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Carbon accumulation in the bulk soil and different soil fractions during the rehabilitation of desertified grassland in Horqin Sandy Land (Northern China)

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ABSTRACT

Desertification, which affects more than two-thirds of the world's arid and semi-arid regions, is a significant global ecological and environmental problem. There is a strong link between desertification of the drylands and emission of CO_2 from soil and vegetation to the atmosphere. The Horqin Sandy Land is a severely desertified area in China's agro-pastoral ecotone due to its fragile ecology, combined with unsustainable land management. We estimated changes of organic carbon content in the bulk soil (0–5 cm), in the light-fraction of soil organic matter (based on density fractionation), and in the various particle-size fractions in areas with mobile sand dunes after implementing grazing exclusion (12 and 27 years) and tree and shrub planting (22 and 24 years). Carbon stocks in the bulk soil and all soil density and particle-size fractions increased significantly in the exclosure and plantation plots. The average rates of carbon accumulation in the bulk soil in the exclosure and plantation plots were 16.0 and 17.8 g m^{-2} y^{-1}, respectively, versus corresponding values of 2.3 and 7.1 g m^{-2} y^{-1} for the light fraction, 4.3 and 8.0 g m^{-2} y^{-1} for the coarse fraction, 5.0 and 3.4 g m^{-2} y^{-1} for the fine sand, 4.5 and 4.2 g m^{-2} y^{-1} for the very fine sand, and 1.8 and 1.8 g m^{-2} y^{-1} for the silt+clay fraction. The older the exclosure and plantation, the more carbon accumulated in the bulk soil and in each fraction. The carbon pool exceeded the level in non-desertified grasslands after 27 years of grazing exclosure and 24 years of the shrub plantation. Our results suggest that both grazing exclusion and planting trees and shrubs can restore desertified grassland, creating a high potential for sequestering soil carbon, but that the plantations appeared to accumulate soil carbon faster than the exclosures.

INTRODUCTION

The threat of climate change exacerbated by increased emission of greenhouse gases has attracted considerable scientific attention with the goal of improving our understanding of ecosystem carbon (C) source and sink relationships (Parker *et al.* 2002). Carbon dioxide (CO_2) is the most important greenhouse gas in terms of its impact on global warming. Land-use change has contributed significantly to global increases in the atmospheric CO_2 concentration, though the magnitude of its impact is smaller than that of fossil fuel use (IPCC 2007). Soil is the largest pool of terrestrial organic C in the biosphere (Jobbágy and Jackson 2000). Therefore, proportionally small changes in the soil organic carbon (SOC) storage can have a large impact on the concentration of atmospheric CO_2 (Kirschbaum 2000). When natural ecosystems are converted into cropland or undergo degradation, this often leads to depletion of the SOC stock, primarily due to reduced input of carbon from litter, decreased aboveground and belowground biomass, enhanced decomposition caused by the physical disturbance, and reduced vegetation cover, which increases soil erosion (Lal 2009, Poeplau and Don 2013).

Desertification, which can be defined as land degradation in arid, semi-arid, and dry sub-humid areas as a result of various factors, including climate variations and human activities (UNCED 1992), is a serious threat to the environment and human welfare. The world's arid and semi-arid regions cover ≅45% of the global land surface and are populated by approximately 1 billion humans (Nosetto *et al.* 2006, Verón *et al.* 2006). Desertification, which affects more than two-thirds of this region, leads to deterioration of the vegetation community and the soil structure, which in turn decreases total ecosystem C storage and increases CO_2 emission from plants and the soil into the atmosphere (Helldén and Tottrup 2008, Lal 2009). The total historic C losses from the plant-soil continuum caused by desertification have been estimated at 19 to 29 Pg globally (Lal 2001). Reversing the widespread degradation that has occurred in arid and semi-arid regions creates a considerable potential to increase C sequestration and improve soil quality. This will become possible when degraded ecosystems are allowed to revert to natural vegetation, are protected from disturbance and erosion, are managed using appropriate management practices, or are planted using perennial vegetation (Nosetto *et al.* 2006, Niu *et al.* 2013, Liu *et al.* 2014).

Soil organic matter (SOM) comprises pools that are divided according to their biological stability (labile, stable, and inert), decomposition rate (fast, slow, and very slow), and turnover time (short, long, and very long) (Trumbore 2009). Studies of organic matter turnover in soil have increasingly relied on physical fractionation based on the characteristics of the SOM and the density and size of the soil particles, because this approach emphasizes the importance of the interactions between the different organic and inorganic soil components during the decomposition and cycling of organic matter (Christensen 2001). Density fractionation, typically using a solution at a density of 1.6 to 2.0 g cm^{-3}, separates SOM into low- and high-density fractions that are referred to as the light fraction (LF) and the heavy fraction (HF), respectively (Swanston *et al.* 2002). The LF is plant-like material with a high C concentration, and is generally considered to be more sensitive to changes in climate and land use than the total SOM and HF pools (Haynes 2000, Six *et al.* 2002). Therefore, the LF has been suggested as an early indicator of land use- or management-induced changes in soil quality (Soon *et al.* 2007, Sequeira *et al.* 2011). The particle-size fractions (sand, silt, and clay) of the soil and their relationships to the C pool have also been widely used to determine the dynamics and turnover of SOM and the underlying mechanisms under various land-use and climate changes (Christensen 2001, Grandy and Robertson 2007, He *et al.* 2012).

The Horqin Sandy Land of northern China (42°41'N to 45°15' N, 118°35'E to 123°30'E, Fig. 1) is one of the most seriously desertified areas in China's agro-pastoral ecotone due to its fragile ecology and harsh environment, combined with unsustainable management of the land. Mobile sand dunes are common as a result of a loss of vegetation cover and the resulting erosion. The loss of organic C in this region caused by desertification during the 20th century is estimated at 101.6 Mt from the soil and 5.91 Mt from the vegetation (Zhou *et al.* 2008). To prevent the expansion of desertification, a range of effective practices such as planting of indigenous shrubs, reintroducing tree species, and establishing grazing exclosures have widely been implemented in the Horqin Sandy Land in recent decades. Thus, the potential for increasing C sequestration in the plants and soils by adopting these practices has aroused increasing attention (Su and Zhao 2003, Li *et al.* 2012, 2013, Zuo *et al.* 2013). However, limited data (Chen *et al.* 2012) are available on the responses of soil C to these restoration efforts in the different soil particle-size and SOM density fractions, particularly for two of the key restoration practices (grazing exclusion and planting) in this semi-arid degraded area.

The objectives of the present study were (1) to investigate the accumulation of organic C in the bulk soil and in its various fractions (based on particle size and SOM density), and (2) to demonstrate the allocation pattern of the C fractions caused by rehabilitation of this severely desertified area.

STUDY AREA

Our study site was located in the southern part of the Horqin Sandy Land, Inner Mongolia, China, near the Naiman Desertification Research Station of the Chinese Academy of Sciences (42°55’52”N, 120°41’56”E, 377 m a.s.l., Fig. 1). The main land-use and cover types in the study area are sand dunes with different degrees of vegetation cover, irrigated cropland that was converted from native grassland, afforested dunes created by planting of trees and shrubs, and grassland with different extents of degradation and desertification. The region has a continental semi-arid monsoon temperate climate regime. The mean annual precipitation is 366 mm, of which 70% falls from June to August, and the mean annual potential evaporation is 1935 mm. The mean air temperature is 6.8°C, with a minimum monthly mean of –13.2°C in January and a maximum monthly mean of 23.5°C in July. The frost-free period ranges from 130 to 150 days. The mean wind speed is 4.3 m s^{-1}, with occasional occurrence of gales with a wind speed ≥ 20 m s^{-1} in winter and spring (Zhao *et al.* 2005). According to the Food and Agriculture Organization of the United Nations soil classification system (FAO 2006), the zonal soils are classified as Kastanozems, but as a result of desertification, the current dominant soils are Arenosols with a coarse texture and a loose structure.

In the present study, we selected six land-use and cover types for sampling:

1. Mobile sand dunes (MSD). This landscape was the dominant feature of the study area in the 1970s as a result of severe grassland desertification, with a very low (<5%) vegetation cover and depleted soil nutrition. The dominant species was the annual forb *Agriophyllum squarrosum*.
2. 12-year grazing exclosure (12EX). Livestock grazing had been excluded from the areas with mobile sand dunes for 12 years at the time of the study (i.e., from 1996 to 2008). The vegetation cover was ≅35%. The dominant species was the perennial shrub *Artemisia halodendron*, accompanied by annual species such as *Bassia dasyphylla*, *Setaria viridis*, and *Euphorbia humifusa*.
3. 27-year grazing exclosure (27EX). Livestock grazing had been excluded from the areas with mobile sand dunes from 1981 to 2008. The vegetation cover was ~60%. The dominant species were the biennial forb *Artemisia scoparia* and the annual forb *Chenopodium glaucum*, and the perennial grass *Cleistogenes squarrosa*.
4. 22-year woodland (22W). The perennial woody vegetation was established in areas with mobile sand dunes using seedlings of Mongolian pine (*Pinus sylvestris* var. *mongolica*) in 1986. The original spacing within and between rows was 1 m × 1.5 m, but the density was 793 trees ha^{-1} at the time of the study.
5. 24-year shrubland (24S). The perennial woody vegetation was established in areas with mobile sand dunes using seedlings of the leguminous shrub *Caragana microphylla* in 1984. The spacing within and between the rows was 1 m × 1.5 m.
6. Non-desertified grassland (NG). This site was open and flat, and had a vegetation cover of more than 75%. The dominant species were the perennial grasses and forbs *Pennisetum centrasiaticum*, *Phragmites communis*, *Leymus secalinus*, and *Melissitus ruthenicus*, and the perennial semi-shrub *Lespedeza davurica*. The soil type at NG was a Kastanozem, whereas that at the other sites was an Arenosol. Images of the landscape of the study area are shown in Fig. 2.

Except for the non-desertified grassland, which local records suggest has never sustained significant degradation, all sites began in the same condition, as mobile sand dunes. Although no data was available on the soil conditions at the six study sites before 1980, soil surveys (Liu *et al.* 1996) suggest that there were no major differences in initial conditions. However, the lack of initial data means that we can only use the mobile sand dunes and the non-desertified site as proxies for those initial conditions. In future research, the current dataset will serve as a true baseline for the changes over time. In summary, the mobile sand dunes represent the most degraded situation, and we used this site as a baseline to evaluate the effectiveness of desertification control measures (i.e.,

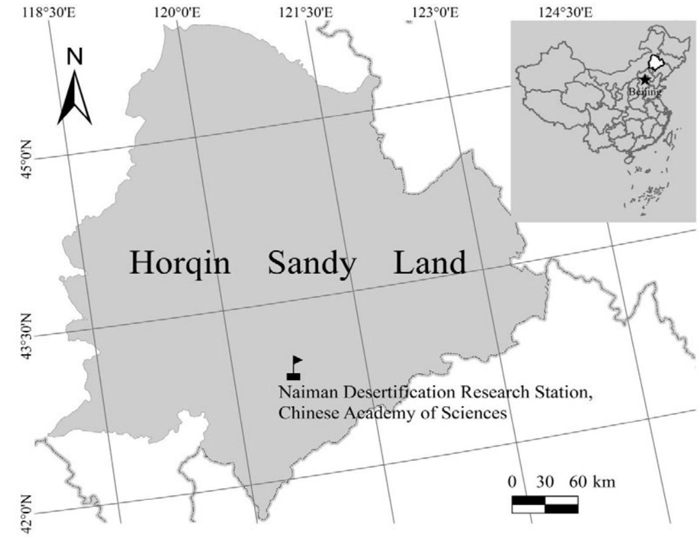

Fig. 1. Location of the study area (the Horqin Sandy Land of Inner Mongolia, China).

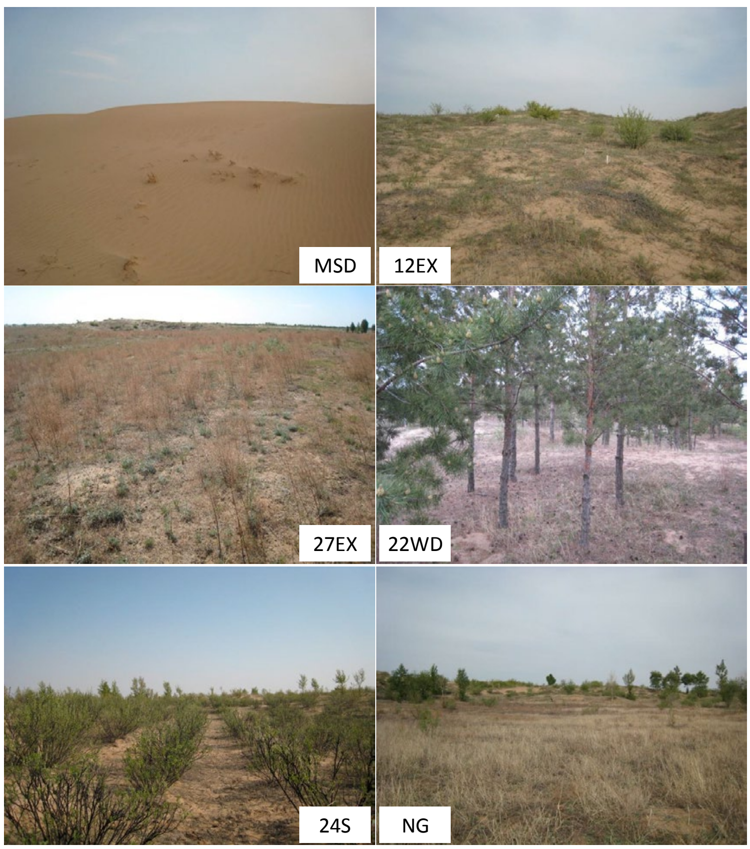

Fig. 2. Typical area of mobile sand dunes (MSD), 12- and 27-year-old grazing exclosures (12EX and 27EX), 22-year-old woodland (22W) planted with Mongolian pine (*Pinus sylvestris* var. *mongolica*), 24-year-old shrubland (24S) planted with a leguminous shrub (*Caraganamicrophylla*), and non-desertified grassland (NG) in Horqin Sand Land, Northern China. All of the exclosure, woodland, and shrubland sites were established in areas with mobile sand dunes. Images are from the authors' own collection, and were obtained in May 2008.

exclosure and plantations of woody species) on SOC sequestration. In contrast, we used the non-desertified grassland to represent the natural situation.

MATERIALS AND METHODS

Experimental design and soil sampling

The soil sampling was conducted in early May 2008. We established three plots (each 30 m × 30 m) in each of the land-use and cover types. All plots, except for those at the NG site (open and flat), faced south, with slopes <15°. Soil samples were collected from six 1 m × 1 m subplots randomly established within each plot, using a stainless-steel cylinder (5 cm in height and 100 cm^3 in volume). Previous research in many ecosystems has shown that C changes primarily near the soil surface after changes in land-use and management practices (De Gryze *et al.* 2004, Shrestha and Stahl 2008), especially in the top 5 cm of soil (Grandy and Robertson 2007). Therefore, we sampled only to a depth of 5 cm. After carefully removing the large surface plant debris by hand, a composite soil sample was prepared using the soil collected to a depth of 5 cm at 10 locations within each subplot. Therefore, within each of the plots, we obtained six composite soil samples, yielding a total of 18 composite samples for each land-use and cover type and 108 samples across all plot types. To determine the soil bulk density, we collected three additional intact soil cores to a depth of 5 cm at each subplot using the same stainless-steel cylinder.

Laboratory analyses

Soil samples were air-dried and hand-sieved through a 2-mm mesh to remove roots and other large debris. No gravel (more than 2 mm in diameter) was found in any of the soil samples. Our preliminary experiment did not detect water-stable aggregates in any of the soils. Therefore, we analyzed the soil particle-size distribution using the dry sieving method. Each soil sample was separated into four fractions using an electronic shaker equipped with a nest of sieves with openings of 2.00, 0.25, 0.10, and 0.05 mm: this system divided the soil into coarse sand+non-water-stable aggregates (NSA; 2.00 to 0.25 mm), fine sand (0.25 to 0.10 mm), very fine sand (0.10 to 0.05 mm), and silt+clay (<0.05 mm).

The LF organic matter was isolated using a modification of the methods described by Janzen *et al.* (1992) and Grandy and Robertson (2007). Approximately 20 g of air-dried bulk soil was weighed into a 100-mL beaker, followed by the addition of 80 mL of aqueous NaI solution at a density of 1.6 g cm^{-3}. The solution was swirled by hand for 30 s, and the content was then dispersed for 1 min using a probe-type sonic disrupter. The beakers were then covered and the suspension was allowed to equilibrate for 48 h at room temperature. The suspended material (the LF) was suctioned onto a Whatman No. 1 filter paper, and washed thoroughly with 5 aliquots of 0.01 M $CaCl_2$ and 10 aliquots of distilled deionized water. After drying at 55 °C for approximately 16 h, the LF was scraped from the filter paper and weighed to the nearest 0.0001 g. The LF dry matter content was expressed as a percentage of the total soil mass. To obtain enough of the LF to carry out the required analyses for the C concentration, it was necessary to extract two to six 20-g portions of each soil.

Subsamples of the air-dried bulk soil and of each particle-size fraction were weighed and dried at 105°C for 24 h to determine the gravimetric water content so that all values could be expressed per unit oven-dry mass. A portion of each bulk soil sample, of the collected LF, and of the soil particles (2.00 to 0.25 mm) was ground to pass through a 0.25-mm mesh before determination of the carbon concentration. The organic C concentrations for the bulk soil, the LF, and each particle-size fraction were determined using the Walkley-Black dichromate oxidation procedure (Nelson and Sommers 1982).

Data analyses

The organic C storage (g m^{-2}) per unit area by volume to a depth of 5 cm in the bulk soil (the SOC storage) and in the LF and the different particle-size fractions (FOC storage) were calculated using the following equations:

$$SOC \text{ storage} = C_S \times BD \times H \times 10 \quad (1)$$

FOC storage = $(P_F \times C_F \times BD \times H)/10$ (2)

where C_s is the bulk SOC (g kg^{-1}), BD is the soil bulk density (g cm^{-3}), H is the soil layer's thickness (= 5 cm), P_F is the proportion (%) of LF (or of each particle-size fraction) dry matter content of the total soil mass, and C_F is the carbon concentration in the LF or in each particle-size fraction (g kg^{-1} dry fraction).

Changes in the land-use and cover types can significantly influence the soil bulk density. As a result, expressing the element storage per unit area as a function of the soil volume can make it difficult to compare the results of different studies. Therefore, we used the normalized soil mass defined by Ellert and Bettany (1995) to estimate the storage per unit area as a function of the equivalent soil mass:

SOC storage* = $(C_S \times EM) \times 10^{-3}$ (3)

FOC storage* = $(P_F \times C_F \times EM) \times 10^{-5}$ (4)

where EM is the equivalent soil mass (g m^{-2}).

Site NG was used as the reference value for the region's natural vegetation. Hence, we estimated the mean soil mass for a 1-m^2 area to a depth of 5 cm at this site, and used this value to normalize the soil mass at the other sites. The value of EM for this site was 6.75×10^4 g m^{-2}.

Data set were subjected to normality tests (Shapiro-Wilk, 5%) and tests of the error homogeneity of variance (Levene, 5%) before further testing. No transformation was required because the data met the assumptions of normality and homogeneity of variance. The measured variables and the resultant C storage were analyzed for the land-use and cover types using one-way ANOVA. When the ANOVA results were significant (P <0.05), we compared the means using the least-significant-difference (LSD) test. Correlations between parameters were calculated using Pearson's correlation coefficient (r). The statistical analysis was performed using version 13.5 of the SPSS software (SPSS, Chicago, IL, USA). Unless otherwise noted, differences were considered statistically significant at P <0.05.

RESULTS

Soil particle-size distributions and bulk density

The proportion of fine sand (0.25 to 0.10 mm) decreased and the proportions of very fine sand (0.10 to 0.05 mm) and silt+clay (<0.05 mm) increased after the establishment of exclosures and plantations in areas with MSD (Table 1). The proportion of coarse sand+NSA (2.00 to 0.25 mm; hereafter, the "coarse fraction") was significantly lower at the 12EX site and higher at the 27EX, 22W, and 24S sites compared to the MSD site.

The 12EX site showed a relatively small effect of short-term grazing exclusion on the soil particle-size distributions. The fine sand and silt+clay contents did not differ significantly between the 12EX and MSD sites. Ex-

Table 1. Soil particle-size distributions and bulk density to a depth of 5 cm at the sites with mobile sand dunes (MSD), 12-year-old grazing exclosure (12EX), 27-year-old grazing exclosure (27EX), 22-year-old woodland (22W), 24-year-old shrubland (24S), and non-desertified grassland (NG).

Land-use and cover types	Particle-size distribution (% of total soil mass)				Bulk density (g cm^{-3})
	Coarse sand+NSA* (2.00-0.25 mm)	Fine sand (0.25-0.10 mm)	Very fine sand (0.10-0.05 mm)	Silt+clay (< 0.05 mm)	
MSD	14.7 ± 7.5 a	82.7 ± 7.0 a	2.1 ± 0.9 a	0.5 ± 0.3 a	1.60 ± 0.02 a
12EX	7.8 ± 2.7 b	75.7 ± 5.8 a	13.7 ± 5.7 b	2.8 ± 2.5 a	1.52 ± 0.06 b
27EX	32.2 ± 2.2 c	41.5 ± 7.6 b	15.7 ± 3.4 bc	10.6 ± 4.4 b	1.46 ± 0.02 bc
22W	26.2 ± 4.3 d	56.8 ± 6.0 c	10.7 ± 2.9 b	6.2 ± 1.7 c	1.47 ± 0.07 bc
24S	21.8 ± 4.4 d	47.9 ± 2.1 b	22.7 ± 5.9 c	7.5 ± 2.9 bc	1.43 ± 0.02 c
NG	10.4 ± 5.2 ab	24.3 ± 9.5 d	57.0 ± 15.2 d	8.3 ± 3.3 bc	1.35 ± 0.12 d

Values (mean ± SD) within a column followed by the same letters do not differ significantly (P <0.05) between land-use and cover types.
*NSA, non-water-stable aggregates.

Table 2. The organic carbon storage in the bulk soil (total SOC), light fraction (LFOC), and the four particle-size fractions at the sites in Horqin Sand Land, Northern China. For explanations of the abbreviations see Table 1 and Fig. 2. Values at the top of the table are based on a given soil volume (to a depth of 5 cm); values at the bottom of the table are based on an equivalent soil mass.

Land-use and cover types	Total SOC	LFOC	Coarse sand+NSA*	Fine sand	Very fine sand	Silt + clay
			C storage per unit area, volume basis (gC m^{-2})			
MSD	49±8a	10±3a	7±4a	37±6a	3±1a	2±1a
12EX	253±38b	37±6 b	56±13b	110±13bd	67±25b	17±15b
27EX	531±241cd	85±55bd	153±87c	161±86c	126±50c	78±30c
22W	406±132c	181±40c	186±69cd	108±35bd	71±22b	41±13de
24S	564±107d	186±113c	224±49d	127±33bc	144±38cd	53±21d
NG	465±48cd	139±34cd	181±32cd	70±21d	181±36d	31±14be
			C storage per unit area, equivalent soil mass basis (gC m^{-2})			
MSD	41±7a	8±2a	6±3a	31±5a	2±1a	2±1a
12EX	226±40b	33±6b	50±12b	98±15bd	60±23b	15±13b
27EX	490±218cd	78±50bd	141±79c	149±78c	116±45c	72±27c
22W	374±126c	168±43c	171±63cd	99±34bd	66±21b	38±13de
24S	530±93d	175±103c	211±44d	119±30bc	135±33cd	50±20d
NG	465±48cd	139±34cd	181±32cd	70±21d	181±36d	31±14be

Values (mean ± SD) for a parameter within a column followed by the same letters do not differ significantly (P <0.05) between land-use and cover types.
*NSA, non-water-stable aggregates.

cept for very fine sand, the contents of the other particle-size fractions differed significantly between the two exclosure durations (12 and 27 years); the coarse fraction and silt+clay content were higher and the fine sand content was lower at the 27EX site. The two plantation types differed significantly for the fine sand content (higher at the 22W site) and the very fine sand content (higher at the 24S site).

Compared with the values at the NG site, only the 12EX site showed a lower value for the coarse fraction content, although the difference was not significant, and the 27EX, 22W, and 24S sites all had a significantly higher coarse fraction content (Table 1). All the five sites showed a significantly higher fine sand content and a significantly lower very fine sand content than the NG site, whereas the 12EX site had a significantly lower silt+clay content. Fine sand was the most abundant particle fraction at all sites except NG, where very fine sand was most abundant. Silt+clay accounted for the smallest proportion of total soil mass at all sites.

The soil bulk density decreased significantly as a result of grazing exclusion and planting of trees and shrubs (Table 1). However, there was no significant difference between two exclosure durations or between the two plantations. The bulk density varied within a narrow range among the sites, and only differed significantly between 12EX (higher) and 24S (lower). The bulk density was significantly lower at the NG site than at the other sites.

Carbon concentrations in the bulk soil and the particle-size and LF fractions

Grazing exclusion and tree or shrub planting significantly influenced the organic C concentration in the bulk soil (Fig. 3). The total SOC concentration increased in the order MSD <12EX <22W <27EX <24S. The total SOC concentration increased to 5.5 to 12.9 times the value of 0.61 g kg^{-1} at the MSD site after grazing exclusion or planting. The level of SOC to a depth of 5 cm in the bulk soil at the 27EX and 24S sites tended to be higher than that at the NG site.

For each particle-size fraction, organic C concentration also increased significantly as a result of grazing exclusion and tree or shrub planting (Fig. 3). The organic C concentration was lowest in the fine sand fraction for all sites, but the particle-size fraction with the highest organic C content differed among the other sites. The C concentration for the other three size fractions was in the following order: at the MSD site, silt+clay > very fine sand > coarse fraction; at the 12EX site, coarse fraction > silt+clay > very fine sand; at the 27EX site, very fine sand > silt+clay > coarse fraction; at the 22W site, coarse fraction > very fine sand > silt+clay; and at the 24S and NG sites, coarse fraction > silt+clay > very fine sand. Across all six plot types, NG had the highest C concentration for the coarse fraction, whereas 27EX had the highest C concentration for fine sand, very fine sand, and silt+clay.

The LF dry matter content increased significantly as a result of grazing exclusion and planting of trees or shrubs, to between 3.1 and 15.5 times the value of 0.08% at the MSD site (Fig. 4A). The highest value occurred at the 22W site, but this site, the 24S site, and the NG site did not differ significantly. The LF dry matter content was significantly positively correlated with the total SOC concentration (Pearson's r = 0.767). The LFOC concentration (Fig. 4B) at the MSD site was significantly lower than at the other sites, with a value of 161 g kg^{-1} (based on the LF dry matter). Among the other plots, the LFOC concentration was lowest at the 12EX site (192 g kg^{-1}) and highest at the 24S site (227 g kg^{-1}), and

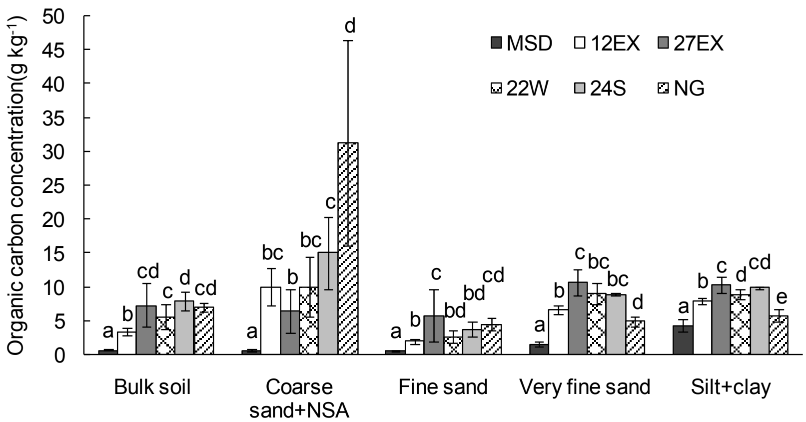

Fig. 3. Organic carbon concentrations in the bulk soil and in the four particle-size fractions (NSA, non-water-stable aggregates) at different sites in Horqin Sand Land, Northern China. For explanations of the abbreviations see Fig. 2. Values represent means ± SD. Bars labeled with different letters differed significantly (P <0.05) between land-use and cover types for each fraction.

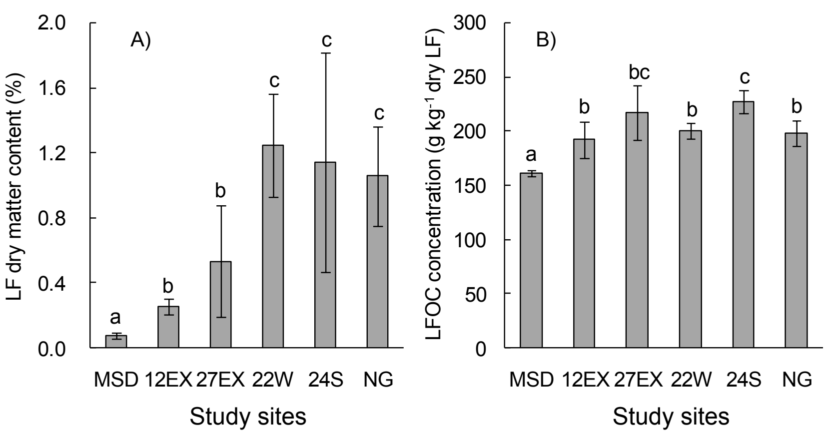

Fig. 4. The values of (A) the light fraction (LF) dry matter content (LF as a % of the total soil mass) and (B) the light fraction organic carbon (LFOC) concentration at different sites in Horqin Sand Land, Northern China. For explanations of the abbreviations see Fig. 2. Values represent means ± SD. Bars labeled with different letters differed significantly (P <0.05) between land-use and cover types.

Table 3. The organic carbon accumulation rate (gC m^{-2} y^{-1}) in the bulk soil (total SOC), light fraction (LFOC), and in the four particle-size fractions following the implementation of grazing exclusion and the planting of trees and shrubs in areas in Horqin Sand Land, Northern China. For explanations of the abbreviations see Table 1 and Fig. 2.

Land-use and cover types	Exclosure			Plantation		
	12EX	27EX	Average	22W	24S	Average
Total SOC	15.4	16.6	16.0	15.1	20.4	17.8
LFOC	2.1	2.6	2.3	7.3	7.0	7.1
Coarse sand+NSA*	3.7	5.0	4.3	7.5	8.5	8.0
Fine sand	5.6	4.4	5.0	3.1	3.7	3.4
Very fine sand	4.8	4.2	4.5	2.9	5.5	4.2
Silt + clay	1.1	2.6	1.8	1.6	2.0	1.8

*NSA, non-water-stable aggregates.

the LFOC concentration at the 24S site was significantly higher than at the 12EX, 22W, and NG sites.

Carbon storage and accumulation rates

We estimated the C storage in bulk soil and in the LF and particle-size fractions (Table 2) using equations (1) to (4). The trends for the volume- and mass-based values were similar, we therefore will only discuss the mass equivalent-based values in the rest of this section. The implementation of grazing exclusion and the planting of trees and shrubs significantly enhanced C storage in the bulk soil and in the LF and each particle-size fractions (Table 2). The total SOC storage at the 27EX and 24S sites exceeded the level at the NG site, but the difference was not significant. The C storage generally increased with increasing exclosure duration and plantation age, and most of these differences were significant.

The total SOC accumulation per year decreased in the following order: 24S > 27EX > 12EX > 22W (Table 3). The C accumulation rate in LF was highest at the 22W site and lowest at the 12EX site, whereas the highest value for the coarse fraction was at the 24S site and the lowest was at the 12EX site. For the fine sand, the highest was at the 12EX site and the lowest was at the 22W site,

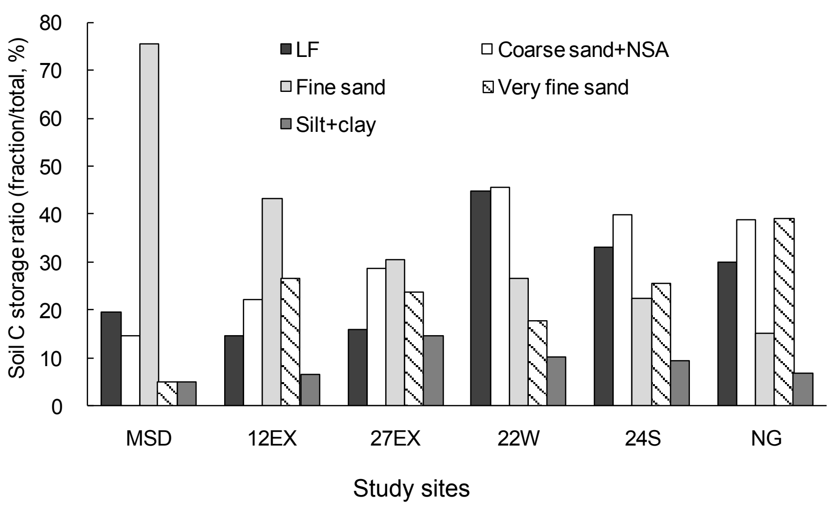

Fig. 5. The proportion of total SOC storage accounted for by C storage in the light fraction (LF) and for each of the particle-size fractions (NSA, non-water-stable aggregates) at the sites in Horqin Sand Land, Northern China. For explanations of the abbreviations see Fig. 2. The calculation was based on the C storage per unit area for an equivalent soil mass.

whereas for very fine sand, the highest was at the 24S site and the lowest was at the 22W site, and for silt+clay, the highest was at the 27EX site and the lowest was at the 12EX site. Based on the averages for exclosure (12EX and 27EX) and for plantation (22W and 24S), the C accumulation rates were greater in plantation than that of exclosure for the bulk soil, LF, and coarse fraction.

The proportion of total SOC storage accounted for by the LFOC (Fig. 5) was lowest at the 12EX site and was highest at the 22W site. The plantation sites had a much higher proportion of LFOC to SOC. At the NG site, this proportion was intermediate between the values for the exclosure and plantation sites. In general, the proportion of total SOC storage accounted for by the C in the coarse fraction, very fine sand, and silt+clay increased after the establishment of the exclosures and plantations, whereas the storage in fine sand decreased (Fig. 5). In addition, the proportion of C storage in the silt+clay changed less than in the other size fractions.

DISCUSSION

Potential of carbon sequestration through grazing exclusion and plantation establishment

The traditional management of rangelands in arid and semi-arid regions (hereafter, "drylands"), which is often associated with a livestock stocking density beyond the carrying capacity, has resulted in changes in the floristic composition and physiognomy, decreases of biomass production and litter inputs, losses of SOC, depletion of soil nutrients, increases in exposed soil, and eventually desertification (Nosetto *et al.* 2006, Sasaki *et al.* 2011, An and Li 2014). Restoring degraded ecosystems by implementing appropriate land-use and management practices could increase their C uptake and storage, and drylands have therefore been regarded as potential C sinks in recent years (FAO 2004, Lal 2009).

Afforestation (tree and shrub planting) and rangeland restoration through grazing exclusion are two potentially important strategies for C sequestration in drylands (Nosetto *et al.* 2006, Malagnoux 2007). These practices can increase ecosystem C pools through the accumulation of plant biomass, can increase biodiversity, and can reduce soil erosion by increasing coverage of the ground by vegetation (Wofsy 2001, Nosetto *et al.* 2006). The low rates of decomposition and soil respiration that occur in a dry environment can also promote C sequestration by the soils of revegetated drylands (Perez-Quezada *et al.* 2011).

Our results indicated that the establishment of grazing exclosures and tree and shrub plantations in the most severely desertified areas of the semi-arid Horqin Sandy Land (areas with mobile sand dunes) increased SOC. The longer the exclosure duration and the older the plantation, the more C accumulated in the soil. The mean accumulation rate for total SOC was higher at the plantation sites than at the exclosure sites (Table 3). After 27 years of grazing exclusion and 24 years of shrub plantation, SOC storage in the top 5 cm of the soil has increased to levels greater than those in non-desertified grassland, although the difference was not statistically significant (Table 2). However, there have been inconsistent results in the research literature for the effects of plantation establishment and grazing exclusion on the SOC pool, primarily due to the complicated interactions among factors such as the regional climate, the duration of previous land-use and subsequent restoration practices, the initial soil structure and type, the composition of the plant community, whether the site had crossed a degradation threshold, and the degree of degradation prior to the implementation of grazing exclusion and plantation establishment (Shrestha and Stahl 2008, Chen *et al.* 2010, Sasaki *et al.* 2011).

For example, Davis and Condron (2002) reported that the conversion of grassland into tree plantations had no net effect on SOC in plantations older than 20 years, because the C loss through increased mineralization may exceed or counteract the C addition due to litter inputs by woody plants. Nosetto *et al.* (2006) compared the C sequestration in semi-arid rangelands with that in *Pinus ponderosa* plantations and grazing exclusion areas in northwestern Patagonia. They found that the SOC storage at the afforestation sites and in the grazing exclosures did not differ from that in adjacent degraded steppes, probably because of slow ecosystem recovery during the short time frame (~15 years) of their study. In contrast, Pei *et al.* (2008) found that SOC increased significantly after only 2 years of grazing exclusion in a desert steppe in the Alxa region of Inner Mongolia, China. In the same region as the present study, Su and Zhao (2003) and Cao *et al.* (2008) also demonstrated that total SOC increased significantly after 5 years in shrub plantations compared with the levels in adjacent mobile sand dunes. It therefore seems that SOC can increase rapidly following the implementation of restoration practices in severely degraded ecosystems, even if soil fertility has decreased drastically.

The plant production and patterns of root allocation as a function of depth in the soil strongly affect the C distribution (Jobbágy and Jackson 2000), and large differences have been detected between the upper and lower soil layers during the restoration of degraded land (Lal 1996, Grandy and Robertson 2007). To evaluate whether soil C responds significantly to tree and shrub planting and grazing exclusion, the soil sampling depth therefore must be taken into account. For example, Noble *et al.* (1999) reported that total SOC increased significantly in the uppermost soil layer (0 to 5 cm) after the conversion of pasture into plantation, but that there was no difference at greater depths (5 to 50 cm). Cunningham *et al.* (2012) also found that tree planting substantially increased total C in the 0 to 5 cm layer, but not for the combined near-surface soil layers (0 to 30 cm). In the present study, to better understand the mechanisms underlying the C accumulation, we only sampled to a depth of 5 cm. In future research, it will be necessary to determine if the results would differ for deeper soil layers.

Effects of land-use and management on carbon in the different soil fractions

Our results indicated that the LFOC represented a relatively large proportion of total SOC storage (ranging from 14.6 to 44.9%) at different site types in the Horqin Sandy Land, compared to the small proportion of LF dry matter content to total soil mass (ranging from 0.08% to 1.24%). Many studies have shown that the LF material, which is composed largely of incompletely decomposed organic residues, can provide a sensitive indicator of the effects of land-use and management practices on SOM. For example, over time frames of 2 to 5 years (Haynes 2000), 4 years (Soon *et al.* 2007), and 6 years (Robles and Burke 1998), the LFOC pool changed significantly due to changes in land-use and management, but the total SOC did not. In the present study, the time frame ranged from 12 to 27 years following the implementation of restoration practices, and storage of both LFOC and SOC increased significantly. Therefore, LFOC was not the only early indicator of changes in SOM in response to grazing exclusion and plantation establishment in our region. However, the significant positive correlation between the total SOC concentration and the LF dry matter content, and the large proportion of total SOC storage accounted for by LFOC, demonstrated that the LF organic matter played a major role in SOC dynamics. In addition, the accumulation of LF was faster in the plantations than in the grazing exclosures, probably due to the greater litter input in the plantations.

Feller and Beare (1997) reviewed the literature and found that the main part of SOM is associated with the silt- and clay-size fractions in most studies that have been conducted in temperate zone soils. Values normally ranged from 10 to 30% of total soil C in the sand-size fraction (> 0.05 mm), 20 to 40% in the silt-size fraction (0.05 to 0.002 mm), and 35 to 70% in the clay-size fraction (< 0.002 mm). Similar results by Christensen (2001) suggested that in temperate arable soils, 50 to 75% of the SOM is present in the clay-size fraction, and that silt accounts for another 20 to 40% of the total, versus < 10% for the sand fraction. In China's black soils region (Liang *et al.* 2009), the C pool in the clay+silt fraction accounted for 75 and 82% of total SOC pool in non-cultivated and cultivated soil, respectively. In contrast, the sand fraction of tropical soils retained a much higher proportion of the total soil C than in the temperate soils, with values ranging from 29 to 64% (Bernhard-Reversat 1981). In the present study, the sand fraction (> 0.05 mm) contained from 83 to 95% of the total SOC storage. The highest value occurred in mobile sand dunes and the lowest in the 27-year-old exclosure.

Many studies have investigated the effects of land-use on SOM and its association with various particle-size fractions. Under vegetated fallow land, the increase in SOC content resulted mainly from an accumulation of particulate organic matter in the sand fraction in sandy soils and to an accumulation of C in the sand and clay fractions in clayey soils (Feller and Beare 1997). In typical steppe ecosystems of Inner Mongolia, He *et al.* (2012) found that C storage mainly increased in the sand and silt fractions in the topsoil (0 to 10 cm) in response to grazing exclusion. In China's Horqin Sandy Land, Chen *et al.* (2010) reported that C in all particle-size fractions (except clay) initially decreased after afforestation of grassland and subsequently increased as the forest matured. In the present study, the C concentration in all particle-size fractions increased rapidly after implementation of the restoration practices, but the C accumulation in the sand fraction was faster than that in the silt+clay fraction. The grazing exclosures and plantations had higher average accumulation rates for the sand fraction than that for the silt+clay fraction. In other studies (e.g., Chen *et al.* 2010, He *et al.* 2012), the C concentrations were significantly higher in the silt+clay fraction than in the sand fraction. Our results are consistent with this trend in all plot types except for the non-desertified grassland, which had a higher C concentration in the sand fraction than in the silt+clay fraction. The differences in the C concentration between the sand and silt+clay fractions were lower in the grazing exclosures and plantations than in the mobile sand dunes.

CONCLUSIONS

In the most severely desertified areas of China's Horqin Sandy Land, the implementation of grazing exclusion and the establishment of tree and shrub plantations significantly increased the carbon stocks in the bulk soil and in each particle-size and density fraction. The plantations showed greater C accumulation than the exclosures in the bulk soil, light fraction, and coarse fraction, but lower accumulation in the fine sand and very fine sand fractions and equal accumulation in the silt+clay fraction. The carbon in the bulk soil and in each particle-size and density fraction increased with increasing exclosure duration and plantation age. However, more comprehensive studies (e.g., extending the soil sampling depth to 50 cm and including measurements of plant biomass) should be conducted, and the study period should be extended, to provide a more complete understanding of C sequestration at our study site.

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