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Grazing intensity impacts soil carbon and nitrogen storage of continental steppe

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Abstract. Recent studies have underscored the importance of grasslands as potential carbon (C) sinks. We performed a grazing experiment with seven stocking rates (SR0, SR1.5, SR3.0, SR4.5, SR6.0, SR7.5, and SR9.0 for 0, 1.5, 3.0, 4.5, 6.0, 7.5, and 9.0 sheep ha^{-1}, respectively) to investigate the effect of increasing grazing pressure on soil C and nitrogen (N) storage in the temperate grasslands of northern China. The results revealed that C and N storage in both 0–10 cm and 10–30 cm soil layers decreased linearly with increasing stocking rates. Carbon storage in the 0–10 cm soil layer was significantly higher in lightly grazed grasslands than in heavily grazed grasslands after a 5-yr grazing treatment. Our findings suggest an underlying transformation from soil C sequestration under light grazing to C loss under heavy grazing, and that the threshold for this transformation is 4.5 sheep ha^{-1} (grazing period from June to September). Results confirmed that grasslands used for grazing in northern China have the capacity to sequester C in the soil under appropriate grazing pressure, but that they lose C under heavy grazing. Therefore, appropriate grazer densities will promote soil C sequestration in the grasslands of northern China.

Key words: carbon; carbon sequestration; carbon storage; grassland; grazing; nitrogen; soil fraction.

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Introduction

Improving our understanding of the effects of land use changes on soil carbon (C) storage or sequestration in terrestrial ecosystems has become a recent topic of interest to researchers (IPCC 2007). Compared with croplands and improved pastures, natural rangelands exhibit relatively low soil C sequestration per unit area (Post and Kwon 2000, Conant et al. 2001, Jones and Donnelly 2004). However, the total amount of C sequestered in natural rangelands can be enormous because these lands occupy half of the earth's land area and contain approximately one-third of the global above- and belowground storage of C (Derner and Schuman 2007, Lal 2009).

Across rangelands, the effects of livestock grazing on soil C storage are variable and inconsistent; depending on the system, these herbivores may facilitate or depress C accretion rates (Milchunas and Lauenroth 1993, Schuman et al. 1999, Liebig et al. 2006, Derner and Schuman 2007, Ingram et al. 2008). The different effects of grazing on soil C storage or sequestration may reflect variations in climate, soil, landscape location, plant community type, and grazing management practices (Milchunas and

Lauenroth 1993, Reeder and Schuman 2002). Moreover, changes in soil C levels over time during biotic community development may be strongly linked with soil N levels (Knops and Tilman 2000). Thus, the influence of grazing on soil C storage in grasslands varies by region.

Researchers studying C storage or sequestration in soil have attempted to identify the fractions of soil organic matter (SOM) that respond more rapidly to land use changes than bulk SOM. These fractions could then serve as early indicators for the overall stock change (Christensen 2001, Olk and Gregorich 2006). Because the particle size fractions (sand, silt, and clay) of C pools are considered to be important factors that control SOM turnover, they are analyzed to evaluate the dynamics and turnover of SOM under various land use practices and climates (Leifeld and Kögel-Knabner 2005, Zinn et al. 2007, He et al. 2009).

Temperate grasslands in northern China cover approximately 110×10^6 ha. Because of their obvious ecological and economic importance, researchers have attempted to quantify their C storage and sequestration (Ni 2002, Fan et al. 2008, Lal 2009). Following the rapid expansion of the livestock industry after 1980, most temperate grasslands in China have undergone some degree of degradation or desertification (Tong et al. 2003). Measures to encourage grassland restoration were implemented in 2000 and are anticipated to increase soil C and N storage in northern Chinese grasslands. However, very few studies have addressed the effects of increasing stocking rates on soil C storage in temperate grasslands, even though such results are important for ecologists and grassland management decision-makers.

In the present study, we conducted a grazing experiment with seven sheep stocking rates in a temperate grassland of northern China to (1) evaluate the influence of different stocking rates on soil C and N storage, (2) explore the influence of grazing on the distribution of C and N in soil fractions, and (3) test the hypothesis that there exists an underlying transformation from C sequestration to C loss with an increase in stocking rates.

Materials and Methods

Study area

Field work was conducted in a typical steppe ecosystem on the Mongolian plateau in northern China (43°33′ N, 116°40′ E) which is administered by the Inner Mongolia Grassland Ecosystem Research Station (IMGERS) of the Chinese Academy of Sciences. The climate is typical of a continental, semiarid climate. Mean annual temperature and precipitation (1982–2009) were 1°C and 334 mm, respectively. Soil is of the chestnut type, i.e., Calcic Kastanozems, equivalent to Calcic-orthic Aridisols in the U.S. soil classification system. The dominant vegetation consists of grassland plants, i.e., perennial rhizome grass (*Leymus chinensis* (Trin.) Tzvel) and perennial bunchgrass (*Stipa grandis* Smirn. and *Cleistogenes squarrosa* (Trin.) Keng) (Chen and Wang 2000).

Grazing experiment and sampling plots

Plots were established on grassland dominated by *L. chinensis* and *S. grandis* prior to the initiation of our experiment in 2004. The experimental area was historically used for raising sheep and goats, and grazing pressure was moderate. Considering the heterogeneity of the grassland, the study area was divided into two blocks; one block was situated on a flat area and the other block was situated on a moderately sloping area (Fig. 1). Seven sheep stocking rates, 0, 1.5, 3.0, 4.5, 6.0, 7.5, and 9.0 sheep ha^{-1} (hereafter designated SR0, SR1.5, SR3.0, SR4.5, SR6.0, SR7.5, and SR9.0, respectively), were set up in each block. The area of each fenced plot was 2 ha except for treatment SR1.5, in which the plot size was 4 ha. Beginning in 2005, sheep were transferred to plots in mid-June and were maintained there until mid-September. The same stocking treatments were applied to both blocks. The basic design of the grazing experiment is outlined in earlier works (Glindemann et al. 2009, Schönach et al. 2009). We also selected two grazing-free grasslands, designated as CK1 and CK2, outside of the fenced area of each block. In total, 16 sampling plots (14 grazing plots and two grazing-free grasslands) were established (Fig. 1).

Field sampling

At the end of the 5-yr grazing experiment (late September 2009), we established five sampling

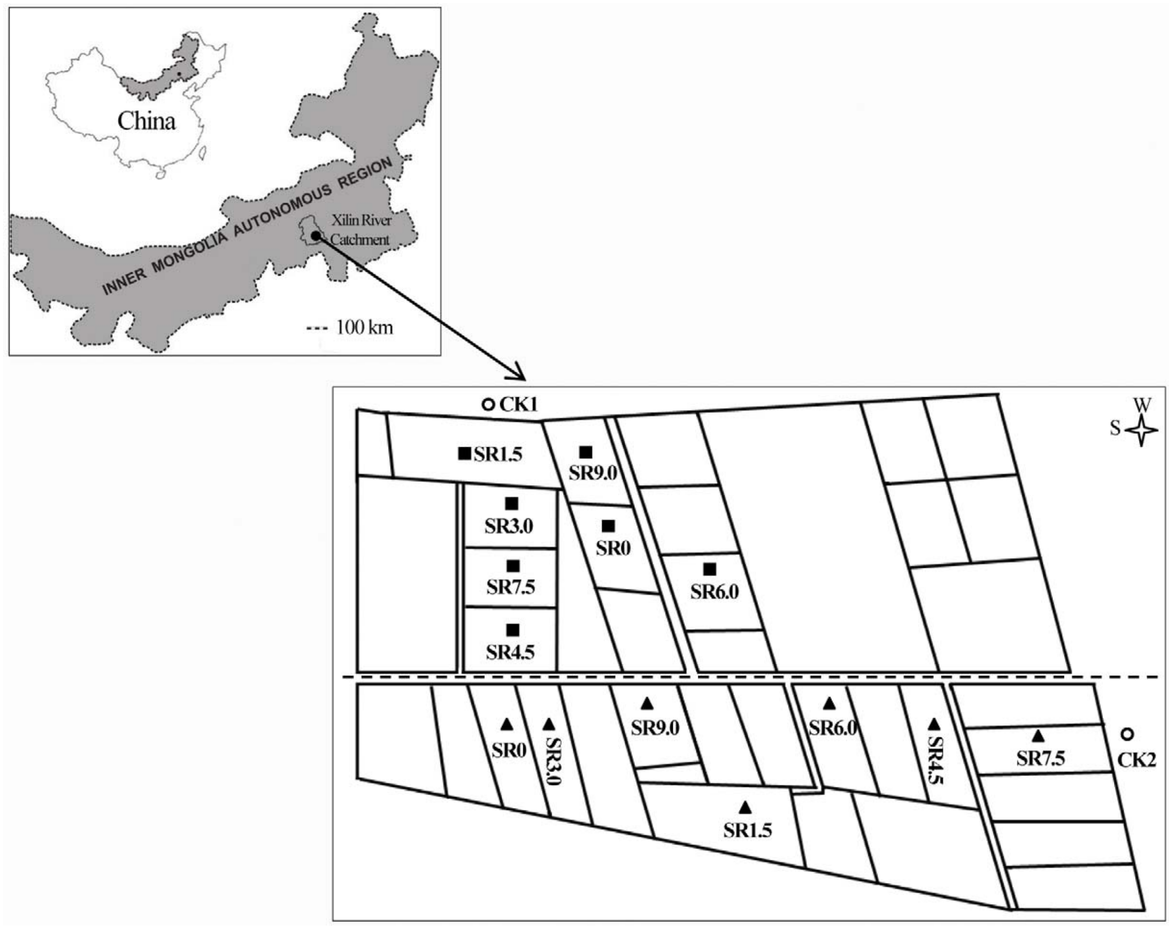

Fig. 1. Experimental sites and their relative positions. The experimental area was divided into two blocks. In each block, we set up seven stocking rates (namely, 0, 1.5, 3.0, 4.5, 6.0, 7.5, and 9.0 sheep / ha), designated as SR0, SR1.5, SR3.0, SR4.5, SR6.0, SR7.5, and SR9.0, respectively. In addition, we set up grazing-free plots outside of each fenced block as controls (CK1 and CK2).

points (a center point and four points each approximately 15 m from the corners) in each of the 16 plots. We established 1 quadrat (1 m × 1 m) at each sampling point and investigated plant community cover and height. Subsequently, aboveground biomass, with all of the plant species combined, was clipped at ground level.

Within each quadrat, three soil cores (2 m apart) were collected and combined from two layers at depths of 0–10 cm and 10–30 cm. A total of 10 soil samples were collected from each experimental plot. We measured soil bulk density at a depth of 0–30 cm at each point using the core method (100 cm^3 volume) (Blake and Hartage 1986); this allowed us to calculate the mass of C and N at each site.

Particle size fractionation and chemical analysis

We fractionated the soil samples into sand (50–2000 μm), silt (2–50 μm), and clay (<2 μm) fractions using ultrasonic energy to disrupt aggregates, following the methods of Roscoe et al. (2000). After manually removing visible root remnants, 50 g of soil (particles <2 mm) was dispersed in 250 ml of distilled water using a KS-600 probe-type ultrasonic cell disrupter system (Shanghai Precision and Scientific Instrument, Shanghai, China) operating for 32 min in continuous mode at 360 W. Under these conditions, the real power input was 56.02 W and the value of applied energy was 430 J ml^{-1} suspension, as calculated on the basis of equations from Roscoe et al. (2000). Sand (50–2000 μm) and coarse silt (20–50 μm) were separated by wet sieving. To further separate fine silt (2–20 μm) and clay (<2 μm), the samples were centrifuged repeatedly at 150 × g for 5 min. The supernatants were collected in 250-ml centrifuge bottles and centrifuged at 3900 × g for 30 min; the precipitated fraction was referred to as clay. All of the fractions were dried at 50°C and ground for further chemical analysis.

Organic C content (%) of the samples was measured using the modified Mebius method (Nelson and Sommers 1982). For this procedure, 0.5 g of soil sample was digested with 5 ml of 1 N $K_2Cr_2O_7$ and 5 ml of concentrated H_2SO_4 at 180°C for 5 min, followed by titration of the digests with standardized $FeSO_4$. Total soil N (%) was analyzed using the modified Kjeldahl wet digestion procedure (Gallaher et al. 1976) and a 2300 Kjeltec Analyzer Unit (FOSS, Höganäs, Sweden).

Calculations

Soil organic C (SOC, Mg C ha^{-1}) and total soil N (TSN, Mg N ha^{-1}) were calculated on an area basis to a soil depth of 30 cm as follows:

$$\mathrm{SOC} = \sum D_i \times S \times B_i \times OM_i \div 100$$

$$\mathrm{TSN} = \sum D_i \times S_i \times B_i \times TN_i \div 100$$

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

Similarly, C and N storage in soil fractions (sand, silt, and clay) (Mg C ha^{-1} and Mg N ha^{-1}) was calculated as follows:

$$\mathrm{C_{storage}}(\mathrm{fraction_i}) = C_{con.}(fraction_i) \times F \times D \times S \times B \div 10^5$$

$$\mathrm{N_{storage}}(\mathrm{fraction_i}) = N_{con.}(fraction_i) \times F \times D \times S \times B \div 10^5$$

where $C_{con.}(fraction_i)$ is the C content of the soil fraction (%), $N_{con.}(fraction_i)$ is the N content of the soil fraction (%), and F is the fraction content in soil (g fraction kg^{-1} soil).

Statistical analysis

Normality and homogeneity of variances were verified for all data using Kolmogorov–Smirnov and Levene tests, respectively. Then we used a two-way ANOVA to compare all means between the two blocks and seven sheep stocking rates. We found no significant interactions between stocking rates and topography (blocks). Therefore, the two blocks were treated directly as replicates, as we had anticipated initially. Then, a one-way ANOVA (with Duncan's test as the post-hoc test for multiple comparisons) was used to evaluate the effect of stocking rates on soil C and N. Regression analysis was used to explore the changing trends of soil C and N storage with increasing stocking rates. Data were represented as mean ± 1 SD (n = 10). All analyses were conducted using SPSS statistical software (ver. 11.0, SPSS, Chicago, IL, USA).

Results

Changes in aboveground plant biomass and soil bulk density

Aboveground plant biomass varied significantly from 163.5 ± 5.7 g m^{-2} in plot SR0 to 55.4 ± 9.3 g m^{-2} in plot SR9.0 ($F = 94.72$, $P < 0.01$), and decreased logarithmically with increasing stocking rates (Fig. 2A). Soil bulk density was significantly different among different stocking rates in the 0–10 cm soil layer ($F = 3.13$, $P < 0.01$), and the values were significantly lower in plots SR0 and SR1.5 than in plots SR6.0, SR7.5, and SR9.0 (Duncan multiple comparisons). Moreover, bulk density increased linearly with increasing stocking rates in both 0–10 cm and 10–30 cm soil layers (Fig. 2B).

Changes in soil C and N storage

The results revealed that, after the 5-yr grazing treatment, C storage in the 0–10 cm soil layer was significantly different among stocking rates ($F = 3.37$, $P < 0.01$), and C storage was higher in plots SR0 and SR1.5 than in plots SR7.5 and SR9.0. The influence of grazing on C storage was relatively small in the 10–30 cm and 0–30 cm soil layers. There were general decreasing trends for C storage with increasing stocking rates in the 0–10 cm, 10–30 cm, and 0–30 cm soil layers, which can be well simulated by linear equations (Fig. 3A).

Nitrogen storage was not significantly different in either the 0–10 cm or 10–30 cm soil layers

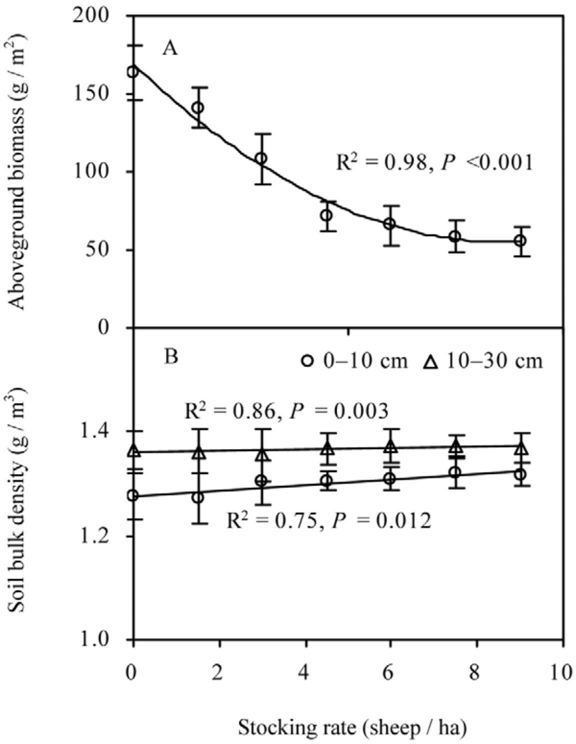

Fig. 2. Changes in aboveground plant biomass and soil bulk density with increasing stocking rates. For aboveground plant biomass, a logarithmic regression was used to determine the underlying relationship (A); bulk density increased linearly with increasing stocking rates (B).

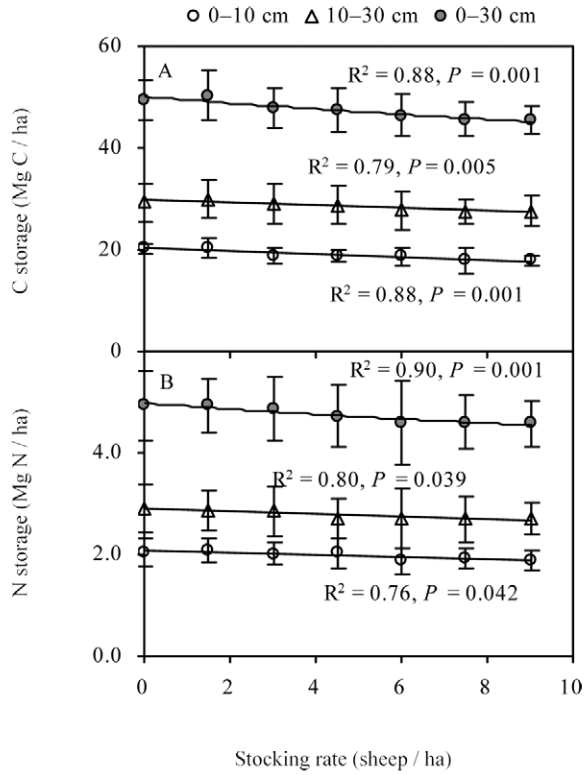

Fig. 3. Changes in soil C (A) and N (B) storage under different grazing rates. Data represented as means ± 1 SD (n = 10). For all corresponding variables, linear regression was used to determine the underlying relationships.

among various grazing levels, although the trend decreased linearly with increasing stocking rates (Fig. 3B). Moreover, the results revealed that the storage of C and N in the surface soil increased logarithmically with increasing aboveground plant biomass (Fig. 4).

Changes in C and N storage of soil fractions

Sand dominated the particle-size distribution across the 14 plots and comprised 67.1–70.6% of total soil weight in the 0–10 cm soil layer; silt comprised 26.5–29.8% of the total soil weight, and the clay content was relatively low (Fig. 5). Sand fractions increased significantly with increasing stocking rates ($F = 4.28$, $P < 0.01$ in the 0–10 cm soil layer; $F = 3.29$, $P < 0.01$ in the 10–30 cm soil layer), and the increasing trends can be well depicted quadratically (Fig. 5). Conversely, the silt fraction was decreased with increasing stocking rates. The influence of grazing on the clay fraction was small.

Carbon storage in the sand fraction (50–2000 μm) of the 0–10 cm soil layer was significantly different among stocking rates ($F = 3.36$, $P < 0.01$), and decreased quadratically with increasing stocking rates (Fig. 6A). Carbon storage in the silt and clay fractions in the 0–10 cm soil layer decreased with increasing stocking rates and showed strong quadratic relationships with stocking rates. Similarly, C stored in sand, silt, and clay in the 10–30 cm soil layer was not significantly different among grazing treatments, but showed quadratic relationships with increasing stocking rates (Fig. 6C).

Nitrogen storage in sand (50–2000 μm), silt (2–50 μm), and clay (<2 μm) in the 0–10 cm and 10–30 cm soil layers was not significantly different among stocking rates. However, N storage in

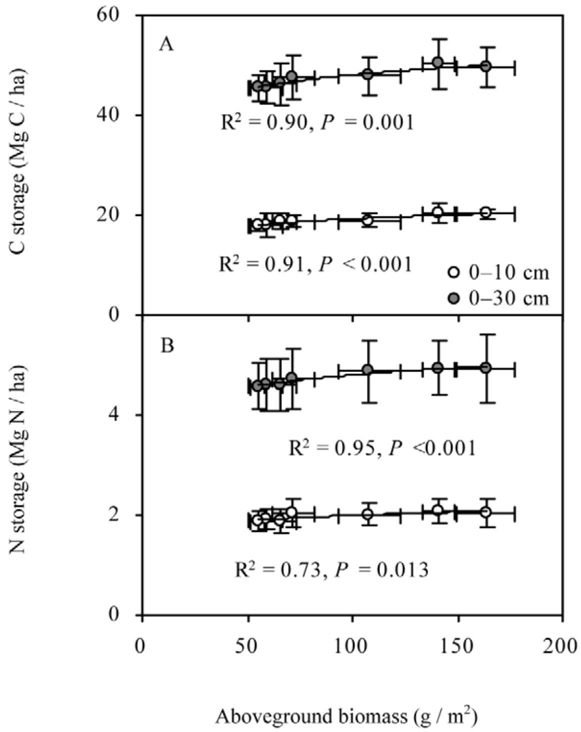

Fig. 4. Relationship between soil C and N storage with aboveground biomass in grazing grasslands. For all corresponding variables, logarithmic regression was used to determine the underlying relationships.

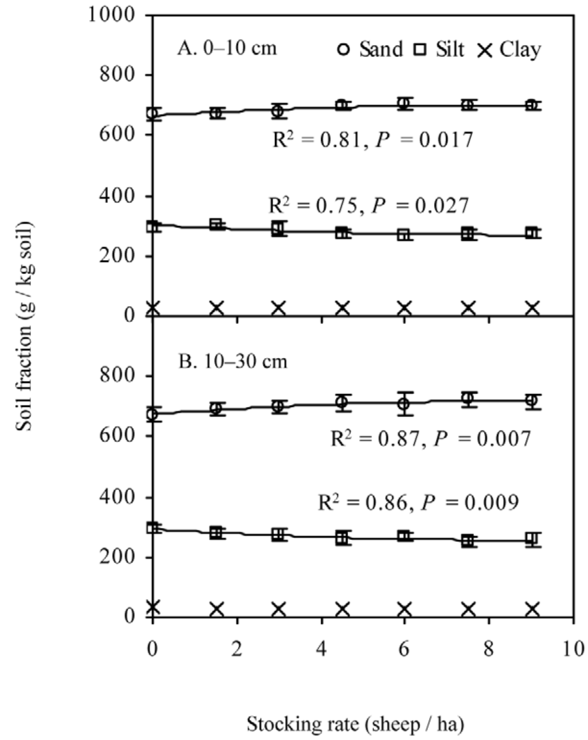

Fig. 5. Changes in the distribution of soil fractions with increasing stocking rates. For all corresponding variables, quadratic regression was used to determine the underlying relationships.

sand and silt in both the 0–10 cm and 10–30 cm soil layers decreased linearly with increasing stocking rates (Fig. 6B and D).

Discussion

Influence of grazing on soil C and N storage

In general, C and N storage in both the 0–10 cm and 10–30 cm soil layers decreased with increasing stocking rates in the Inner Mongolian grasslands. Wu et al. (2008) found that soil C and N storage (0–40 cm) increased logarithmically with the duration of grazing exclusion in Inner Mongolian grasslands. In the current study, soil C and N storage were slightly higher in plot SR1.5 compared to plot SR0 (Fig. 3). Possible explanations for soil C storage enhancement with light grazing include increases in production, elevated nutrient availabilities, and facilitation of vegetation regeneration (Frank and McNaughton 1993, Milchunas and Lauenroth 1993, Han et al. 2008). Another possible explanation is gains from dust deposition, which contribute considerably to increases in soil C and N. Hoffmann et al. (2008) estimated that the net deposition of C and N from dust in lightly grazed sites in the region reached 10.9 g C m^{-2} yr^{-1} and 1.0 g N m^{-2} yr^{-1} in 2005 and 2006; these rates were influenced by variations in vegetation height and coverage as a result of grazing activity. Our findings also suggest that the apparent increase in C and N storage in the sand and silt fractions of lightly grazed sites represented an important contribution to the new C and N accumulation of the entire soil. Moreover, the increases of C storage in silt and clay (<50 μm) also indicated that light grazing favors the accumulation of stable SOM (Christensen 2001).

Carbon and N storage declined in the heavily grazed grasslands, and soil acted as a C source. Declines in soil C and N storage under long-term heavy grazing have been reported previously (Cui et al. 2005, Elmore and Asner 2006, Han et

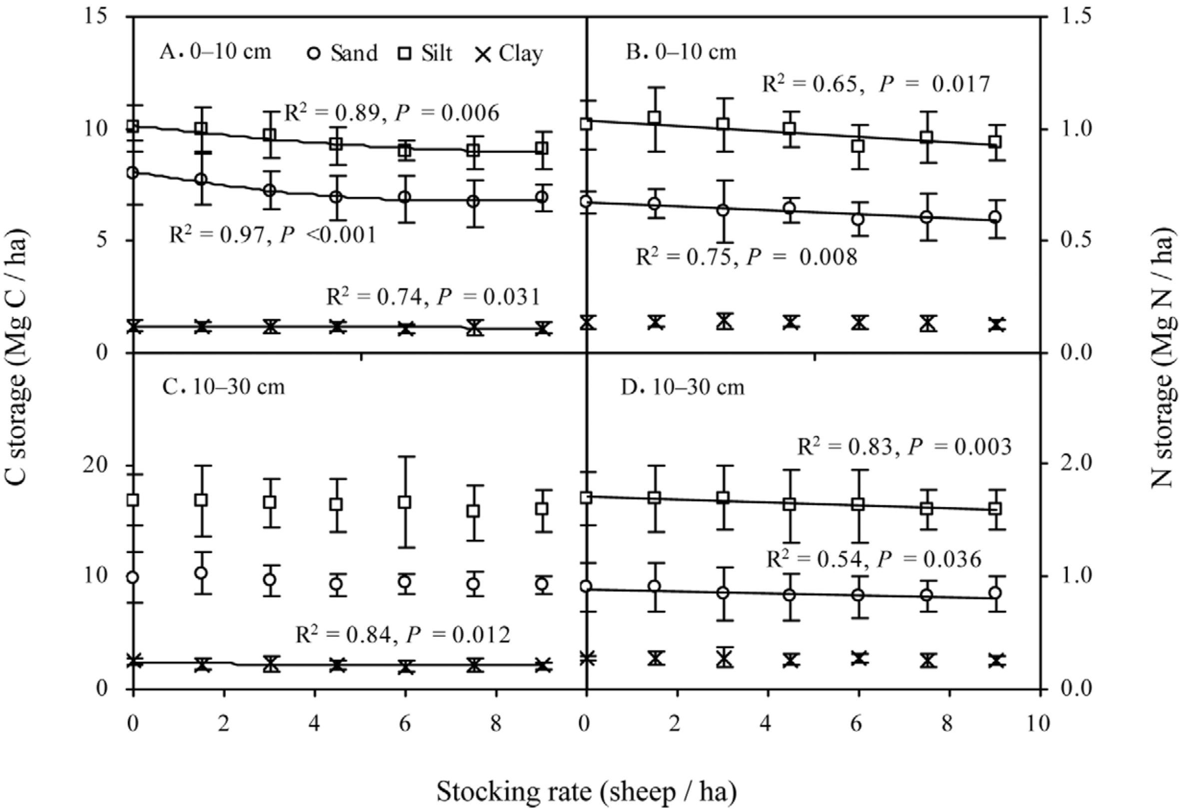

Fig. 6. Changes in C and N storage in soil fractions with increasing stocking rates. For C storage (A and C) and N storage, quadratic and linear regressions were used to determine the underlying relationships, respectively.

al. 2008, Steffens et al. 2008). Ingram et al. (2008) reported that heavy grazing resulted in a 30% loss in soil C storage (0–60 cm) in a mixed-grass ecosystem; losses were attributed mainly to plant community changes and the resultant accumulation of SOC closer to the soil surface, making it more vulnerable to loss. Several mechanisms have been proposed to explain decreases in soil C and N storage: (1) biomass removal by heavy grazing significantly decreases the input of organic matter from aboveground biomass and roots (Johnson and Matchett 2001), (2) heavy grazing may decrease productivity due to decreases in soil infiltrability and nutrient availability (Savadogo et al. 2007), and (3) disruption of the structure of soil aggregates and surface crust by livestock trampling enhances SOM decomposition and renders soil susceptible to water and wind erosion (Neff et al. 2005). Hoffmann et al. (2008) estimated average soil organic C and N losses in heavy grazing sites of 4.73 g C m^{-2} yr^{-1} and 0.44 g N m^{-2} yr^{-1} in the spring of 2005 and 2006. In contrast, some studies have reported that soil C storage is higher in heavy grazing sites, mainly because of increased root production in the surface soil that accompanies changes in species composition (Frank et al. 1995, Reeder and Schuman 2002, Liebig et al. 2006).

C and N sequestration under different stocking rates

We calculated the C and N sequestration rates in grassland soils under different stocking rates based on the differences between these grazing plots and CK plots (grazing-free grasslands). The results revealed that C sequestration rates (0–30 cm) were 59.6, 74.8, and 27.5 g C m^{-2} yr^{-1} in plots SR0, SR1.5, and SR3, respectively. In contrast, grasslands exhibited C and N loss under heavy grazing pressure. Overall, C and

N sequestration in the total soil and the silt plus clay fractions exhibited strong quadratic decreases with increasing stocking rates (data not shown). Our findings indicated that there was an underlying transformation from soil C and N sequestration under low grazing to C and N loss under heavy grazing, and the threshold for the transformation was 4.5 sheep ha^{-1}. As reported by Lal (2009), C sequestration rates ranged from 0–200 g C m^{-2} yr^{-1} in semiarid regions. Australian pastures can sequester 50–60 g C m^{-2} yr^{-1} (Gifford et al. 1992). By improving grassland management, soil C sequestration reached 59 g C m^{-2} yr^{-1} in North American study sites and 28 g C m^{-2} yr^{-1} in Australia (Conant et al. 2001). Conant et al. (2003) demonstrated that the soil C sequestration rate averaged 41 g C m^{-2} yr^{-1} in four managed intensive grazing sites in the southeastern U.S. Differences in the sampling depths and inadequate evaluation of C distribution in grazing ecosystems may have contributed to inconsistencies among the results (Schuman et al. 1999).

In general, we expected grazing exclusion to be a practical and important approach for achieving the soil C sequestration potential of temperate grasslands in northern China. Compared to the grazing-free grasslands (CK plots), grazing exclusion (SR0) annually enhanced C and N storage (0–30 cm) by 1.1% and 0.9%, respectively. In a meta-analysis by Conant et al. (2001), changes in grazing management and fertilization were demonstrated to lead to annual increases of 2.9% and 2.2% in C and N storage, respectively. Our results confirm the literature reports that C and N storage undergoes an initial rapid increase with the introduction of grazing exclusion in *L. chinensis* grasslands in northern China, where annual increase rates are 3.0% and 2.6% for C and N storage, respectively (He et al. 2008). Moreover, grazing exclusion enhances soil C storage in sand grassland in the Horqin region (Su et al. 2005) and in grassland of the agropastoral ecotone in Duolun County, China (Zhou et al. 2007). These increases were restricted mainly to the upper soil layer and were logarithmic throughout the duration of grazing exclusion (Wu et al. 2008). As mentioned in the first section of the discussion, dust deposition can partly contribute to this rapid increase (Hoffmann 2008). It is therefore certain that grazing exclusion can enhance soil C and N storage in temperate grasslands in northern China.

Trading C credits in the future opens new opportunities for promoting the use of terrestrial C sinks. The use of C sequestration programs would be of particular benefit to degraded or desertified grassland ecosystems in Asia, because the rehabilitation of these degraded lands is an urgent concern of global importance (Lal 2009). On the basis of our results, temperate grasslands in northern China have tremendous potential for increasing their C storage under low to moderate stocking rates. We also demonstrated that inappropriately heavy grazing would degrade soil C storage. Fortunately, deteriorating environmental conditions recently prompted the autonomous Inner Mongolian government to officially restrict or ban livestock grazing in the region after the year 2000. Therefore, an increase in soil C and N storage is anticipated in the grasslands of northern China as a result of the implementation of these measures aimed at encouraging grassland restoration.

In summary, soil C and N storage decreased with increasing stocking rates in Inner Mongolian grasslands. Compared with grazing-free grasslands, lightly grazing grasslands showed an apparent capacity to sequester C and N in soil, but heavily grazed grasslands exhibited a C and N loss. The grazing grasslands in northern China have the capacity to sequester C in soil under appropriate grazing pressure, but they exhibit C loss under heavy grazing. Our findings indicate that there exist a system transformation from soil C sequestration under low grazing to C and N loss under heavy grazing, and that the threshold for this transformation was 4.5 sheep ha^{-1} (grazing period from June to September). Our results are important for regional C budget considerations and for optimizing grassland management to improve SOM storage.

Acknowledgments

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