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## Effect of long-term grazing on soil organic carbon content in semiarid steppes in Inner Mongolia

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**Abstract** To clarify the response of soil organic carbon (SOC) content to season-long grazing in the semiarid typical steppes of Inner Mongolia, we examined the aboveground biomass and SOC in both grazing (G-site) and no grazing (NG-site) sites in two typical steppes dominated by *Leymus chinensis* and *Stipa grandis*, as well as one seriously degraded *L. chinensis* grassland dominated by *Artemisia frigida*. The NG-sites had been fenced for 20 years in *L. chinensis* and *S. grandis* grasslands and for 10 years in *A. frigida* grassland. Aboveground biomass at G-sites was 21–35% of that at NG-sites in *L. chinensis* and *S. grandis* grasslands. The SOC, however, showed no significant difference between G-site and NG-site in both grasslands. In the NG-sites, aboveground biomass was significantly lower in *A. frigida* grassland than in the other two grasslands. The

SOC in *A. frigida* grassland was about 70% of that in *L. chinensis* grassland. In *A. frigida* grassland, aboveground biomass in the G-site was 68–82% of that in the NG-site, whereas SOC was significantly lower in the G-site than in the NG-site. Grazing elevated the surface soil pH in *L. chinensis* and *A. frigida* communities. A spatial heterogeneity in SOC and pH in the topsoil was not detected the G-site within the minimal sampling distance of 10 m. The results suggested that compensatory growth may account for the relative stability of SOC in G-sites in typical steppes. The SOC was sensitive to heavy grazing and difficult to recover after a significant decline caused by overgrazing in semiarid steppes.

**Keywords** Aboveground biomass · Light grazing · Carbon storage · Grasslands · Compensatory growth

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### Introduction

Grasslands of various types cover approximately 25.4% of total land area (IPCC 2000). Most carbon (C) in grassland ecosystems is stored in the soil, which accounts for about 39% of the terrestrial inventory (IPCC 2000). The large proportion of soil C storage indicates that grassland ecosystems have a high capacity to sequester C in soil. Some studies show that world grassland is currently an active C sink (Thornley et al. 1991; Fisher et al. 1995) and its size may increase greatly in the future (Scholes and Hall 1996; Scurlock and Hall 1998). On the other hand, grassland ecosystems are fragile and generally sensitive to climate change and human activities (UNEP 1993). Hence, the huge C stock in grassland soil may become a latent risk of large C emission into the atmosphere under future climate conditions (Parton et al. 1995) or improper management (Batjes 1999).

Grazing is one of the most important factors that could change the soil C stock in grassland ecosystems. Today, natural grasslands are very rare or even non-

existent (IUCN 1994). Slight or moderate grazing intensity is generally beneficial to maintain biodiversity and aboveground biomass production of grassland ecosystems as compared with grazing exclusion (McNaughton 1979; Hik and Jefferies 1990; Collins et al. 1998; Frank et al. 2003). Overgrazing, however, leads to reduction of grassland production (Stuth et al. 1996; Wang and Ripley 1997; Shang et al. 2003). It is well known that soil C storage depends on the C input mainly through the belowground parts of vegetation and C release mediated by soil processes, which are influenced by soil physical, chemical, and biological properties (Batjes 1999). Belowground processes may respond differently from aboveground vegetation to grazing. Frank et al. (1995) showed that change of community structure induced by grazing did not necessarily lead to decreased soil C storage. The ratio of C allocation to shoot and root also responded differently to animal grazing (Caldwell et al. 1981; Holland et al. 1992; Biondini et al. 1998; Leriche et al. 2001; Reeder and Schuman 2002). The effect of grazing intensity on soil physical, chemical and biological properties differed greatly among soil categories, vegetation types, grazing animals, and other factors (Lavado et al. 1995). Up to now, there is still a discrepancy regarding the response of soil C storage to grazing pressure (Conant et al. 2001b; Reeder and Schuman 2002, and references therein). For example, under moderate grazing pressure, soil C varied from a decrease of 58% to an increase of 170% during the experimental period. The absolute change of soil C storage is in the range of a decrease of  $2.40 \times 10^4$  kg C ha<sup>-1</sup> to an increase of  $4.98 \times 10^4$  kg C ha<sup>-1</sup> (Conant et al. 2001a).

The typical steppes of Inner Mongolia are located in the east of the vast Euroasian grassland. Grazing has a long history in Inner Mongolia grasslands and is still one of the main management regimes. However, with the transition from nomadism to settlement of herdsman from the late 1940s, herd quantity increased greatly and distribution patterns of herds changed substantially. Consequently, grazing pressure was generally elevated, particularly around settlements. The herd quantity in Inner Mongolia rapidly increased from 19.3 million in 1947 to 74.1 million sheep units in 1975 and then oscillated around 75.0 million sheep units thereafter. The grazing patterns could change the soil C dynamics in this grassland ecosystem, one of the largest temperate grasslands in the world. Some showed a general trend of soil C loss by grazing (Li et al. 1998), while other studies reported a positive effect on soil C storage by grazing (Wang and Chen 1998; Chuluun et al. 1999). However, little detailed information is available for our understanding or predicting the soil C stock of this temperate grassland.

Fenced non-grazing sites have been established for 20 years in two typical steppes, *Leymus chinensis* and *Stipa grandis* grasslands, in Inner Mongolia (L-NG and S-NG sites). Former studies indicated that the neighboring grazing sites (L-G and S-G sites) were subject to

light grazing intensity (Xiao et al. 1997). In researches concerning grazing in Inner Mongolia, non-grazing, light-grazing, moderate-grazing, and heavy-grazing referred to grazing pressure corresponding to locations from enclosure to dwelling or water points (Li and Wang 1997). Nevertheless, the aboveground standing living biomass in the NG-sites contrasted remarkably with the corresponding G-sites and could be easily recognized in satellite images (Xiao et al. 1997). This paper focuses on comparison of soil organic carbon (SOC) in NG-sites with G-sites in order to address whether long-term season-long grazing reduced SOC in the semiarid steppes. A third grassland (A-NG and A-G sites), degraded from *L. chinensis* steppe by overgrazing, was also included to determine if there was difference in response of SOC to grazing by natural and degraded grasslands.

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## Materials and methods

### Experimental treatment and site description

The experiment was conducted in three grasslands, two natural ones, *L. chinensis* and *S. grandis* steppes, and a degraded one, an *Artemisia frigida* grassland, each with a fenced non-grazing site (L-NG, S-NG, and A-NG sites) and a neighboring grazing site (L-G, S-G, and A-G sites). The L-NG and S-NG sites (25 ha each) were fenced in 1980 and the A-NG site was established in *A. frigida* grassland in 1991, after serious degradation from *L. chinensis* grassland.

The study site (43°32'N and 116°40'E) is in the Xilin River basin where the UNESCO/MAB Xilingol Biosphere Reserve is located. *L. chinensis* grassland and *S. grandis* grassland are typical dominant steppes in this area. Xiao et al. (1997) has given a detailed description of vegetation, soil, and climate of the studied sites. In brief, the dominant species are *L. chinensis*, *S. grandis*, *Koeleria cristata* and *Agropyron cristatum* in the *L. chinensis* grassland. *S. grandis* is the dominant species at the *S. grandis* site. The degraded *L. chinensis* grassland is dominated by *A. frigida* and *Cleistogenes squarrosa*. The soils at all the sites are Kastanozems. They are without a distinct CaCO<sub>3</sub> layer at the *L. chinensis* and *A. frigida* sites and with a clear CaCO<sub>3</sub> layer below 50 cm at the *S. grandis* site. The average annual temperature is 0.2°C and annual precipitation is 350 mm at the study site. The growing season is from late April to early October. Soil bulk density at 0–20 cm was 1.09, 1.16, and 1.19 g cm<sup>-3</sup> in the L-NG, S-NG, and A-NG sites, and 1.16, 1.21, and 1.32 g cm<sup>-3</sup> in L-G, S-G, and A-G sites, respectively. In other soil layers, the G-site had similar soil bulk density to the corresponding NG-site in all the three grasslands. On average, soil texture is 21% clay, 19% silt, and 60% sand in the *L. chinensis* site and 21% clay, 30% silt, and 49% sand at the *S. grandis* site (Xiao et al. 1997). Soil texture has not been determined at the *A. frigida* site.

## Collection and preparation of soil samples

Soil samples were taken from the NG-sites and the nearby G-sites at each of the three grasslands on 18 May and 18 September 2000. For each sampling site, five 1×1 m quadrats were determined randomly. Three points were randomly taken with a soil auger in each quadrat, and the samples were collected every 20 cm to a soil depth of 80 cm. For each layer the soils from the three drills were mixed into a bulk sample. A 60–80 cm layer was only taken in *L. chinensis* and *A. frigida* grasslands in May. It was too dry for soil sampling in September and in the *S. grandis* grassland. On 12 August, soil sampling (0–10 cm) was conducted at 10 m intervals along two 1,200 m long transects at the NG-site and the neighboring G-site in *L. chinensis* grassland. The data showed no significant spatial auto correlation for SOC or soil pH in either of the grasslands (Fig. 1, analyzed with the geo-statistical methods described in Pannatier 1996). Based on the 121 samples from each site, we calculated that five random samples were sufficient to ensure that the mean of SOC and pH were within  $\pm 10\%$  of the respective expectation with 95% confidence.

The samples were air dried in the laboratory and ground to pass a 100-mesh screen. Tiny litter or roots were removed through adsorption by a plastic rod rubbed with a silk cloth before determination of chemical properties.

## Determination of aboveground biomass of vegetation

Together with soil sample collection, aboveground plant biomass was gathered in five quadrats from each site. Plants were cut above the soil surface. Litter and standing dead parts were removed. The living plant parts were oven-dried at 80°C to constant weight and ground for analysis. In August, five 1×1 m quadrats were randomly selected in each of these transects in *L. chinensis* grassland.

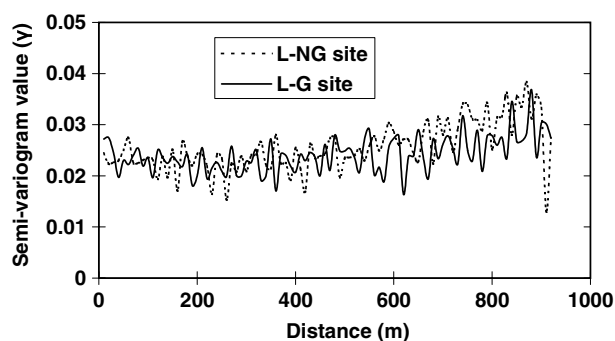


Fig. 1 Semi-variogram of soil total C content in NG-site and G-site location in *L. chinensis* grassland. The shortest lag was 10 m

## Analysis of soil and plant chemical properties

Soil total C content was measured with an EC-12 Carbon Determinator (LECO, USA). Soil was weighed to about 0.3000 g and mixed with Cu and Fe powder. The sample was then combusted in a high frequency furnace and the released CO<sub>2</sub> content was measured by means of an IR cell. Soil inorganic C was determined by adding 25 ml of 3 mol l<sup>-1</sup> HCl solution to 5.0 g soil and measuring the volume of the released CO<sub>2</sub> after shaking for 4 min. The SOC was calculated by subtracting the inorganic C content from the total C content. Soil pH was measured by means of a digital pH meter (Metrohm E500) in a suspension of 10.0 g soil and 25.0 ml 0.01 mol l<sup>-1</sup> CaCl<sub>2</sub> solution.

Plant samples (2.000 g) were digested in a muffle furnace at 480°C overnight. The residue was resolved with 10 ml of 4 mol l<sup>-1</sup> HNO<sub>3</sub> for 2 h and diluted to 50 ml with deionised water. In the filtrated solution the phosphorus (P) and potassium (K) contents were measured. The P was determined by spectrometry after reacting with ammonium molybdate reagent and K was determined by flame photometry (Eppendorf ELEX 6361).

## Estimation of SOC storage

The SOC storage was calculated by summing the product of SOC content in each soil layer and the corresponding bulk density within an 80 cm depth. As bulk density was not determined simultaneously, mean bulk density and mean SOC content were used for each soil layer. Therefore, standard deviation could not be calculated, as shown in Fig. 2.

## Data analysis

SPSS and StatView were used for statistical evaluation to compare the mean values of plant biomass, soil C and pH between G-sites and NG-sites in each grassland. VARIOWIN was adopted for geo-statistic

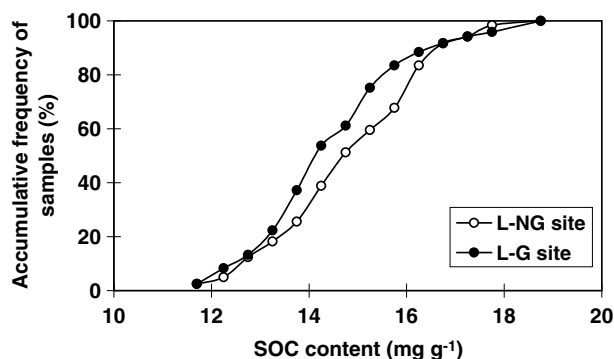


Fig. 2 Accumulative frequency distribution of SOC samples in 0–20 cm soil layer in NG-sites and G-sites in *L. chinensis* grassland

**Table 1** Distribution of SOC content along soil profile in the three grasslands in grazing (G-site) and non-grazing (NG-site) sites. Samples taken in May and September were treated as replicates in statistical analysis because soil organic content was relatively constant within such short times and no significant difference was detected between these two sampling periods. The *t*-test was performed to compare means of SOC content in the

Layer(cm)	Soil organic carbon content (mg g <sup>-1</sup> soil)						
	<i>L. chinensis</i>		<i>S. grandis</i>		<i>A. frigida</i>		
	NG-site	G-site	NG-site	G-site	NG-site	G-site	G/NG
0–20	16.17±1.76	16.71±1.36	15.12±1.70	14.84±0.73	11.64±3.11*	8.34±2.15	0.72
20–40	9.90±0.61	9.86±0.36	7.57±2.05	9.06±3.53	6.17±2.77***	2.74±0.67	0.44
40–60	8.48±0.24	8.76±0.67	5.68±0.83	5.63±1.03	4.46±2.00*	2.40±1.34	0.54
60–80	8.04±0.28	8.11±0.15	–	–	4.71±1.32**	1.63±0.78	0.35

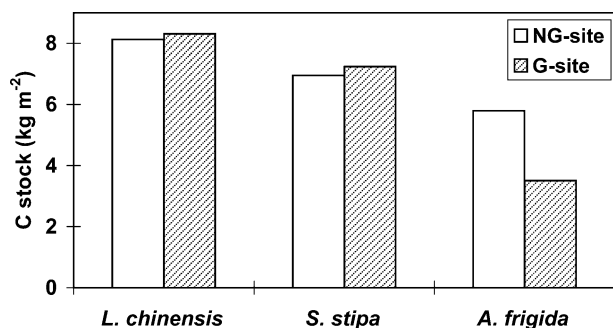
calculation. Lag tolerance was set to 0.5 (5 m). Spherical and Gaussian models were used to fit the semi-variogram curves. Only the points with more than 30 data pairs were used in diagrams and calculations.

## Results

### Soil organic carbon content

The SOC content was relatively constant during growing seasons. It was significantly higher in the surface layer than the lower soil layers in all the three sites ( $P < 0.0001$ , Table 1). The ratio of SOC content in the 0–20 cm layer to that in the 20–40 cm layer was higher in L-G sites than in L-NG sites (1.70 vs 1.49,  $n = 20$ ,  $P = 0.129$ ) and in A-G than in A-NG sites (3.10 vs 2.00,  $n = 20$ ,  $P < 0.0001$ ). The difference between S-G and S-NG sites was unclear (1.80 vs 2.44,  $n = 20$ ,  $P = 0.308$ ).

Grazing regime did not significantly alter SOC content (Table 1) or SOC stock (Fig. 3) in *L. chinensis* and *S. grandis* grasslands; neither did grazing induce significant change in the spatial pattern (Fig. 1) or distribution of SOC along soil profile in either grassland (Table 1).



**Fig. 3** Soil organic carbon stock in the upper 60 cm soil layers in *L. chinensis*, *S. grandis*, and *A. frigida* grasslands under grazing and non-grazing treatments

same soil layer in NG-site and G-site in each of the three grasslands. \*, \*\*, and \*\*\* denote difference at the level of 0.05, 0.01, and 0.001, respectively. No significant difference was detected between the NG-site and the G-site in any soil layer in *L. chinensis* and *S. grandis* grasslands. Soil depth was less than 60 cm in *S. grandis* grassland and no samples were taken for the 60–80 cm layer

The *A. frigida* dominated degraded grassland contained significantly lower SOC content in every soil layer than that in *L. chinensis* grassland in NG-sites ( $P < 0.01$ , Table 1). Its SOC stock was approximately 70% of that in *L. chinensis* grassland in NG-sites (Fig. 3). Compared with the NG-site in *L. chinensis* grassland, SOC in NG-site in *A. frigida* grassland was 38–48% and 28% lower in 20–80 cm layers and the surface layer (0–20 cm, Table 1) respectively. Grazing induced a further decrease of SOC by about 40% in *A. frigida* grassland (Fig. 3). In this grassland, grazing caused a significant decrease of SOC for all the soil layers (Table 1), and the declination was more remarkable in lower soil layers than in the surface soil layer, as shown by the rate of SOC in the G-site and NG-site (G/NG in Table 1).

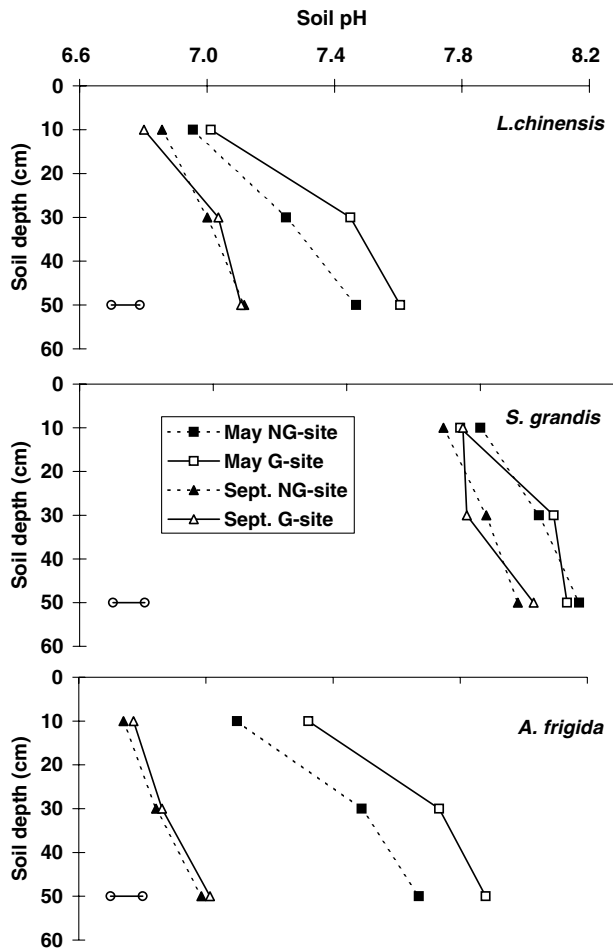
### Soil pH

Soil pH tended to increase from the surface layer downward. It was significantly different between May and September. Soil pH in *S. grandis* grassland was higher than in the other two grasslands and did not show an obvious response to grazing regime (Fig. 4). Although not statistically significant, grazing induced an increase in soil pH in both *L. chinensis* and *A. frigida* communities in May (Fig. 4). In the *L. chinensis* community, grazing led to a significantly higher soil pH at the surface soil layer in August (6.66 in the G-site vs 6.52 in NG-site,  $n = 242$ ,  $P < 0.05$ ). Spatial samples showed a negatively weak correlation between pH and SOC in surface soil ( $R = -0.179$ ,  $n = 242$ ,  $P = 0.007$ ).

### Plant aboveground biomass, P and K contents

The aboveground biomass was in general lower in G-sites than NG-sites, but the grazing effect differed in grasslands dominated by different species. Grazing resulted in a significant decrease by 65–79% of aboveground





**Fig. 4** Soil pH along soil profile of *L. chinensis*, *S. grandis*, and *A. frigida* grasslands under grazing and non-grazing treatment in May and September. SD of data set for each point is not presented. The SE calculated by LSD method was given as the short bar in each diagram for clarity

**Table 2** Comparison of aboveground plant biomass of *L. chinensis*, *S. grandis*, and *A. frigida* grasslands in grazing (G-site) and non-grazing (NG-site)-sites. The *t*-test was performed to compare means of biomass between the G-site and the NG-site sampled on the same date in each of the three grasslands. \* and \*\*\* denote difference at the level of 0.05 and 0.001, respectively

Grasslands indicated by dominant species	Grazing regime	Biomass (g m <sup>-2</sup> )	
		August 13	September 19
<i>L. chinensis</i>	NG-site	157.2 ± 31.4***	152.1 ± 30.3
	G-site	49.4 ± 21.2	150.9 ± 31.7
<i>S. grandis</i>	NG-site	105.0 ± 24.0***	75.4 ± 8.3*
	G-site	21.9 ± 7.4	26.7 ± 4.8
<i>A. frigida</i>	NG-site	81.4 ± 14.3	80.2 ± 7.0
	G-site	55.2 ± 31.7	65.80 ± 4.4

biomass in *L. chinensis* and *S. grandis* communities, as compared with the NG-sites in August (Table 2).

Aboveground living biomass of plant communities generally peaked around late August in Inner Mongo-

lian steppe. Therefore, there was no significant difference in aboveground biomass measured on August 13 and September 19 in all the sampling-sites, except for G-site in *L. chinensis* grassland (Table 2). Plant aboveground biomass increased by about two times and became similar to that in the fenced-site in this case, owing to exclusion of grazing by establishing a new fence immediately after the first biomass measurement in the G-site in *L. chinensis* grassland.

Grazing significantly reduced plant P and K content in May in *L. chinensis* and *A. frigida* grasslands. It induced a declination of K content by 29–49% in August. Nevertheless, plant P content was similar between G-sites and NG-sites in August (Table 3). Current stock of P and K in aboveground biomass was much higher in NG-site than in grazing locations in all the three vegetation types.

## Discussion

The SOC is determined by C input mainly from aboveground litter production, root turnover and animal excreta, and C output through soil respiration, soil erosion and leaching. Soil erosion and leaching loss of SOC were not significant in the semiarid steppes (Li et al. 2004).

Effect of grazing on aboveground NPP, and consequently litter production, varied remarkably in different experiments. Many studies have observed maintenance or stimulation of aboveground NPP under grazing, which is termed compensatory or over-compensatory growth response to grazing (McNaughton 1979; Hik and Jefferies 1990; Pandey and Singh 1992; Leriche et al. 2001). The underlying mechanisms included stimulation of tillering (Briske and Richards 1995), reduction of self-shading and promotion of photosynthesis in remaining tissues (Doescher et al. 1997). In this study, a large reduction of standing living biomass was also observed in G-sites of all the grasslands (Table 2). In Inner Mongolian grazing regimes, animals browse grasslands daily even on days with snow cover. Aboveground biomass was greatly depressed throughout the whole growing season even under light grazing, which differed remarkably from other studies where vegetation recovered largely or fully after grazing in rotational grazing regimes (McNaughton 1983; Manske 1999). The data for *L. chinensis* grasslands clearly indicated a compensatory growth after exclusion of grazing (Table 2). Former studies showed that cutting stimulated photosynthetic capacity of the remaining part of leaves in this species by 25 and 40% and of recruited leaf by 68 and 55% in strong and weak light (Cui 2000). The large increase of C fixation capacity interacting with reduction of self-shading may lead to similar or even higher vegetation production in G-sites than NG-sites. Soil nutrient contents were reported to be similar in *L. chinensis* and *S. grandis* grasslands, and they did not differ significantly between NG-sites and lightly grazed locations

**Table 3** Aboveground plant P and K content in May and August as well as current stock of P and K in aboveground biomass in August in *L. chinensis*, *S. grandis*, and *A. frigida* grasslands in grazing (G-site) and non-grazing (NG-site)-sites. Statistics was

performed only for contents of the same element in the same grassland. Different letters denote significant difference at  $P=0.05$  with Duncan's post hoc test

Site	Treatment	P content (mg g <sup>-1</sup> )		K content (mg g <sup>-1</sup> )		Stock in plant (mg m <sup>-2</sup> )	
		May	August	May	August	P	K
<i>L. chinensis</i>	NG-site	2.30 ± 0.12a	1.10 ± 0.17c	22.77 ± 3.27a	11.64 ± 1.81c	86.6	915.6
	G-site	1.88 ± 0.21b	0.97 ± 0.03c	15.39 ± 0.68b	9.40 ± 1.28c	23.9	232.0
<i>S. grandis</i>	NG-site	1.88 ± 0.19a	0.74 ± 0.08c	15.51 ± 1.37a	11.37 ± 1.79b	38.6	597.2
	G-site	1.75 ± 0.07a	0.92 ± 0.08b	14.31 ± 1.02a	5.83 ± 0.61c	10.1	63.9
<i>A. frigida</i>	NG-site	1.84 ± 0.30a	1.41 ± 0.21b	10.36 ± 2.81ab	13.85 ± 1.72a	57.6	563.5
	G-site	1.13 ± 0.35b	1.42 ± 0.24b	5.70 ± 3.20c	9.82 ± 1.73b	39.1	271.0

in either grassland (Guan et al. 1997). Therefore, *S. grandis* grassland was likely to have compensatory growth because soil nutrient status mainly controlled the direction and degree of responses of biomass production and soil biological processes to grazing (Milchunas et al. 1988; Leriche et al. 2001; Harrison and Bardgett 2004). Over-compensation was also observed in *A. frigida* grassland under light grazing pressure (Wang et al. 1998).

Belowground NPP was more than three times aboveground NPP in *L. chinensis* and *S. grandis* grasslands (Li et al. 2002b). Since daily browsing by herds removed most of the litter, organic matter input into the soil should principally depend on root turnover and animal excreta in G-sites. Some studies reported that grazing depressed belowground NPP and root biomass (Holland et al. 1992; Schuman et al. 1999), whereas others did not observe detectable effects (Holland et al. 1992; Biondini et al. 1998; McNaughton et al. 1998). In grazing sites, roots rapidly proliferate in localized soil regions of high mineralization, such as sites with dung and urine (Jackson and Caldwell 1993). Root/shoot allocation was reported to decrease for grazing tolerant species, increase or remain unchanged in other species (Caldwell et al. 1981; Holland et al. 1992; Pandey and Singh 1992; Biondini et al. 1998; McNaughton et al. 1998). The dominant species in natural steppes in Inner Mongolia, *L. chinensis* and *S. grandis*, were grazing sensitive. The ratio of root/shoot was significantly higher in the heavily grazed location than in NG-sites in *L. chinensis* grassland (14.38 in the heavily grazed site vs 9.26 in the NG-site, Li et al. 2000). As mentioned before, compensatory growth was common in the G-sites. Since aboveground growth cannot increase or be maintained when root growth is inhibited (McNaughton et al. 1998), it is reasonable to assume that root productivity and turnover was maintained or stimulated in G-sites in the grasslands studied. This was supported by a study in *L. chinensis* grassland (Li et al. 2002b).

Grazing decreased soil respiration in a tallgrass prairie and an alpine meadow (Bremer et al. 1998; Cao et al. 2004) but did not decrease it in shortgrass steppe or northern mixed prairie in the USA (Mayeux et al. 2002). In the studied steppe, soil respiration was highly

correlated with soil moisture in the *L. chinensis* (Li et al. 2000) and *S. grandis* (Cui et al. 2000) grasslands. Root biomass showed an insignificant correlation with soil respiration (Cui et al. 2000; Li et al. 2002a). Grazing led to a decrease in soil moisture status, increased soil bulk density and surface soil hardness (Jia et al. 1997). Soil pH is one of the important environmental factors that influence SOC accumulation and decomposition (Trumbore 1997; Nott et al. 1998; Sjögersten et al. 2003). Topsoil pH was found to closely correlate with precipitation at a large spatial scale in arid areas. Soils under less precipitation have higher pH due to higher content of cations and CaCO<sub>3</sub> in topsoil layers (Schillinger et al. 2003; Bhattacharyya et al. 2004). Such correlation also exists in terms of temporal scale. For instance, soils in Hunsandak in Inner Mongolia had significantly lower pH 4,000–8,000 years BP than that at present, indicating more humid conditions at that time (Zhou and Zhang 1992). In this manuscript, pH at 0–10 cm soil was significantly higher in May than in September in *L. chinensis* and *A. frigida* (Fig.4) grasslands, because of precipitation during the growing season. Generally, soil drying may induce pH increase in top layers in arid zones. Animal grazing decreased land cover of vegetation and caused soil compaction. Water loss increased through evaporation owing to higher temperature and wind at soil surface, and through runoff owing to soil compaction. Surface soil drying was observed in grazed grasslands (Dolan and Taylor 1972). Therefore, an elevation of surface soil pH reflected a decrease in soil humidity in *L. chinensis* and *A. frigida* grassland following grazing (Fig.4). All these factors may reduce soil respiration in G-sites. The maintenance of soil respiration in heavily grazed-sites was caused by either increased soil C content (Mayeux et al. 2002) or a large deposit of animal excreta (Li et al. 2000).

The SOC content was little influenced by light grazing in *L. chinensis* and *S. grandis* grasslands in this study (Table 1). As analyzed previously, compensatory growth induced higher translocation of assimilated C to belowground, as well as lower soil respiration loss of SOC was supposed to largely account for the relative stability of SOC under light grazing.

While total SOC did not change obviously, its labile component may decrease substantially under grazing. Such a feature was observed in a mixed prairie under moderate grazing (Holt 1997). The active part of soil organic matter lost the greatest proportion in shortgrass steppe after reducing plant inputs (Kelly et al. 1996). Bacteria number decreased by more than 57% after light grazing in *L. chinensis* grassland (Liu and Liao 1997). Labile SOC content was 0.207% in the L-NG site (Wang et al. 2003). This accounted for approximately 12% of total SOC, within the reported range of from 2 to more than 50% in semiarid grasslands (Chan 1997; Gill et al. 1999; Kaye et al. 2002).

The SOC content in the typical steppes in Inner Mongolia was suggested to be sensitive to grazing (Xiao et al. 1996; Ojima et al. 1999). The SOC was reduced in *L. chinensis* grassland in northwestern China under grazing (Wang and Ripley 1997). In this study, the G-sites were near to the NG-sites and far away from either dwellings or water points. Based on grassland area, herd quantity, and distribution, the grazing intensity in G-sites was estimated to be around one sheep unit per hectare. The much lower value of SOC in the NG-site of *A. frigida* grassland than that in *L. chinensis* indicated that overgrazing led to a significant loss of SOC that was difficult to recover from in the grasslands (Table 1). The SOC continued to decrease under light grazing in the degraded grassland (Table 1), though aboveground biomass was removed by a much lower proportion due to grazing in this grassland than in the others (Table 2). Greater organic C loss occurred in the lower soil layers than in the surface soil layer (Table 1). This was mainly due to the change of dominant species from *L. chinensis* to *A. frigida* and *C. squarrosa*. The latter species had less root quantity and a shallower distribution in soil (Wang and Wang 2001). Root productivity may decrease as a result of depression in aboveground NPP. These species were grazing tolerant and the ratio of root/shoot decreased in grazed locations (13.60 vs 9.64 in A-NG and A-G sites, Wang and Wang 2001). Consequently, C input through root turnover and exudation also reduced, which had a greater effect on deep soil layers than the surface layer where most of roots were distributed. Furthermore, change of soil physical properties, such as increase of hardness and bulk density, limited water and SOC transportation into lower soil layers.

In summary, SOC content and spatial heterogeneity did not change obviously with the current grazing regime under light grazing pressure for 20 years in the typical steppe of Inner Mongolia (Figs. 1, 3, Table 1). Nevertheless, subtle influences of long-term grazing could be discerned. For example, nutrient cycling was depressed to some extent. The P and K content and stock in aboveground phytomass were sometimes significantly lower in G-sites (Table 3). The decrease of recycling through litter production and removal from the grassland by animal production decreased available soil nutrient content. Where the active component of SOC decreased in the G-sites, soil available nitrogen

content also declined because N mineralization rate was the highest in this component (Paul 1984; Kerek et al. 2003). Besides, higher surface soil pH in the *L. chinensis* and *A. frigida* grasslands also demonstrated a tendency of deteriorating soil physical properties and decreasing SOC induced by grazing (Fig. 4). High soil pH, in turn, had negative effects on nutrient availability and uptake (Marschner 1995), and consequently on plant compensatory growth (Proulx and Mazumder 1998). Although seasonal samples did not detect significant changes of SOC (Table 1), intensive sampling by transects did show a 3% decrease ( $P=0.072$ ,  $n=242$ ) by long-term grazing in *L. chinensis* grassland (Fig. 2). This study, therefore, suggests that long-term observation is needed to clarify SOC dynamics under light grazing. Nevertheless, SOC was sensitive to overgrazing and SOC was slow to recover merely by enclosure when it declined following overgrazing in the semiarid typical steppes in Inner Mongolia.

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