distributions of soil organic carbon and soil water content. In contrast, light grazing intensity created the spatial heterogeneity of soil mineral N and maintained a similar spatial pattern of soil organic carbon and total nitrogen as grazer exclosure. Loss of cross-correlation between AGB and soil NO_3^- in grazed plots indicates that grazing can directly alter the relationship between spatial patterns of vegetation and soil nutrient. Even though plant aboveground biomass had a different spatial pattern from other soil nutrients, including soil organic carbon and total nitrogen, my results suggest that changed spatial pattern of vegetation by grazing can still indirectly contribute to an altered soil nutrient spatial distribution by affecting plant turnover, soil loss and/or species composition.

Percent plant cover had strong correlations with stocking rate and most of the spatial metrics evaluated, indicating that plant cover reflected the change of the spatial pattern of vegetation along the desertification gradient. Patch-area distributions deviated from power law to truncated power law in one of the light grazed plots and in all plots under intermediate or heavy grazing pressure. These results suggest that both percent plant cover and power-law modeling could be used to indicate desertification processes in the studied desert steppe.

Overall, grazing intensities had distinct impacts on spatial patterns of vegetation and soil nutrients in this desert steppe. Heavy grazing generally had a negative effect on those spatial patterns, whereas light grazing maintained or even created patchy spatial patterns. Complex interactions between plant and soil induced different spatial patterns between vegetation and soil nutrients, but similar trends of spatial heterogeneity along the grazing gradient. This study supported that spatial vegetation pattern can provide insight to understand the degradation and desertification processes in arid and semi-arid ecosystems.

4.2 Management implications and future research

My results suggest that light grazing is essential in rangeland management to preserve biotic and abiotic spatial patterns, soil fertility and biodiversity. In fact, the two lightly grazed plots presented large variations in terms of spatial vegetation pattern, suggesting that light intensity grazing should also be combined with other management practices to maintain healthy vegetation structure and achieve sustainable forage production. On the contrary, heavy grazing intensity used in this study led to degradation of vegetation and soil fertility. Local management should avoid heavy grazing and restore rangelands that used to be heavily grazed.

Spatial patterns in arid and semi-arid ecosystems have not been well studied in a temporal scale. This study only provided a snap-shot about the rangeland along the grazing gradient and little is known about long-term (more than 10 years) effects of grazing intensity on the rangeland. Long-term study can provide more information about the feedbacks between spatial patterns and grazing intensity, which is extremely important for sustainable rangeland management. Long-term results are also valuable in better interpreting the highly varied responses of light grazed plots.

My results indicate that grazing intensity can have different impact on functions of ecosystems through changing spatial patterns of vegetation and soil nutrient availabilities. With other studies (Hobbs 1996, van de Koppel et al. 2002, Olofsson et al. 2008), these results underscore the importance of understanding grazing effects on spatial properties of rangeland. Future studies should consider linking the grazing-induced changes of spatial patterns to the processes and functions of ecosystems, such as greenhouse gas emissions, species colonization and water conservation.

Grazing can have strong interactions with other disturbances, such as climate change and fire, to affect the spatial patterns in arid and semi-arid ecosystems (Maestre and Reynolds 2006). However, little is known about the mechanisms within these interactions. With increasing disturbances on rangelands, multiple-factor experiments can help local management to mitigate the impacts of disturbances. Simulation models are also useful tools in understanding the complex feedbacks of rangelands to multiple disturbances.

4.3 References

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