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Land-use impact on soil carbon and nitrogen sequestration in typical steppe ecosystems, Inner Mongolia

HE Nianpeng[1], ZHANG Yunhai[2], DAI Jingzhong[3], HAN Xingguo[2], BAOYIN Taogetao [4], *YU Guirui[1]

1. Key Laboratory of Ecosystem Network Observation and Modeling, Institute of Geographic Sciences and Natural Resources Research, CAS, Beijing 100101, China;
2. State Key Laboratory of Vegetation and Environmental Change, Institute of Botany, CAS, Beijing 100093, China;
3. Inner Mongolia Agricultural University, Hohhot 010018, China;
4. Institute of Natural Resources, Inner Mongolia University, Huhhot 010021, China

Abstract: To explore the optimal land-use for soil carbon (C) sequestration in Inner Mongolian grasslands, we investigated C and nitrogen (N) storage in soil and soil fractions in 8 floristically and topographically similar sites which subjected to different land-use types (free-grazing, grazing exclusion, mowing, winter grazing, and reclamation). Compared with free-grazing grasslands, C and N storage in the 0–50 cm layer increased by 18.3% (15.5 Mg C ha^{-1}) and 9.3% (0.8 Mg N ha^{-1}) after 10-yr of grazing exclusion, respectively, and 21.9% (18.5 Mg C ha^{-1}) and 11.5% (0.9 Mg N ha^{-1}) after 30-yr grazing exclusion, respectively. Similarly, soil C and N storage increased by 15.3% (12.9 Mg C ha^{-1}) and 10.2% (0.8 Mg N ha^{-1}) after 10-yr mowing, respectively, and 19.2% (16.2 Mg C ha^{-1}) and 7.1% (0.6 Mg N ha^{-1}) after 26-yr mowing, respectively. In contrast, soil C and N storage declined by 10.6% (9.0 Mg C ha^{-1}) and 11.4% (0.9 Mg N ha^{-1}) after 49-yr reclamation, respectively. Moreover, increases in C and N storage mainly occurred in sand and silt fractions in the 0–10 cm soil layer with grazing exclusion and mowing. Our findings provided evidence that Inner Mongolian grasslands have the capacity to sequester C and N in soil with improved management practices, which were in the order: grazing exclusion > mowing > winter grazing > reclamation.

Keywords: carbon; grazing; land-use; nitrogen; reclamation; soil fractions

1 Introduction

Land-use changes are widely recognized as key drivers of global carbon (C) dynamics

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Author: He Nianpeng, Ph.D, specialized in carbon cycles, biogeochemistry, and grassland ecology. E-mail: henp@igsnrr.ac.cn
***Corresponding author:** Yu Guirui, Professor, E-mail: yugr@igsnrr.ac.cn

(IPCC, 2007; Yang *et al*., 2009), and grasslands have received much attention for their substantial potential to act as C sinks in the past few decades (Post and Kwon, 2000; Conant *et al*., 2001; Guo and Gifford, 2002; Soussana *et al*., 2004, Fang *et al*., 2007). With improved management practices, such as soil fertilization, promotion of native vegetation, and sowing of legumes and replanting perennial grasses, most grasslands worldwide are considered to be C sinks (Conant *et al*., 2001; Jones and Donnelly, 2004; Fang *et al*., 2007). However, overgrazing and poor pasture management have led to the loss of C from soil (Elmore and Asner, 2006; He *et al*., 2008; 2011).

Atmospheric CO_2 is sequestrated in soil through the deposition and decomposition of litter, roots, root exudates, and other soil organic matter (SOM). SOM can take the form of labile or stable fractions depending on the chemical composition and recalcitrance of the source and the interaction of its physical and chemical environment (Christensen 1992, 2001). With respect to soil C sequestration, attempts have been made to identify SOM fractions which respond more rapidly to land-use changes than bulk SOM, and former thus serve as early indicators for the overall C stock change (Christensen, 2001; Olk and Gregorich, 2006). Nowadays, particle-size fractions of C pools are widely considered as the important factors controlling SOM turnover, and therefore particle-size fractions (sand, silt, and clay) have been used to evaluate the dynamics and turnover of SOM under various land-use and climate changes (Amelung *et al*., 1998; Leifeld and Kögel-Knabner, 2005; Zinn *et al*., 2007; He *et al*., 2009; He *et al*., 2011).

Temperate grasslands in northern China are approximately 110×10^6 ha. Any management practices that benefit or impair their soil C storage have important implications for regional and global C budgets (Lal, 2009). Fortunately, the implementation of government initiatives to protect grasslands in China since 1998, such as decreasing grazing intensity, grazing exclusion and mowing, is anticipated to increase soil C storage in this region. Recently, researchers have investigated the effect of grazing exclusion and land reclamation on soil C and N storage in different sites in Inner Mongolian grasslands (Zhou *et al*., 2007; He *et al*., 2008; Wang *et al*., 2009). However, without a direct comparison of different land-use types at the same sites, it is difficult to clarify the optimal approaches for grassland C sequestration in this region.

In the present study, we investigated C and N storage in soil and soil fractions in 8 adjacent grassland plots subjected to free-grazing (FG), winter grazing (WG), 10-yr grazing exclusion (GE10), 30-yr grazing exclusion (GE30), 10-yr mowing (M10), 26-yr mowing (M26), 19-yr reclamation (R19), and 49-yr reclamation (R49), respectively. Our objectives were to (i) assess the influence of different land-use types on the C and N storage in soil and soil fractions, and (ii) explore optimal land-use types for soil C sequestration in Inner Mongolian grasslands.

2 Materials and methods

2.1 Study area

The study was conducted in a typical steppe ecosystem in the Mongolian Plateau in northern China (43°33′N, 116°40′E) at the Inner Mongolia Grassland Ecosystem Research Station (IMGERS), Chinese Academy of Sciences (CAS). The climate of this region is semi-arid continental and the mean annual temperature and precipitation values from 1982 to 2009 were 0.96℃ and 334 mm, respectively. The soil is of the chestnut type, i.e., Calcic Kastanozems. The dominant vegetation comprises grassland plants, i.e., *Leymus chinensis* Tzvel, *Stipa grandis* Smirn., and *Cleistogenes squarrosa* Keng (Chen and Wang, 2000).

2.2 Site selection

Eight experimental plots were selected, and subjected to long-term free-grazing (FG), winter grazing (WG), 10-yr grazing exclusion (GE10), 30-yr grazing exclusion (GE30), 10-yr mowing (M10), 26-yr mowing (M26), 19-yr reclamation (R19), and 49-yr reclamation (R49) (Figure 1). Plot WG had been grazed since the winter 1999. Plots GE30 and GE10 were established by fencing previously free-grazing grasslands in 1979 and 1999, respectively. Plot M10 had been mowed annually since mid-August 1999 (Liu *et al*., 2007), and plot M26 had been mowed annually since 1982 (Bao *et al*., 2004). Plots R49 and R19 had been reclaiming since 1960 and 1990, respectively; the cropping system commonly used on these croplands consisted of a wheat-(*Triticum* spp.) or rapeseed (*Brassica napus*)-fallow rotation. Wheat or rapeseed is usually sowed in early May and harvested in late August. In the fallow year, all the weeds were incorporated into the soil as green manure by plowing. Fertilizers containing 54–83 kg N ha^{-1} and 42–71 kg P ha^{-1} were applied at sowing once each cropping year. The 8 plots are contiguous, and distributed in the same upper basalt platform with similarly former floristics and topography (Figure 1).

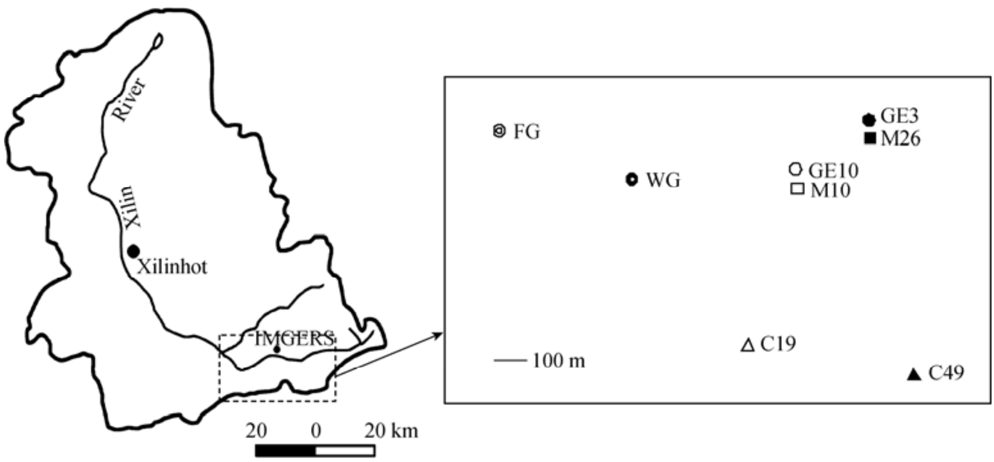

Figure 1 Experimental plots and their relative positions
FG, free-grazing; WG, winter grazing; GE10, 10-yr grazing exclusion; GE30, 30-yr grazing exclusion; M10, 10-yr mowing; M26, 26-yr mowing; R19, 19-yr reclamation; R49, 49-yr reclamation. IMGERS, Inner Mongolia Grassland Ecosystem Research Station (43°33′N, 116°40′E)

2.3 Field sampling

In early May 2009, before tilling, we selected 5 random sampling sites within each plot to serve as replicates. The sites were at least 30 m apart from each other and from the boundary of plots FG, WG, GE10, GE30, R19, and R49. Sampling sites for plots M10 and M26 were based on the originally random sampling quadrants of Liu *et al*. (2007) and Bao *et al*. (2004), respectively. At each sampling site, 3 soil cores (1 m apart) were collected and combined from 3 layers at depths of 0–10 cm, 10–30 cm, and 30–50 cm. A total of 15 soil samples were taken from each experimental plot. Soil bulk density was measured using soil cores at depths of 0–10 cm, 10–30 cm, and 30–50 cm, with 3 replicates for each plot (Blake and Hartage, 1986); this allowed us to estimate the mass of C and N at each site.

2.4 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). In brief, 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 Co., Ltd., 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℃ and ground for chemical analysis.

2.5 Calculations

Soil organic carbon (SOC, Mg C ha^{-1}) and total nitrogen (STN, Mg N ha^{-1}) storage was calculated on a ground area basis to 50 cm soil depths as follows:

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

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

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

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

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

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

where $\mathrm{C}_{con.}(fraction_i)$ is the C content of the soil fraction (%), $N_{con.}(fraction_i)$ is the N content of the soil fraction (%), and F is the content of fraction in the soil (g fraction kg^{-1} soil). Moreover, storage of SOC and STN, in the 0–10 cm, 10–30 cm, and 30–50 cm soil layers were standardized to 1 cm depth in order to better compare the differences among soil layers (Tables 3 and 4).

2.6 Statistical analysis

All data are presented as mean ± 1 SD (n = 5). One-way ANOVA (with Duncan's post-hoc test for multiple comparisons) was used to evaluate the influence of land-use types on C and

N storage in soil and soil fractions. A significance level of $P < 0.05$ was used for all tests. All analyses were conducted following the program SPSS, ver. 11.0.

3 Results and discussion

3.1 Bulk density and particle-size fractions

Compared with plot FG, grazing exclusion or mowing significantly decreased bulk density (BD) in the 0–10 cm soil layer ($P < 0.01$), whereas reclamation increased soil BD ($P < 0.01$) (Table 1). BD was not significantly different among land-use types in the 10–30 cm and

Table 1 Characteristics of soil particle-size fractions and bulk density in the 8 experimental plots

Depth (cm)	Sites	Particle-size fractions			Bulk density ($g\ cm^{-3}$)
		Sand (50–2000 μm) ($g\ kg^{-1}$ soil)	Silt (2–50 μm) ($g\ kg^{-1}$ soil)	Clay (<2 μm) ($g\ kg^{-1}$ soil)	
0–10	FG †	747.7 ± 26.4 a ‡	226.1 ± 29.4 a	26.3 ± 5.3 a	1.40 ± 0.02 a
	WG	720.5 ± 17.9 ab	253.4 ± 14.5 ab	26.1 ± 3.5 a	1.38 ± 0.02 ab
	GE10	694.3 ± 5.1 bc	273.9 ± 8.2 abc	31.8 ± 6.0 a	1.34 ± 0.03 c
	GE30	708.6 ±17.2 abc	256.6 ± 13.1 abc	34.8 ± 4.2 a	1.28 ± 0.02 d
	M10	660.4 ± 10.5 c	309.4 ± 12.0 c	30.2 ± 9.7 a	1.31 ± 0.01 d
	M26	710.5 ± 53.5 bc	258.3 ± 49.1 abc	31.2 ± 4.9 a	1.34 ± 0.03 c
	R19	685.2 ± 8.2 bc	289.1 ± 9.7 bc	25.7 ± 6.4 a	1.34 ± 0.03 c
	R49	719.5 ± 39.2 bc	254.3 ± 36.9 abc	26.3 ± 3.3 a	1.36 ± 0.01 bc
	F-value	2.52*	2.36*	1.28 NS	15.20 **
10–30	FG	733.3 ± 5.1 a	239.7 ± 9.8 a	26.9 ± 6.0 a	1.44 ± 0.02 a
	WG	718.7 ± 31.4 ab	249.7 ± 29.6 ab	31.6 ± 3.1 a	1.44 ± 0.02 a
	GE10	728.3 ± 21.1 ab	239.5 ± 15.2 a	32.3 ± 7.5 a	1.43 ± 0.02 a
	GE30	730.2 ± 16.5 a	235.6 ± 17.7 a	34.2 ± 3.9 a	1.42 ± 0.04 a
	M10	688.1 ± 43.2 ab	282.8 ± 35.8ab	29.1 ± 13.0 a	1.43 ± 0.01 a
	M26	704.5 ± 24.6 ab	264.7 ± 20.5 ab	30.8 ± 4.9 a	1.43 ± 0.02 a
	R19	681.8 ± 13.8 b	287.9 ± 13.6 b	30.2 ± 10.2 a	1.45 ± 0.02 a
	R49	721.6 ± 33.7 ab	250.0 ± 33.9 ab	28.5 ± 2.1 a	1.48 ± 0.02 b
	F-value	1.87 NS	2.29 NS	0.34 NS	3.03 NS
30–50	FG	751.1 ± 33.6 a	221.5 ± 31.5 a	27.4 ± 6.2 a	1.50 ± 0.02 a
	WG	741.5 ± 7.6 ab	227.9 ± 6.1 ab	30.6 ± 5.0 a	1.49 ± 0.02 a
	GE10	741.6 ± 18.1 ab	228.3 ± 17.5 ab	30.1 ± 2.2 a	1.49 ± 0.02 a
	GE30	737.8 ± 23.0 abc	228.7 ± 23.4 ab	33.5 ± 0.7 a	1.48 ± 0.01 a
	M10	713.5 ± 46.6 abc	257.3 ± 44.9 abc	29.2 ± 3.3 a	1.48 ± 0.03 a
	M26	687.5 ± 54.9 bc	282.3 ± 54.6 bc	30.2 ± 10.1 a	1.49 ± 0.02 a
	R19	681.3 ± 8.9 c	289.9 ± 7.4 c	28.8 ± 5.8 a	1.49 ± 0.02 a
	R49	702.3 ± 55.9 abc	270.0 ± 58.5 bc	27.7 ± 3.0 a	1.49 ± 0.03 a
	F-value	2.44 NS	2.54 NS	0.43 NS	0.64 NS

† FG, free grazing; WG, winter grazing; GE10, 10-yr grazing exclusion; GE30, 30-yr grazing exclusion; M10, 10-yr mowing; M26, 26-yr mowing; R19, 19-yr reclamation; R49, 49-yr reclamation

‡ Data are represented as mean ± 1 SD (n = 5), and designated with the same letters are not significantly different ($P <$ 0.05).

30–50 cm soil layer. This result is generally consistent with earlier findings that soil BD declined with increasing SOM because of increased soil porosity (Whalen *et al.*, 2003). Soil BD is higher in croplands than in grassland, possibly because of the duration and intensity of cultivation (Mikhailova *et al.*, 2000).

Sand was the most abundant particle in the particle-size distribution of the 8 soils. In the 0–10 cm soil layer, the sand fraction accounted for 72–75% of the total soil weight; silt accounted for 7.1–14.9% of the total soil weight, whereas the clay content was low (Table 1). Land-use changes had an apparent effect on the particle-size distribution in the 0–10 cm soil layer.

3.2 C and N storage in bulk soil

SOC storage in the 0–50 cm soil layer differed significantly among land-use types ($F = 14.10$, $P < 0.001$). SOC storage was the highest in plot GE30 (103.12 Mg C ha^{-1}) and the lowest in plot R49 (75.65 Mg C ha^{-1}) (Figure 2a). Compared with plot FG, grazing exclusion and mowing significantly increased SOC storage in the 0–50 cm layer (Table 2). SOC storage in plot WG was slightly but not significantly higher than that in FG. In contrast, SOC storage in plot R49 was significantly lower than that in plot FG (Figure 2a and Table 2). The influence of land-use changes on SOC storage was more evident in the 0–10 cm and 10–30 cm soil layers, and this effect declined with soil depth (Table 3).

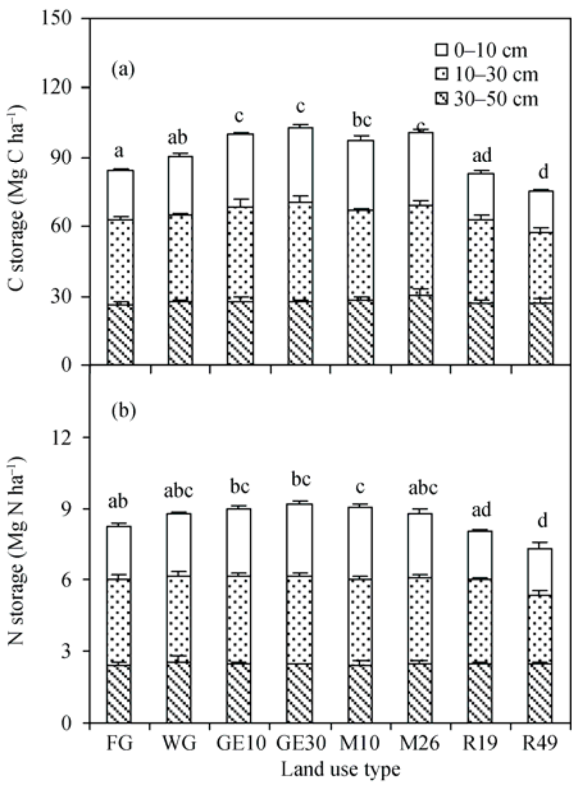

Figure 2 Soil C and N storage in the 0–50 cm layer under different land-use types. Values are means ± 1 SD (n = 5). Data with the same lower letter are not significantly different among different treatments (Duncan's test as post-hoc test for multiple comparisons)

Table 2 Absolute changes and relative changes in soil C and N storage in 0–50 cm soil layer

	Absolute change		Relative change	
	Total C (MgC ha^{-1})	Total N (Mg N ha^{-1})	Total C (%)	Total N (%)
WG	5.92 †	0.55	7.00	6.67
GE10	15.47 *	0.76 *	18.28	9.26
GE30	18.52 *	0.94 *	21.89	11.50
M10	12.91 *	0.84 *	15.26	10.18
M26	16.24 *	0.58	19.19	7.08
R19	–1.77	–0.21	–2.09	–2.53
R49	–8.95 *	–0.94 *	–10.58	–11.41

† Data are different with the data of plot FG (means, n = 5), and absolute change with asterisk indicates significant difference (P = 0.05).

Table 3 C storage in soil fractions under different land-use types

			C storage (Mg C ha^{-1} cm^{-1} depth)			
Depth (cm)	Sites	Bulk soil	Sand (50–2000 μm)	Silt (2–50 μm)	Clay (<2 μm)
0–10	FG	2.20 ± 0.02 ^{a†}	1.04 ± 0.07 ^{a}	1.03 ± 0.09 ^{ab}	0.16 ± 0.02 ^{ab}
	WG	2.59 ± 0.12 ^{b}	1.30 ± 0.08 ^{b}	1.12 ± 0.03 ^{abc}	0.17 ± 0.01 ^{ab}
	GE10	3.17 ± 0.07 ^{c}	1.38 ± 0.08 ^{b}	1.28 ± 0.04 ^{c}	0.21 ± 0.02 ^{bc}
	GE30	3.28 ± 0.11 ^{c}	1.51 ± 0.15 ^{b}	1.20 ± 0.06 ^{bcd}	0.22 ± 0.01 ^{c}
	M10	3.06 ± 0.19 ^{c}	1.29 ± 0.08 ^{b}	1.10 ± 0.10 ^{abcd}	0.20 ± 0.01 ^{abc}
	M26	3.20 ± 0.12 ^{c}	1.43 ± 0.11 ^{b}	1.32 ± 0.05 ^{d}	0.19 ± 0.03 ^{abc}
	R19	2.00 ± 0.18 ^{ad}	0.89 ± 0.12 ^{ac}	1.16 ± 0.07 ^{abcd}	0.15 ± 0.02 ^{a}
	R49	1.83 ± 0.07 ^{d}	0.77 ± 0.04 ^{c}	1.01 ± 0.06 ^{a}	0.16 ± 0.01 ^{ab}
	F-value	29.23 **	9.82 **	3.47 *	2.83 *
10–30	FG	1.82 ± 0.09 ^{a}	0.76 ± 0.06 ^{a}	1.03 ± 0.04 ^{ac}	0.16 ± 0.02 ^{a}
	WG	1.85 ± 0.04 ^{a}	0.71 ± 0.01 ^{a}	1.06 ± 0.12 ^{abc}	0.18 ± 0.01 ^{a}
	GE10	2.02 ± 0.16 ^{a}	0.76 ± 0.05 ^{a}	1.04 ± 0.06 ^{abc}	0.21 ± 0.01 ^{a}
	GE30	2.12 ± 0.14 ^{a}	0.78 ± 0.09 ^{a}	1.06 ± 0.02 ^{abc}	0.20 ± 0.02 ^{a}
	M10	1.94 ± 0.05 ^{a}	0.74 ± 0.06 ^{a}	1.17 ± 0.04 ^{c}	0.19 ± 0.05 ^{a}
	M26	1.91 ± 0.11 ^{a}	0.71 ± 0.05 ^{a}	1.12 ± 0.06 ^{bc}	0.19 ± 0.02 ^{a}
	C19	1.79 ± 0.09 ^{ab}	0.65 ± 0.04 ^{a}	0.92 ± 0.06 ^{ab}	0.16 ± 0.02 ^{a}
	C49	1.51 ± 0.09 ^{b}	0.57 ± 0.18 ^{a}	0.86 ± 0.16 ^{a}	0.15 ± 0.01 ^{a}
	F-value	3.17 *	0.90 ^{NS}	1.84 ^{NS}	1.06 ^{NS}
30–50	FG	1.31 ± 0.07 ^{a}	0.40 ± 0.01 ^{a}	0.73 ± 0.07 ^{a}	0.14 ± 0.02 ^{a}
	WG	1.38 ± 0.03 ^{a}	0.43 ± 0.03 ^{a}	0.77 ± 0.04 ^{a}	0.17 ± 0.01 ^{a}
	GE10	1.39 ± 0.09 ^{a}	0.42 ± 0.08 ^{a}	0.75 ± 0.04 ^{a}	0.15 ± 0.01 ^{a}
	GE30	1.39 ± 0.04 ^{a}	0.40 ± 0.03 ^{a}	0.72 ± 0.03 ^{a}	0.16 ± 0.01 ^{a}
	M10	1.40 ± 0.08 ^{a}	0.41 ± 0.01 ^{a}	0.78 ± 0.09 ^{a}	0.15 ± 0.03 ^{a}
	M26	1.53 ± 0.11 ^{a}	0.43 ± 0.04 ^{a}	0.75 ± 0.08 ^{a}	0.16 ± 0.01 ^{a}
	C19	1.35 ± 0.05 ^{a}	0.38 ± 0.02 ^{a}	0.77 ± 0.04 ^{a}	0.14 ± 0.02 ^{a}
	C49	1.35 ± 0.11 ^{a}	0.37 ± 0.12 ^{a}	0.77 ± 0.11 ^{a}	0.15 ± 0.01 ^{a}
	F-value	0.87 ^{NS}	0.21 ^{NS}	0.15 ^{NS}	0.66 ^{NS}

† Data are standard to per cm depth (mean ± 1 SD, n = 5), and data designated with the same letters are not significantly different at P = 0.05 level.

STN storage in the 0–50 cm soil layers was significantly different among land-use types ($F = 5.31$, $P < 0.001$; Figure 2b). Compared with plot FG, grazing exclusion significantly enhanced STN storage in the 0–50 cm soil layer (Table 2); STN storage significantly increased in plot M10, but the increase was not apparent in plot M26. Moreover, STN storage in plot R49 was significantly lower than that in plot FG. In general, the influence of land-use changes on STN storage was greater in the 0–10 cm and 10–30 cm soil layers than in the 30–50 cm soil layer (Table 4).

Table 4 N storage in soil fractions under different land-use types

Depth (cm)	Sites	Bulk soil	N storage (Mg N ha^{-1} cm^{-1} depth)		
			Sand (50–2000 μm)	Silt (2–50 μm)	Clay (<2 μm)
0–10	FG	0.221 ± 0.013 ^{a†}	0.099 ± 0.006 ^{ab}	0.104 ± 0.011 ^{a}	0.017 ± 0.002 ^{ab}
	WG	0.261 ± 0.010 ^{b}	0.118 ± 0.014 ^{b}	0.121 ± 0.006 ^{ab}	0.018 ± 0.001 ^{ab}
	GE10	0.283 ± 0.010 ^{b}	0.117 ± 0.007 ^{bc}	0.132 ± 0.009 ^{b}	0.022 ± 0.002 ^{abc}
	GE30	0.301 ± 0.010 ^{b}	0.128 ± 0.002 ^{c}	0.155 ± 0.006 ^{bc}	0.023 ± 0.005 ^{c}
	M10	0.301 ± 0.015 ^{b}	0.135 ± 0.003 ^{bc}	0.131 ± 0.008 ^{c}	0.026 ± 0.003 ^{bc}
	M26	0.271 ± 0.015 ^{b}	0.128 ± 0.020 ^{bc}	0.121 ± 0.007 ^{abc}	0.022 ± 0.001 ^{abc}
	R19	0.202 ± 0.006 ^{a}	0.074 ± 0.007 ^{a}	0.117 ± 0.003 ^{ab}	0.016 ± 0.002 ^{a}
	R49	0.193 ± 0.026 ^{a}	0.075 ± 0.009 ^{a}	0.104 ± 0.015 ^{a}	0.017 ± 0.001 ^{ab}
	F-value	11.62 **	6.68 **	4.40 **	2.85 *
10–30	FG	0.180 ± 0.011 ^{a}	0.059 ± 0.005 ^{a}	0.104 ± 0.005 ^{a}	0.017 ± 0.002 ^{ab}
	WG	0.180 ± 0.009 ^{a}	0.061 ± 0.003 ^{a}	0.106 ± 0.012 ^{a}	0.019 ± 0.001 ^{ab}
	GE10	0.184 ± 0.007 ^{a}	0.059 ± 0.007 ^{a}	0.104 ± 0.006 ^{a}	0.020 ± 0.002 ^{ab}
	GE30	0.180 ± 0.007 ^{a}	0.056 ± 0.005 ^{a}	0.116 ± 0.005 ^{a}	0.019 ± 0.005 ^{ab}
	M10	0.185 ± 0.008 ^{a}	0.056 ± 0.008 ^{a}	0.106 ± 0.004 ^{a}	0.023 ± 0.003 ^{b}
	M26	0.182 ± 0.006 ^{a}	0.058 ± 0.005 ^{a}	0.113 ± 0.012 ^{a}	0.020 ± 0.002 ^{b}
	R19	0.176 ± 0.005 ^{a}	0.057 ± 0.006 ^{a}	0.101 ± 0.004 ^{a}	0.017 ± 0.003 ^{ab}
	R49	0.144 ± 0.011 ^{b}	0.046 ± 0.010 ^{a}	0.089 ± 0.006 ^{a}	0.016 ± 0.001 ^{a}
	F-value	3.36 **	0.61 ^{NS}	1.50 ^{NS}	0.91 ^{NS}
30–50	FG	0.120 ± 0.007 ^{a}	0.031 ± 0.004 ^{a}	0.075 ± 0.008 ^{a}	0.015 ± 0.002 ^{a}
	WG	0.128 ± 0.012 ^{a}	0.034 ± 0.003 ^{a}	0.075 ± 0.005 ^{a}	0.018 ± 0.001 ^{a}
	GE10	0.123 ± 0.004 ^{a}	0.031 ± 0.005 ^{a}	0.076 ± 0.005 ^{a}	0.016 ± 0.001 ^{a}
	GE30	0.122 ± 0.008 ^{a}	0.032 ± 0.004 ^{a}	0.076 ± 0.007 ^{a}	0.017 ± 0.001 ^{a}
	M10	0.123 ± 0.002 ^{a}	0.031 ± 0.002 ^{a}	0.073 ± 0.002 ^{a}	0.018 ± 0.001 ^{a}
	M26	0.122 ± 0.007 ^{a}	0.031 ± 0.002 ^{a}	0.075 ± 0.007 ^{a}	0.017 ± 0.004 ^{a}
	C19	0.124 ± 0.004 ^{a}	0.034 ± 0.003 ^{a}	0.075 ± 0.005 ^{a}	0.015 ± 0.001 ^{a}
	C49	0.123 ± 0.003 ^{a}	0.030 ± 0.004 ^{a}	0.074 ± 0.014 ^{a}	0.015 ± 0.002 ^{a}
	F-value	0.13 ^{NS}	0.22 ^{NS}	0.21 ^{NS}	0.54 ^{NS}

† Data are standard to per cm depth (mean ± SD, n = 5), and data designated with the same letters are not significantly different at $P = 0.05$ level.

In agreement with some previous studies (e.g., Zhou *et al*., 2007; He *et al*., 2008), our results suggested that SOC and STN storage increased significantly with grazing exclusion in

Inner Mongolian grasslands. Conant *et al*. (2001) reported that annual increases of C and N storage were 2.9% and 2.2% under grazing management and fertilization, respectively. On the basis of our data, the annual increases in C and N storage in the 0–50 cm soil layer was 1.83% and 0.93% in plot GE10, and 0.73% and 0.38% in plot GE30, respectively (Table 2). The findings support the hypothesis that soil C and N storage sustained an initial rapid increase with the introduction of grazing exclusion, followed by a steady phase over time (Wu *et al*., 2008). Presumably, long-term grazing exclusion (i.e. 30 years), in comparison to short-term grazing exclusion (i.e. 10 years), may be that an increase in ANPP would drive greater competition for resource, such as nutrients and water, and that increase in nutrient demand would drive more SOM mineralization when natural disturbances – including large-animal grazing and fire – are excluded. Another plausible 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 SOM decomposition or soil respiration, but make the precipitation unavailable for plant uptake in arid region (Huxman *et al*., 2004; Schwinning and Sala, 2004).

The results suggested that mowing practices can significantly enhance SOC and STN storage in Inner Mongolian grasslands. Compared with plot FG, SOC and STN storage increased by 15.26% and 10.18% in plot M10, and by 19.19% and 7.08% in plot M26, respectively (Table 2). The increase in SOC occurred more slowly with more frequent mowing, although total SOC storage continued to increase. Unexpectedly, STN storage was higher in plot M10 than in plot M26, which indicates that STN storage initially increased rapidly with the introduction of mowing, and then slightly decreased. The different trends for SOC and STN under continuous mowing are interesting and unexpected. One possible explanation for this discrepancy is that long-term continuous mowing could lead to significant nutrient and element loss, especially for soil N. This would restrain primary production and SOC and STN accumulation because N deficiency is one important factor controlling primary production in Inner Mongolian grasslands (Yuan *et al*., 2006). Another possibility is that changes in the composition of dominant species and plant functional groups could influence the sequestration potential of SOC and STN in grasslands. In a study on the compositional dynamics of plant functional groups and community stability during 17 years of successive mowing in plot M26, Bao *et al*. (2004) found that rhizome grasses, which were previously dominant, were replaced successively by annuals and biennials, and bunch grasses.

The C and N sequestration capacity was lower with winter grazing than with grazing exclusion and mowing. SOC and STN storage increased by 7.0% and 6.7%, respectively, during 11 years of winter grazing (Table 2). This finding has significant implications for C sequestration in grasslands, since it is a common management practice in the region.

Our results suggested that SOC and STN storage declined significantly with long-term reclamation in Inner Mongolian grasslands. Moreover, the greatest changes occurred in the 0–10 cm soil layer. SOC and STN in the 0–50 cm soil layer in plot R49 decreased by 10.6% and 11.4%, respectively. Wang *et al*. (2009) reported that, after 42 years of reclamation, SOC and STN storage in the 0–30 cm soil layer decreased by 25% and 16%, respectively. This difference of data was likely a result of the different soil depths in the two studies; if our calculations only included the 0–30 cm soil layer, SOC and STN would decrease by

17.6% and 17.2, respectively. In a meta-analysis, Guo and Gifford (2002) reported that conversion of grassland into arable land decreased 60% soil C, which mainly occurred in the upper soil horizons. Disruption of soil aggregate structure due to cultivation enhances the decomposition of SOM and renders the soil susceptible to water and wind erosion (Neff *et al*., 2005). In arid Inner Mongolian grasslands, considerable loss of SOC and STN is caused by wind and water erosion, particularly in areas with sandy soil and high wind speed (Su *et al*., 2004; Yan *et al*., 2005; Hoffmann *et al*., 2008).

Overall, our findings have significant implications for assessing the C and N sequestration of different land-use types and suggest that improved management practices are important approaches to implement the C sequestration of Inner Mongolian grasslands. In a review by Conant *et al*. (2001), rates of C sequestration in grasslands with different types of improved management practices (e.g., fertilization, improved grazing management, conversion from cultivation and native vegetation, sowing of legumes and grasses, earthworm introduction, and irrigation) varied from 0.11 to 3.04 Mg C ha^{-1} yr^{-1}, with a mean of 0.54 Mg C ha^{-1} yr^{-1}, and were markedly affected by biome type and climate. Lal (2009) estimated that C sequestration rates ranged from o to 2 Mg C ha^{-1} yr^{-1} in arid regions. Deteriorating environmental conditions prompted local government in Inner Mongolia to officially restrict or ban livestock grazing as of the year 2000. Therefore, it is anticipated that soil C storage will increase in the region because grazing exclusion and mowing are the encouraged land-use types.

3.3 C and N stored in particle-size fractions

For all soil layers, the C and N concentration in soil fractions (g C kg^{-1} fraction) was in the following order: clay > silt > sand. C storage in the sand fraction (50–2000 μm) differed significantly among land-use types in the 0–10 cm soil layer (F = 9.82, $P < 0.01$) (Table 3). Compared with plot FG, plots GE10, GE30, M10, and M26 showed a significant increase in the C storage of sand fraction; however, C storage in the sand fraction deceased in plot R49. Moreover, grazing exclusion and mowing significantly enhanced C storage in the silt fraction (F = 3.47, $P < 0.01$). For the 10–30 cm and 30–50 cm soil layers, there were no significant differences in the C storage in soil fractions (sand, silt, and clay) among land-use types (Table 3). The relative C pools in the sand fractions found in the 0–10 cm soil layer were significantly different among land-use types ($P < 0.01$); these were higher in plots subjected to grazing exclusion and mowing, but lower in plots managed by reclamation (Figure 3). In the soil profile, the relative contribution of the sand fraction to C storage decreased with soil depth; accordingly, silt and clay increased relatively.

The N storage in all soil fractions (sand, silt and clay) varied significantly among land-use types in the 0–10 cm soil layer (Table 4). Grazing exclusion significantly increased N storage in the sand fraction, compared with free-grazing or reclamation ($P < 0.05$). Similar trends were also observed in silt and clay fractions. For the 10–30 cm soil layer, N storage in sand fraction was significantly different among land-use types (F = 3.36, $P < 0.01$), but not significant for the silt and clay fractions. The relative N pool in sand fractions found in the 0–10 cm soil layer decreased significantly in the reclamation plot ($P < 0.05$), but increased in the grazing exclusion and mowing plots. In the soil profile, the contribution of sand fractions to N pool decreased with soil depth, while the contribution of silt and clay fractions increased (Figure 3).

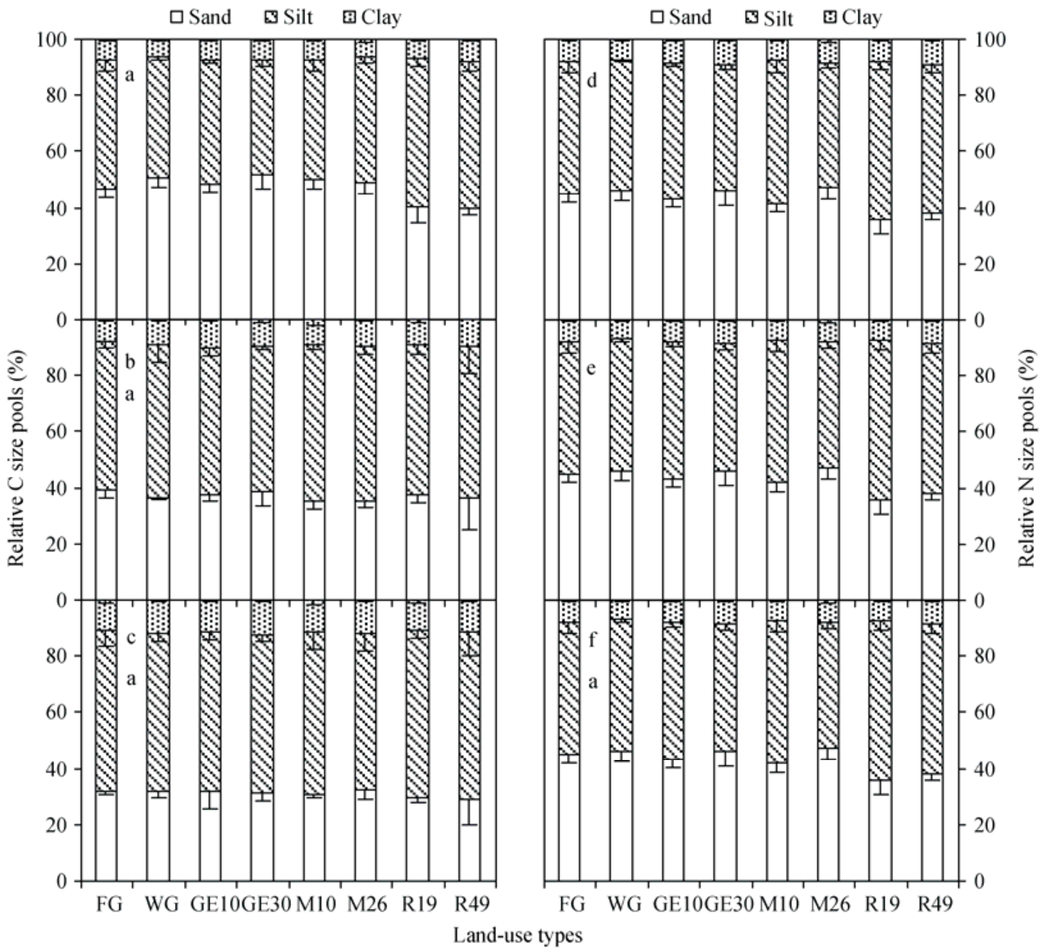

Figure 3 Relative C and N size pools (%) in different soil particle-size fractions. Figures a and d were relative C and N pools in 0–10 cm soil layer, respectively; similarly, b and e in 10–30 cm soil layer; c and f in 30–50 cm soil layer.

The significant increase in the C and N storage of sand and silt fractions in plots GE10, GE30, M10 and M26 suggested that such fractions are important contributors to the accumulation of C and N in whole soil. The accumulation of C and N in the grassland soils was considerable; it was a consequence of increased primary production, and subsequent increases in SOM inputs through surface litter and roots from grazing exclusion and mowing practices (Wu *et al*., 2008). In contrast, long-term grazing and reclamation lead to a substantial decomposition of SOM in the sand fraction and thus decreased the C pool (Pei *et al*., 2008; Steffens *et al*., 2008). Christensen (2001) demonstrated that management practices have a greater effect on the SOM in the sand fraction compared with clay and silt fractions. Some researchers suggest that land disturbances such as cultivation and overgrazing can decrease the stable C content of alpine pastureland, possibly by exposing the protected SOM to microbial attack (Li *et al*., 2007).

Previous researches have documented that land-use changes or particular management can have a marked effect on the storage of C in particle-size fractions. Therefore, SOC in particle fractions could serve as an early indicator of the effect of land-use changes on soil C storage (Solomon *et al*., 2002; Leifeld and Kögel-Knabner, 2005). He *et al*. (2009) reported that long-term grazing exclusion had an apparent effect on the C and N concentrations of sand fraction, and significantly increased the C and N storage in the 0–10 cm soil layer. Furthermore, the concentrations of C and N in soil fractions tended to increase with decreasing particle size; this trend could be attributed to the effect of the dilution of SOM (Amelung *et al*., 1998; Zinn *et al*., 2007).

3.4 C:N ratio

C:N ratios of bulk soil in the 0–10 cm soil layer were slightly higher in plots subjected to grazing exclusion and mowing, but were not significantly different among land-use types (Figure 4). Similar trends were observed in the 10–30 cm and 30–50 cm soil layers. For all plots, the C:N rations were in the order: sand > silt > clay for all plots. Grazing exclusion and mowing significantly enhanced the C:N ratio of sand fraction in the 0–10 cm and 10–30 cm soil layers (F = 4.56, $P < 0.01$ for 0–10 cm soil layer; F = 3.98, $P < 0.01$ for 10–30 cm soil layer). The changes in the C:N ratio of silt and clay was small.

The higher C:N ratios of whole soil and sand fraction found in plots of grazing exclusion and mowing suggests that abundant slightly decomposed raw organic material accumulates in the soil during such practices, particularly in the 0−10 cm soil layer (Covaleda *et al*., 2006). Sand fraction had a higher C:N ratio than those of silt and clay fractions; this can be attributed to the incompletely humified organic material and suggest that fine roots and fungi serve as active binding agents. In contrast, the SOM associated with silt and clay is more processed and stable (Christensen, 2001). The C:N ratio governs the degradation of fresh plant residues and is important for soil C sequestration.

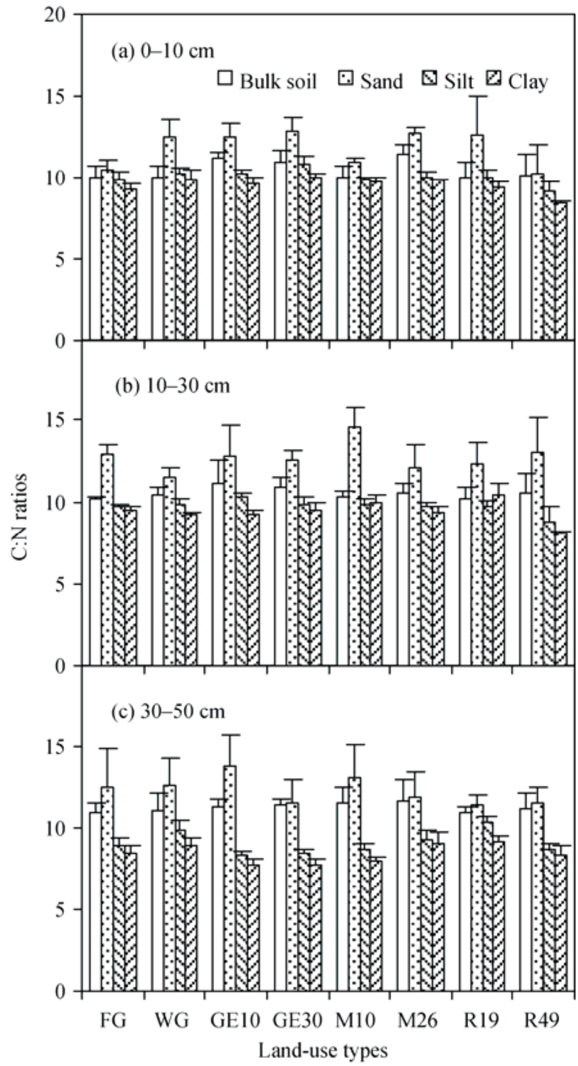

Figure 4 C:N ratios of soil and soil fraction under different land-use types. Values are means ± 1 SD (n = 5).

4 Conclusions

Land-use changes have a significant influence on SOC and STN storage in Inner Mongolian grasslands. Management practices such as grazing exclusion, mowing, and winter grazing can enhance the storage of SOC and STN, and they were in the order: grazing > mowing > winter grazing. Moreover, the findings provide support for the hypothesis that soil C storage sustained an initial rapid increase with the introduction of grazing exclusion and mowing, followed by a relatively steady phase over time that was likely a result of N deficiency. In contrast, SOC and STN storage declined significantly with long-term reclamation in Inner Mongolian grasslands, and the greatest changes occurred in the 0–10 cm soil depth. Grazing exclusion and mowing increased C and N storage in sand and silt fractions, and therefore significantly contributed to the newly accumulated C and N in whole soil. Given the evidence for increasing trends in SOC, STN, and C:N ratios with long-term grazing exclusion and mowing, we assume that the ability of these practices to enhance C sequestration is closely related to N input, and t the stability of newly increased SOM needs to be further evaluated.

To our knowledge, this is the first study to examine the influence of different land-use types on C and N storage in soil and soil fractions in adjacent plots in Inner Mongolian grasslands. Our findings provide direct evidence that Inner Mongolian grasslands have the capacity to sequester C in soil with improved management practices, particularly grazing exclusion and mowing. In general, the results support the implementation of improved and viable grassland management in view of grassland C sequestration other than the conversion of grassland to cropland.

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