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Carbon and nitrogen store and storage potential as affected by land-use in a *Leymus chinensis* grassland of northern China

Nianpeng He [a,b,*], Qiang Yu [a], Ling Wu [a], Yuesi Wang [b], Xingguo Han [a]

[a] State Key Laboratory of Vegetation and Environmental Change, Institute of Botany, CAS, 20 Nanxincun Xiangshan, Beijing 100093, China
[b] LAPC, Institute of Atmospheric Physics, CAS, Beijing 100029, China

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A B S T R A C T

Understanding the store and storage potential of carbon (C) and nitrogen (N) helps us understand how ecosystems would respond to natural and anthropogenic disturbances under different management strategies. We investigated organic C and N storage in aboveground biomass, litter, roots, and soil organic matter (SOM) in eight sites that were floristically and topographically similar, but which had been subjected to different intensities of disturbance by grazing animals. The primary objective of this study was to ascertain the impact of grazing exclusion (GE) on the store and storage potential of C and N in the *Leymus chinensis* Tzvel. grasslands of northern China. The results revealed that the total C storage (including that stored in aboveground biomass, litter, roots, and SOM, i.e. top 100-cm soil layer) was significantly different among the eight grasslands and varied from 7.0 kg C m^{-2} to 15.8 kg C m^{-2}, meanwhile, the total N storage varied from 0.6 kg N m^{-2} to 1.5 kg N m^{-2}. The soil C storage decreased substantially with grassland degradation due to long-term heavy grazing. 90% C and 95% N stored in grasslands were observed in the SOM, and they were minor in other pools. The limit range of C and N storage observed in these grassland soils suggests that GE may be a valuable mechanism of sequestering C in the top meter of the soil profile.

1. Introduction

Human activity has adversely affected global C and N cycles, and contributed to an alteration of climate that will generate discernible feedbacks to all organisms and ecosystems on earth. In recent decades, extensive work has been conducted toward improving our understanding of global C reserves and quantifying the pools and fluxes that constitute the cycles. Since the amount of C stored in SOM is approximately twice that in the atmosphere (Schimel, 1995), the accumulation of C in the terrestrial biosphere could partially offset the effect of anthropogenic carbon dioxide (CO_2) emissions at the atmospheric CO_2 level (Houghton et al., 1999).

Grasslands in some regions may serve as important global C sinks, and the annual sink is approximately 0.5 Pg C for tropical grasslands where the rates mainly depended on the baseline SOC level and annual precipitation (Davidson et al., 1995; Scurlock and Hall, 1998; Tan et al., 2006). In the past few decades, it has become clear that the C storage in grasslands has been significantly affected by changes in land-use and various ecosystem management strategies (Lugo and Brown, 1993; Post and Kwon, 2000; Jones and Donnelly, 2004; Billings, 2006; Elmore and Asner, 2006; Liao et al., 2006). Improved management practices – such as soil fertilization, promotion of native vegetation, and sowing of legumes and grasses – can increase soil C content and concentration (Conant et al., 2001). Since the C and N cycles in soil are tightly coupled, there is substantive concern that land-use change and its integrative effect can lead to an alteration in soil C and N storage (Houghton et al., 1999).

The potential of C and N storage is difficult to quantify in the field, because it requires studying stable and mature ecosystems – few of which now remain (Smithwick et al., 2002). Understanding potential C and N storage capacities will help us to (i) predict the quantity of C and N that can be sequestered by specific terrestrial ecosystems and (ii) assess the impact of natural and anthropogenic disturbances on C and N storage (Knops and Tilman, 2000; Smithwick et al., 2002). Following the economic reforms that have taken place in China since the late 1970s, the rapid expansion of livestock counts has caused significant degradation or desertification of the temperate grasslands of northern China (Li, 1994; Dong and Zhang, 2005). Nowadays, improved grassland management, particularly GE, is a common measure to protect the *Leymus chinensis* Tzvel. grasslands that are widely distributed in northern China (Chen and Wang,

* Corresponding author. State Key Laboratory of Vegetation and Environmental Change, Institute of Botany, CAS, 20 Nanxincun Xiangshan, Beijing 100093, China. Tel.: +86 10 62836581; fax: +86 10 82595771.
E-mail address: nphe@ibcas.ac.cn (N. He).

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2000). An increase in the soil C and N storage in the grasslands of northern China is anticipated with the implementation of measures aimed at encouraging grassland protection. However, there is a dearth of information regarding the potential of C and N storage due to the absence of stable or mature grassland ecosystems. We surveyed the C and N store and their storage potential in the *L. chinensis* grasslands in northern China using eight grassland plots ranging from severely degraded to those in which grazing has been excluded for approximately 28 years. The objective of this study was to assess the effect of GE on the store and storage potential of C and N in the *L. chinensis* grasslands of northern China.

2. Methods and materials

2.1. Study sites

This study was conducted in a typical steppe ecosystem in the Inner Mongolia Grassland Ecosystem Research Station (IMGERS), which is associated with the Chinese Academy of Sciences (CAS); the ecosystem lies between lat. 43°26′ N and 44°08′ N and long. 116°04′ E and 117°05′ E, and has an average elevation of 1200 m above sea level. The mean annual temperature for the area is 1.1 °C. The average annual precipitation is 345 mm. The soil is of chestnut type, i.e. Calcic Kastanozems, which is equivalent to Calcic-orthic Aridisol, according to the US soil classification system. The vegetation of the region predominantly comprises grassland plants, i.e. *L. chinensis* (44.5%, relative biomass), *Stipa grandis* Smirn. (34.0%), *Cleistogenes squarrosa* Keng (8.7%), and *Agropyron michnoi* Gaertn (6.8%). Moreover, the *L. chinensis* community represents the most widely distributed grassland community in northern China (Chen and Wang, 2000).

Eight experimental sites were selected and were subjected to SD, MD, LD, GE3, GE8, GE20, GE24, and GE28, respectively (Table 1). The eight experimental sites are all distributed in the same upper basalt platform, and they are contiguous (Fig. 1). Site SD had been exposed to long-term heavy grazing; in fact, an estimated 90% of the aboveground biomass had been consumed by livestock each year. The grasslands were severely degraded as indicated by the extremely sparse vegetation cover (<10%). Site MD had also been exposed to long-term heavy grazing, with an estimated 75% of the aboveground biomass consumed by livestock each year. It was moderately degraded with an existing vegetation cover of 10–25%. Site LD had been subjected to long-term free-grazing; an estimated 65% of the aboveground biomass had been consumed by livestock each year with existing vegetation cover at 25–30%. Site GE3 was established in 2004 by fencing a 40-ha area of a previously free-grazing grassland; similarly, plots GE8, GE20, GE24, and GE28 had been established in 1999, 1987, 1983, and 1979, respectively. Undeniably, there are pseudo-replication issues with only one plot per grazing regime, but this is a common problem with these types of studies. However, it should be certain that changes in SOC and SON across the eight plots in this study are mainly caused by grazing intensity and length of exclusion, because the eight experimental plots are floristically and topographically similar, and all are distributed in the same upper basalt platform (Table 1 and Fig. 1).

2.2. Field sampling and laboratory analysis

In early May 2006, we selected representative plots at the SD, MD, and LD sites; we fenced them to exclude livestock grazing, in order to measure the aboveground and belowground biomass and the C and N content in plants, litter and roots. A field sampling was conducted in mid-August 2006, which is the typical period when aboveground biomass attains its peak value (Jiang, 1985; Bai et al., 2004). In each plot, 10 sampling quadrats (each 1 m × 1 m) were established at 10-m intervals along a random transect. The aboveground biomass in these quadrats was clipped at the ground level, and this quantity was considered approximately equal to the aboveground net primary productivity (ANPP) of the current year. Litter was collected subsequently.

Root biomass was determined using a soil corer (diameter, 7 cm) and 10 sampling points for each site. The samples were separately collected from six layers at the depths of 0–10 cm, 10–20 cm, 20–40 cm, 40–60 cm, 60–80 cm, and 80–100 cm in each sampling point; there were 60 root samplings for each site. Similarly, soil sampling was conducted using a soil sampler (diameter, 4 cm), and, the samples were separately collected from six layers at the depths of 0–10 cm, 10–20 cm, 20–40 cm, 40–60 cm, 60–80 cm, and 80–100 cm in each sampling point; there were 60 soil samplings for each site. Soil bulk density was measured using the soil cores (volume, 100 cm^3) obtained from the six layers, with three replicates for each site; this allowed us to estimate the mass of SOC and SON at each site. The pH of 0–10 cm soil samples, in H_2O (soil:water ratio 1:5), was tested with a PHS-3S pH meter (Sartorius, Germany). For particle-size fraction (i.e. into sand, silt, and clay), 50 g of soil

Table 1
Characteristics of experimental plots

Land-use types	Position	Dominant and sub-dominant species	pH	Sand (%)	Silt (%)	Clay (%)	Grassland Condition	Land-use history
SD	43°24′33″ N 116°38′09″ E	*L. chinensis*, *S. grandis*, *C. squarrosa*, *Artemisia frigida* Willd., *Salsola collina* Pall., *Achnatherum sibiricum* Keng	7.4 ± 0.1^a	639.4 ± 8.1^a	330.6 ± 8.1^a	30.0 ± 1.8^a	Severe degradation	Free-grazing, long-term heavy grazing
MD	43°33′34″ N 116°39′06″ E	*L. chinensis*, *S. grandis*, *C. squarrosa*, *A. frigida*, *Kochia prostrate* Schrad, *A. sibiricum*	7.3 ± 0.1^a	638.3 ± 6.4^a	332.9 ± 6.8^a	28.8 ± 2.0^a	Moderate degradation	Free-grazing, long-term heavy grazing
LD	43°33′47″ N 116°40′13″ E	*L. chinensis*, *S. grandis*, *C. squarrosa*, *A. sibiricum*, *A. frigida*, *S. collina*	7.3 ± 0.1^a	636.8 ± 7.5^a	335.6 ± 9.5^a	27.7 ± 2.3^a	Light degradation	Free-grazing, light grazing
GE3	43°32′39″ N 116°40′00″ E	*L. chinensis*, *S. grandis*, *A. michnoi*, *A. sibiricum*, *C. squarrosa*, *A. frigida*	7.2 ± 0.1^a	636.1 ± 5.3^a	334.9 ± 5.3^a	29.0 ± 1.1^a	Excellent	40-ha plot that has been fenced since 2004
GE8	43°33′00″ N 116°40′20″ E	*L. chinensis*, *S. grandis*, *A. michnoi*, *A. sibiricum*, *C. squarrosa*, *Carex korshinskyi*	7.1 ± 0.1^a	642.2 ± 13.9^a	316.3 ± 13.7^a	41.5 ± 0.4^b	Excellent	35-ha plot that has been fenced since 1999
GE20	43°33′06″ N 116°40′ 08″ E	*L. chinensis*, *S. grandis*, *A. michnoi*, *A. sibiricum*, *C. squarrosa*, *C. korshinskyi*	7.2 ± 0.1^a	664.2 ± 7.1^a	289.5 ± 6.8^b	46.3 ± 0.7^{bc}	Excellent	24-ha plot that has been fenced since 1987
GE24	43°35′50″ N 116°44′19″ E	*L. chinensis*, *S. grandis*, *K. cristata*, *A. sibiricum*, *C. squarrosa*, *C. korshinskyi*	7.2 ± 0.1^a	698.4 ± 9.2^b	253.8 ± 13.4^{bc}	47.8 ± 9.9^{bc}	Excellent	20-ha plot that has been fenced since 1983
GE28	43°33′06″ N 116°40′20″ E	*L. chinensis*, *S. grandis*, *K. cristata*, *A. sibiricum*, *C. squarrosa*, *C. korshinskyi*	7.2 ± 0.1^a	700.4 ± 28.7^b	248.2 ± 30.0^c	51.4 ± 1.7^c	Excellent	24-ha plot that has been fenced since 1979

SD, severe degradation; MD, medium degradation; LD, light degradation; GE3, 3-y GE; GE8, 8-y GE; GE20, 20-y GE; GE24, 24-y GE; GE28, 28-y GE. Values (0–10 cm soil layer) are represented as mean ± SEM ($n = 3$) and designated by the same letters in the same column are not significantly different at $P < 0.05$.

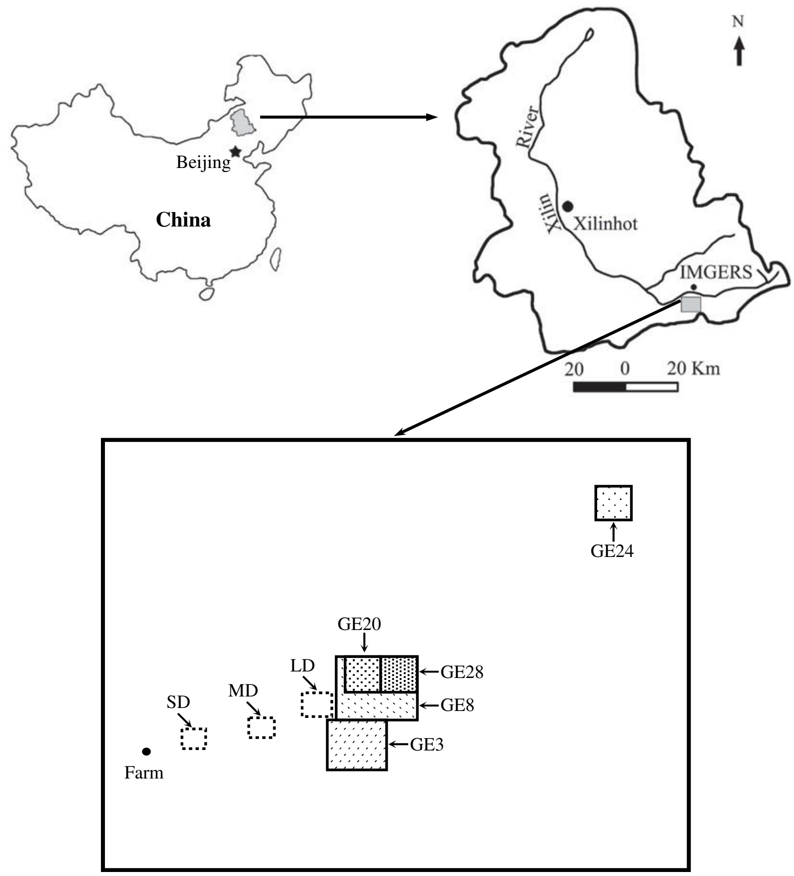

Fig. 1. Eight experimental sites and relative positions. See Table 1 for site abbreviations.

(<2 mm) was dispersed in 250 ml of distilled water with a KS-600 probe-type Ultrasonic Cell Disrupter System (Shanghai Precision & Scientific instrument Co. Ltd., China) set at 360 W; we then detached the different particle-size fractions as per Morra et al. (1991).

The organic C content (%) in the samples of plant, litter, root, and soil was measured using a modified Mebius method (Nelson and Sommers, 1982). Briefly, 0.5-g soil samples were digested with 5 ml of 1 N $K_2Cr_2O_7$ and 10 ml of concentrated H_2SO_4 at 180 °C for 5 min, followed by titration of the digests with standardized $FeSO_4$. The total N (%) of plants, litter, roots, and soil was measured using the modified Kjeldahl wet digestion procedure (Gallaher et al., 1976), using 2300 Kjeltec Analyzer Unit (FOSS, Sweden).

We calculated the total SOC density (TSOC; g C m^{-2}) and total soil nitrogen (TSN, g N m^{-2}) on a ground area basis up to a 100-cm depth as follows:

$$\mathrm{TSOC} = \sum D_i \times P_i \times \mathrm{OM}_i \times S$$

$$\mathrm{TSN} = \sum D_i \times P_i \times \mathrm{TN}_i \times S$$

where D_i, P_i, OM_i, TN_i, and S represent, respectively, the soil thickness (cm), bulk density (g cm^{-3}), organic C concentration (%), total N concentration (%), and cross-sectional area (cm^{-2}) of the ith layer; i = 1, 2, 3, …, and 6.

2.3. Statistical analysis

All data were expressed as mean ± 1 standard error of mean (SEM). The data for the 0–100-cm soil layer were used to analyze the C and N storage potentials of the grassland. An analysis of variance (ANOVA) was used to assess the effect of land-use change on C and N storage. Regression analyses were used to test the relationships between stand age and total C and N. All statistical analyses were performed using the software program SPSS, ver. 10.0 (SPSS Inc., Chicago, IL, USA).

3. Results

ANPP values differed significantly among the eight sites ($P < 0.01$), varying from 35.0 g m^{-2} for plot SD to 146.7 g m^{-2} for plot GE24 (Fig. 2a). The C and N stored in the aboveground biomass were less than 72 g C m^{-2} and 5 g N m^{-2}, respectively (Fig. 3a), accounting for negligible amounts (<1% of the total) of total C and N storage in the ecosystem. The total C storage (including C stored in aboveground biomass, litter, roots, and 0–100-cm soil layers) differed significantly among the eight sites ($P < 0.01$), and varied from 7.0 kg C m^{-2} to 15.8 kg C m^{-2}. The total C storage decreased substantially with grassland degradation, and increased to a significant extent with the introduction of GE (Fig. 2b).

The C storage varied remarkably among the different pools (Fig. 3). The amount of C stored in SOM accounted for over 90% of the total C storage, and the C stored in litter was very low (<1%), compared to other pools. The amount of C stored in the roots varied from 557 g C m^{-2} for plot SD to 879 g C m^{-2} for plot GE8, and it accounted for 8–10% of C storage in the grassland. The C concentration – in both SOM and roots – was far higher in 0–10-cm and 10–20-cm soil layer than in other soil layers (Fig. 5).

The total N storage (including N stored in aboveground biomass, litter, roots, and 0–100-cm soil layers) differed significantly among different grasslands ($P < 0.01$). The total N storage varied from 658.6 g N m^{-2} for plot SD to 1491.4 g N m^{-2} for plot GE8 (Fig. 2c).

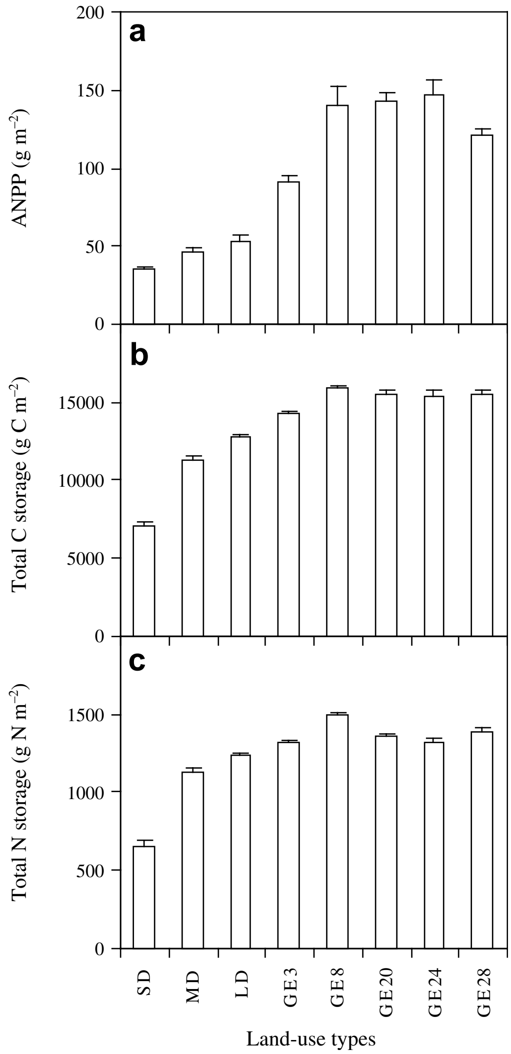

Fig. 2. Changes in ANPP (a), total C storage (b), and total N storage (c) based on different land-use types in the *L. chinensis* grasslands of northern China. Total C and N storage (b and c, respectively) includes that in ANPP, litter, roots, and SOM (i.e. top 100-cm soil layer). Data are represented as mean ± SEM ($n = 10$). See Table 1 for site abbreviations.

Compared to all GE sites, more than 50% of the total N in SD had been lost. The N stored in SOM accounted for 95% of the total soil N storage (Fig. 4d). The N stored in litter and roots was very low, compared to that in SOM. The N concentration – in both SOM and roots – was far higher in the 0–10-cm and 10–20-cm soil layers than in other soil layers (Fig. 6).

Compared to LD, GE8 increased the total C and N storage in the 0–100-cm soil layer by 21.3% and 20.6%, and the annual increase rates were 3.0% and 2.6% for the total C and N storage, respectively. The total C storage increased logarithmically with the duration of GE ($R^2 = 0.77$, $P < 0.05$) (Fig. 7a). The total N storage did not show a similar trend of increase with the duration of GE ($R^2 = 0.28$, NS) (Fig. 7b).

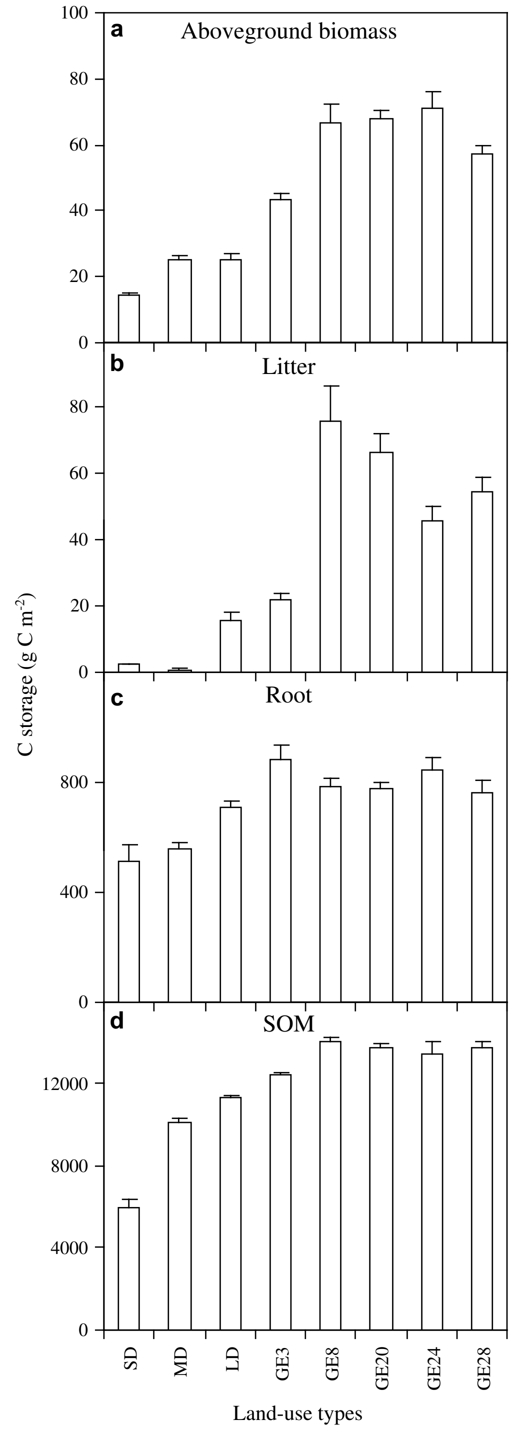

Fig. 3. Carbon storage in aboveground biomass (a), ground litter (b), roots (c), and SOM (i.e. top 100-cm soil layer) (d) based on different land-use types in the *L. chinensis* grasslands of northern China. Data are represented as mean ± SEM ($n = 10$). See Table 1 for site abbreviations.

4. Discussion

The potential of C and N storage in the *L. chinensis* grassland are approximately 15.8 kg C m^{-2} and 1.5 kg N m^{-2}, respectively. The grasslands subjected to GE20 were stable or mature in terms of their productivity (Xiao et al., 1996; Bai et al., 2004), soil respiration (Li et al., 1998, 2002), and C and N storage in this study. Moreover, results from a 4-year study (2003–2006) using the eddy covariance technique in GE28 suggest that *L. chinensis* grasslands after more than 20 years of grazing exclusion are a very weak C source (personal communication). One plausible albeit theoretical explanation of the relatively lower C and N storage value after long-term exclosure (i.e. >20 years) in comparison to relative short-term exclosure (i.e. 8 years) (not significantly different, $P > 0.05$) may be that an increase in ANPP would drive greater competition for all resource-some of them being nutrients and water, and that

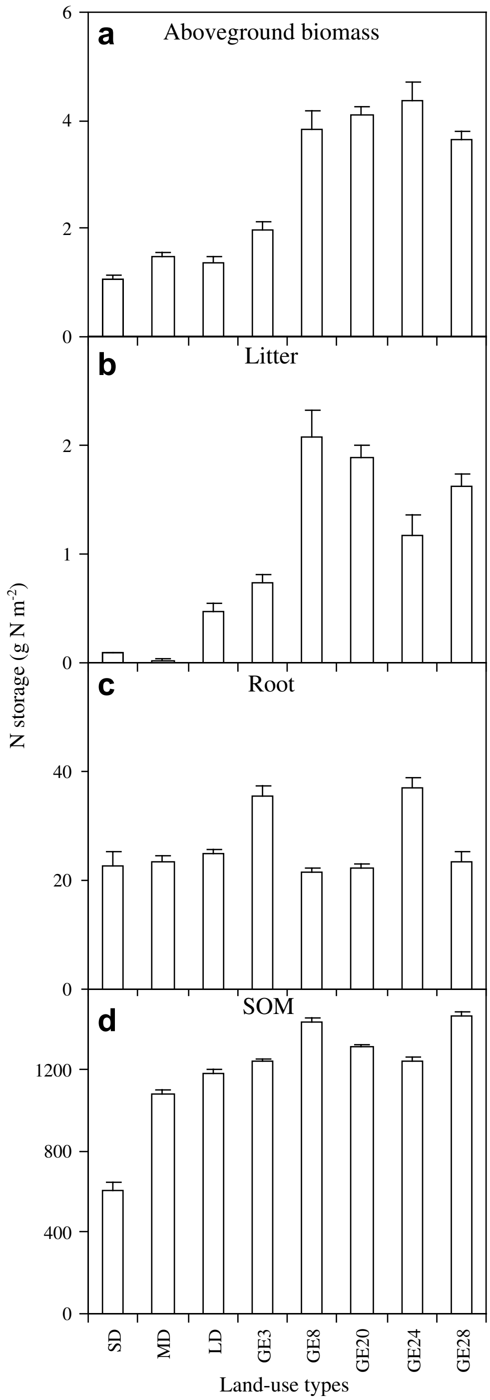

Fig. 4. Nitrogen storage in aboveground biomass (a), ground litter (b), roots (c), and SOM (top 100-cm soil layer) (d) based on different land-use types in the *L. chinensis* grasslands of northern China. Data are represented as mean ± SEM ($n = 10$). See Table 1 for site abbreviations.

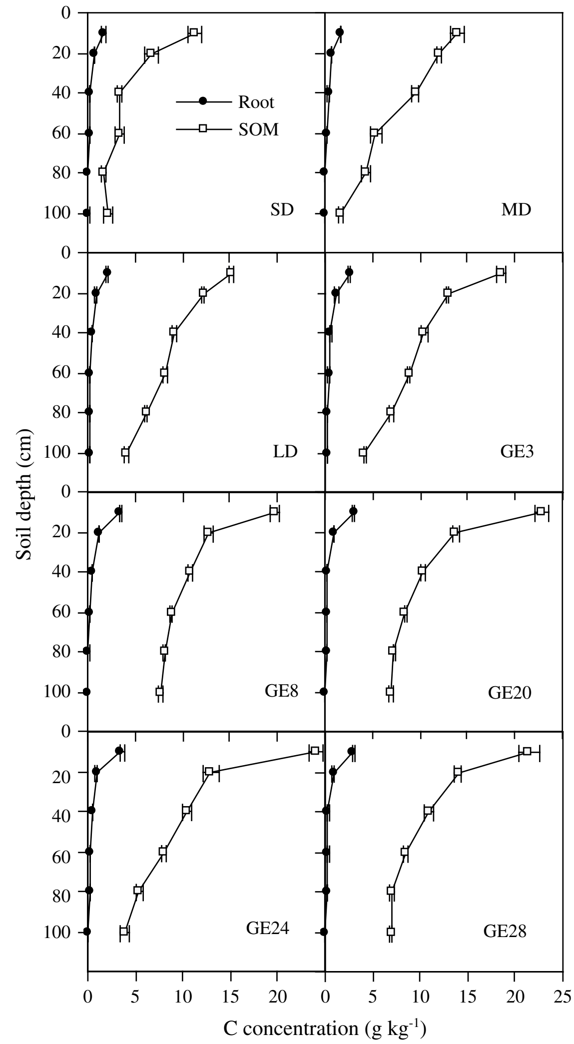

Fig. 5. Carbon concentration and distribution in roots and SOM based on soil depth in the *L. chinensis* grasslands of northern China. Horizontal bars indicate SEM ($n = 10$). See Table 1 for site abbreviations.

increase in nutrient demand would drive greater gross rates of SOM; this, similarly, would likely result in more SOM mineralization when the natural disturbances – including large-animal grazing and fire – are excluded. Another possible explanation is that increasing the accumulation of fresh litter and partially decomposed organic materials keeps precipitation from permeating to the mineral soil, especially during dry years or small rain events; this would result in more precipitation soaking into litter that would, in turn, facilitate organic matter decomposition or soil respiration, but make the precipitation unavailable for plant uptake (Collins and Adams, 1983; Huxman et al., 2004; Schwinning, 2004; Schwinning and Sala, 2004).

The estimates of potential storage capacity place a limit on the C and N storage of *L. chinensis* grasslands in the region, which is based on an assumption that all grasslands eventually reach a stable or mature condition. In fact, grassland utilization (e.g. grazing, mowing, and land-use changes), natural disturbances (e.g. fire and drought), natural conditions (e.g. soil types, stand, and gradient) can, together, create a mosaic of grasslands and exert a significant impact on the C and N storage of grassland ecosystems. Fire and grazing influence the dynamics of C and N in a grassland ecosystem by changing productivity condition, nutrient availability and heterogeneity, soil-water content, and so forth (Hulbert, 1988; David et al., 1991; Collins and Smith, 2006; MacNeil et al., 2008). Despite these caveats, estimating the potential storage capacity helps us to distinguish systematically the effects of different management strategies on the C and N storage of grasslands in northern China.

Carbon stored in SOM varied from 6.5 kg C m^{-2} to 14.9 kg C m^{-2} among the eight *L. chinensis* grasslands in this study, and the value for site LD was 12.0 kg C m^{-2}, which was in agreement with the former estimate of 10–12 kg C m^{-2} for the region (Wu et al., 2003); it was also higher than the global mean value of 10.6 kg C m^{-2} (Post et al., 1982). Moreover, Zhou et al. (2006) reported that the

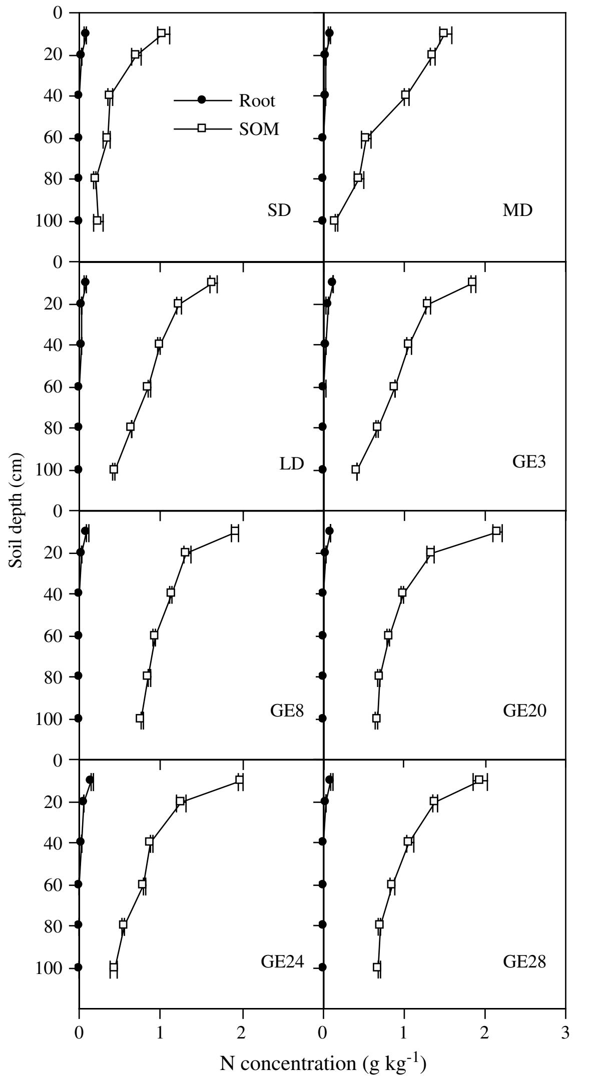

Fig. 6. Nitrogen concentration and distribution in roots and SOM based on soil depth in the *L. chinensis* grasslands of northern China. Horizontal bars indicate SEM ($n = 10$). See Table 1 for site abbreviations.

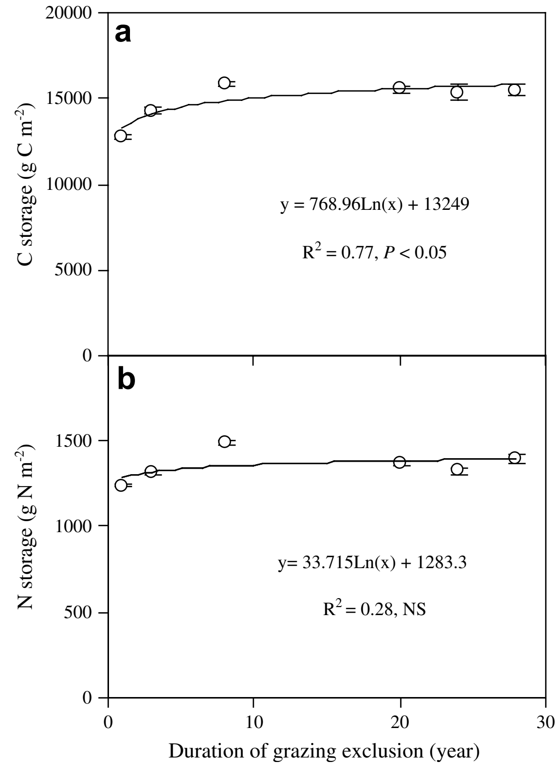

Fig. 7. Changes in total C (a) and N (b) storage with the durations of GE. Horizontal bars indicate SEM ($n = 10$).

C storage in SOM in the agro-pastoral ecotone of Duolun County ranged from 8 kg C m^{-2} to 10 kg C m^{-2}. The discrepancy can be explained by differences in vegetation and soil types, because the productivity of the *Stipa krylovii* steppe in Duolun County is generally low, compared to the *L. chinensis* grasslands (Li et al., 1998). Moreover, the total soil C content is, to a certain extent, dependent on soil type but only in part on the type of land-use (Tan et al., 2004; Breuer et al., 2006).

A two-decade GE would restore the *L. chinensis* grassland from a state of light degradation to a stable condition in terms of productivity, SOC, and N storage. Werth et al. (2005) have suggested that the SOM reserves in grassland ecosystems in southwest Germany are expected to remain stable for two decades. Burke et al. (1995) have demonstrated that a 50-year period is adequate for the recovery of active SOM and nutrient availability, but the recovery of total SOM pools is a considerably slower process; therefore, we suggest that a GE of at least two decades' duration would be appropriate for restoring the *L. chinensis* grasslands from a state of light degradation to state similar in productivity, SOC, and SON storage to undisturbed natural grassland. However, the time required to restore degraded grasslands to their original steady states could be much longer, if the integrity of biological interactions and SOM quality – rather than quantity along – are to be considered.

The net effect of grazing on SOM depends on grazing intensity and the community types considered. In general, light and moderate grazing will negligibly affect or even benefit grassland ecosystems in terms of dry matter production, nutrient-cycling and C and N storage, possibly due to increased nutrient availability and facilitated vegetation regeneration (McNaughton, 1985; Frank and McNaughton, 1993; Han et al., 2008). For example, short-grass prairie in the Great Plains of North America tended to increase SOC by 24% in plots under light and moderate grazing than in plots without grazing, while mid-grass and tall-grass prairies showed an opposite trend (i.e. 8% lower) (Derner et al., 2006). Another study by Schuman et al. (1999) also revealed that the total C and N pools in the top 60 cm of soil were slightly affected by light and heavy grazing; The storage of soil C and N, however, declines significantly in the presence of long-term heavy grazing (Cui et al., 2005; Elmore and Asner, 2006; Han et al., 2008; Steffens et al., 2008). Some of the mechanisms at work include the fact that (1) light grazing increases nutrient availability and facilitates both vegetation regeneration and C and N storage (Habbs et al., 1991; Derner et al., 1997); (2) biomass removal by heavy grazing significantly decreases the input of organic matter from aboveground biomass and roots (Johnson and Matchett, 2001); (3) heavy grazing may decrease biomass productivity, due to decrease soil infiltrability and nutrient availability (Savadogo et al., 2007); and (4) the disruption of the soil's aggregate structure and the crust of surface soil by livestock stamping will enhance SOM decomposition and render the soil susceptible to water and wind erosion (Hiernaux et al., 1999; Belnap, 2003; Liu et al., 2003; Neff et al., 2005).

Compared to LD, GE8 can increase C and N storage in the 0–100 cm soil layer by 21.3% and 20.6%, and the annual increase rates are 3.0% and 2.6% for C and N, respectively. Conant et al. (2001) reported by meta-analysis that changes in grazing management and fertilization can led to annual increases of 2.9% and 2.2% in C and N storage, respectively. These increases are mainly restricted to the upper soil depth (Derner et al., 1997). The soil C and N storage sustained an initial rapid increase with the introduction of GE, followed by a steady phase of C and N storage with time; the rapid increase of C and N in the former phase may be partially explained by increases in root production and turnover. Guo et al. (2007) estimate that, in a native pasture, the annual inputs of fine root litter to the top 100 cm soil are 3.6 Mg C ha^{-1} and 81.4 kg N ha^{-1}. Moreover, Johnson and Matchett (2001) demonstrate that grazing diminishes root growth, especially in heavily grazed patches (~30% less than in fenced controls); this is, because grazing increases N cycling and availability so that grazed plants experience C limitation as shoots regrow and plants allocate less C to roots. In temperate grasslands, considerable loss of C and N is caused by wind and water erosion, particularly in areas with sandy soil and high wind velocity (Hiernaux et al., 1999; Liu and Tong, 2003; Liu et al., 2003). Hu et al. (2005) report that rates of wind erosion in lightly degraded and heavily degraded grasslands, in southern Inner Mongolia, were 1808 and 4270 g m^{-2} a^{-1}, respectively, and illustrate that a decrease in vegetation cover due to long-term heavy grazing accelerates wind erosion. Moreover, related research has demonstrated that dust deposition rates on ungrazed *L. chinensis* grassland in the region averaged 2.4 g m^{-2} d^{-1}, which partially explained the rapid increase of C and N with the introduction of GE (Hoffmann et al., 2008). Therefore, we should be careful to explain rapid increases in C and N, as the effect of GE on soil C and N storage could otherwise be overestimated.

Free-grazing has been an extensively employed land-use practice in the temperate grasslands of northern China. Long-term overgrazing has led to a decline in grassland productivity and caused grassland degradation and soil loss in vast areas (Dong and Zhang, 2005). Soil C and N storage in the *L. chinensis* grasslands in 1980 were 13.6 kg C m^{-2} and 1.3 kg N m^{-2}, respectively (Wang and Cai, 1988); in our study, compared to those data, the soil C storage had decreased by 52.2%, 21.2%, and 11.6% for the SD, MD, and LD plots, respectively; and the soil N storage had decreased by 51.2%, 15.3%, and 6.6%, respectively. Therefore, a large amount of C and N has been lost in the last two decades across grasslands subjected to long-term heavy grazing; some of this C and N has been emitted into the atmosphere, but most has been lost to wind erosion (Hu et al., 2005). In total, the degradation of temperate grasslands due to long-term heavy grazing has reversed the sequestration potential and led to C and N loss by erosion and oxidation, instead of the C sequestration that was desirable in the past two decades.

The deteriorating environmental conditions have now prompted the local and autonomous Inner Mongolia government to officially restrict or ban livestock grazing in the region, from the year 2000 onwards. As a consequence, GE and mowing have become the encouraged land-use types for grasslands. Different management practices that enhance or weaken C storage in the grasslands of the region have significant implications on the global C budget, because the grasslands of northern China comprise a significant portion of the Eurasian continent (Ojima et al., 1993). With regard to C storage, the grasslands with a higher potential for C sequestration are those that have been depleted by poor management strategies in the past (Jones and Donnelly, 2004). Based on our results, we conclude that the temperate grasslands of northern China exhibit tremendous potential for increasing their C storage.

5. Conclusion

Land-use change has significant effects on C and N storage in the grasslands of northern China. C and N storage has decreased substantially on account of grassland degradation due to long-term heavy grazing; GE has increased significantly the storage of C and N in these grasslands. Our results showed that a GE period of at least two decades would be appropriate for restoring the *L. chinensis* grasslands from a state of light degradation to one having productivity, SOC, and N storage similar to undisturbed natural grassland. The storage potential of C and N in the *L. chinensis* grasslands are approximately 15.8 kg C m^{-2} and 1.5 kg N m^{-2}, respectively; therefore, there is tremendous potential for increasing C storage in the temperate grasslands of northern China by improving grassland use or management.

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