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Influences of grazing and exclosure on carbon sequestration in degraded sandy grassland, Inner Mongolia, north China

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Abstract Livestock grazing is recognised as one of the main causes of vegetation and soil degradation/desertification in the semi-arid Horqin sandy steppe of northern China. In this paper, soil-plant system carbon (C) in a representative degraded sandy grassland in the Horqin sandy steppe (42°58′ N, 120°42′E altitude c. 360 m a.s.l.) was measured. Three situations: long-term continuous grazing (CG), exclosure for 5 years (5EX), and exclosure for 10 years (10EX), were compared to assess the effect of grazing management on C sequestration. Ground cover increased from the CG (35%) to the 5EX (63%) and to the 10EX (81%), and accordingly soil organic C at 0–15 cm depth and total plant components C increased from the CG (492 and 98 g m^{-2}) to the 5EX (524 and 134 g m^{-2}) and to the 10EX (584 and 317 g m^{-2}). The results suggested that continuous grazing in the erosion-prone sandy grassland is very detrimental to vegetation and soil. Under exclosure conditions, vegetation restoration and litter accumulation significantly increased plant-soil system C storage, and thus sequestration of atmospheric C. It was concluded that the degraded sandy grassland could contribute to significant C sequestration with the implementation of protective practices.

Keywords carbon sequestration; grazing; exclosure; sandy grassland; north China

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INTRODUCTION

Grazing lands comprise the largest and most diverse single land resource in the world, and as such represent an important component of terrestrial C cycling and sequestration (Reeder & Schuman 2002). In recent years, the magnitude and distribution of C stored in grazing lands, and the effects of grazing management on the ecosystem processes that control C cycling and distribution in grassland ecosystems have received extensive attention (Bruce et al. 1999; Abril & Bucher 2001; Reeder & Schuman 2002; Schuman et al. 2002). Although current literature suggests no clear relationships between grazing management and C sequestration, the general conclusions concerning the impact of grazing on grassland ecosystem C can be drawn from some studies (Frank et al. 1995; Manley et al. 1995; Franzluebbers et al. 2000; Abril & Bucher 2001; Wienhold et al. 2001). Where the vegetation cover and production capacity of grasslands are not adversely affected by grazing, there is little change of soil organic C. However, in areas where overgrazing has seriously degraded vegetation cover and primary production, soil C will be lower due to increased erosion losses and reduced C inputs.

Sandy grasslands occupy a large area of native grassland of China. Over 1.34×10^8 ha of sandy grasslands are distributed in the arid and semi-arid regions of north China (Zhang et al. 1998). The response of sandy grasslands to the disturbance from anthropogenic activities and variation of climate are very sensitive due to their greater vulnerability. In recent decades, more and more attention has been paid to the severe deterioration/desertification of sandy grassland and its influence on the environment (Li et al. 2000; Wang 2000). However, research quantifying the magnitude and distribution of C stored in these lands is lacking.

Horqin sandy steppe, a typical sandy grazing land located in the semi-arid region of north China (42°41′–45°15′N, 118°35′–123°30′E, elevation 180–650 m), is one of the most severe desertification regions in China (Zhu & Chen 1994). Here the effect of overgrazing on grassland productivity, vegetation cover degradation, and wind erosion is very great and leads to the expansion of the desertification processes (Zhao et al. 2000). In recent decades, projects of desertification control including grazing management have been implemented, and a series of research works concerning vegetation dynamics is ongoing. However, the effects of grazing management on the biogeochemical processes that control the exchange of C between the soil and atmosphere are not well understood in this region. The objective of this study was to quantify the impacts of grazing and exclosure on C sequestration in the degraded sandy grassland ecosystem of the erosion-prone region.

MATERIALS AND METHODS

Study site

The study was conducted at the Naiman Desertification Research Station (NDRS), Academic Sinica, located in the centre of Naiman County, Inner Mongolia, China (42°58′N, 120°43′E). The topography of this area is characterised by sand dunes alternating with gently undulating interdunal lowlands (altitude c. 360 m a.s.l.), with 20–120 m thickness of sandy deposits. It has a semi-arid continental temperate monsoon climate, with windy and dry winters and springs, and warm and comparatively rain-rich summers followed by short and cool autumns. The average annual temperature is 6.8°C with monthly averages ranging from a minimum of –13.1°C in January to a maximum of 23.7°C in July. The 40-year mean annual precipitation is 360 mm with 70% occurring between July and September. In the last 10 years, annual rainfall was 376 ± 88 mm from 1993 to 1997 and 295 ± 94 mm from 1998 to 2002, showing a strong annual variability. Average annual wind velocity varies between 3.4 and 4.1 m s^{-1} with frequent gales (wind speeds > 20 m s^{-1}) in winter and spring. The zonal soils are identified as degraded sandy Chestnut soils according to the Chinese soil classification system (Chinese Soil Taxonomy Cooperative Research Group 1995), which are mostly equivalent to the Orthi-Sandic Entisols of sand origin in terms of the FAO-UNESCO system. These soils are characterised by their coarse texture and loose structure with high proportion of sand (85–95%) and low organic matter content (0.15–0.5% organic C) in the top horizon, and are highly susceptible to wind erosion (Li 1996). The degraded sandy grassland is covered by weed communities and generally dominated by psammophytes including some grasses (e.g., *Cleistogenes squarrosa* (Trin.) Keng, *Setaria viridis* (L.) Beauv., *Phragmites australis* Trin. ex Steudel Nomencl., *Digitaria ciliaris* (Rotz) Koeler, *Leymus chinensis* (Trin.) Tzvel., *Pennisetum centrasiaticum* Tzvel.), forbs (*Mellissitus ruthenicus* (L.) C.W.Chang, *Salsola collina* Pall., *Corispermum elongatum* Bge. ex Maxim., *Agriophyllum squarrosum* (L.) Moq., *Artemisia scoparia* Waldst. et Kit.), shrubs (e.g., *Caragana microphylla* Lam., *Lespedeza davurica* (Laxm.) Schindl.), and subshrubs (e.g., *Artemisia halodendron* Turcz ex Bess., *Artemisia frigida* Willd.).

The field site was an open, flat, degraded, sandy grassland with an area of 50 ha, belonging to the long-term observation site of NDRS. It was subjected to disorganised and continuous livestock grazing and had been slightly desertified in the early 1990s, but the characteristics of soil environment and vegetation cover was relatively homogeneous. The dominant plant species were grasses, accompanied by some legumes and forbs; shrubs and subshrubs were few (Li et al. 2000). At that time, a restoration project was initiated. The exclosures were established gradually and all grazing by domestic herbivores was gradually excluded, allowing the natural vegetation to recover.

Sampling design

Three sites along a gradient of non-grazed restoration time were selected for sampling: (1) 10EX, non-grazed exclosures for 10 years; (2) 5 EX, non-grazed exclosures for 5 years; and (3) CG, areas outside exclosure with continuous grazing.

Field sampling and investigation

In August 2002, three sampling plots (200 × 50 m) were located within each site. Two parallel 200-m transects 20 m apart were established within each plot. Soil and plant samples were collected at peak standing crop at 20-m intervals along each 200-m sampling transect (60 per treatment). The above-ground plant component was sampled by clipping a 0.5-m^2 quadrant at each sampling point along each transect, and partitioning plant phytomass into surface litter, standing dead material, and live biomass by plant species. All plant litter was removed from the soil surface before soil samples were collected to 15-cm depth using a soil auger. Separate cores (7.5 cm diameter) at 0–15 cm were collected at each sampling point to assess root biomass and litter mass incorporated into soils and their C content. Duplicate soil cores (stainless steel cylinder with a volume of 100 cm^3) at 0–15 cm were also taken at each sampling point for soil bulk density determination.

In addition, three randomly located 10-point transects in a 100-m length in each plot were used to determine percent bare ground, live vegetation, and dead vegetation. Standing litter and fallen litter heights of each major plant species or group were measured at 30 randomly located points in each plot (90 points per treatment).

Although the lack of true replication of exclosure treatments in this study restricts inference regarding the impact of sandy grassland management on a large scale in the Horqin steppe, it does not preclude the general comparison of C distribution under continuous grazing and non-grazed exclosure.

Laboratory analyses

Soil samples were passed through a 2-mm screen to remove plant crowns and visible roots and other debris. Particle size analysis was done by pipette method (ISSCAS 1978). Then, subsamples were air-dried and finely ground to pass a 0.25-mm sieve and analysed for organic C by the Walkley-Black dichromate oxidation procedure (Nelson & Sommers 1982). The measurement of roots and litter incorporated into soil was accomplished by hand washing. The separated litter from root cores contained a small amount of root hairs that could not be distinguished. Each plant component was dried at 50°C, weighed, and ground. Then, plant samples taken from three adjacent sample points were mixed thoroughly within each category and analysed for organic C content using the same method as the soil samples (20 per treatment within each category).

Data analyses

Because the exclosures are at one location and not replicated, we have followed the approach of Frank et al. (1995) and considered each of the three plots a replication of summary statistics. Values from all sampling points within each plot were averaged. Then, one-way analysis of variance (ANOVA) procedures were used to detect the differences in parameters examined between sites. The least significant difference (LSD) was performed to determine the significance of treatment means at $P < 0.05$.

RESULTS

Surface characteristics

Soil surface characteristics and plant cover for each site are presented in Table 1. Live vegetation and litter cover was highest in 10EX, intermediate in 5EX, and lowest in CG. After exclosure for 10 years, bare ground decreased by 3.4 times compared with the grazed site. Standing and fallen litter heights also increased with increased restoration time. Standing litter was 2.7–3.4 times higher in 5EX and 10EX than in the grazed site, and fallen litter was 3–5.7 times higher. Also, changes of plant species occurred. Dominant plant species in the CG site were some forbs including *Artemisia scoparia* and *Salsola collina*. In the 5EX site, major species were *A. scoparia*, *S. collina*, the annual grass *Setaria viridis*, and the legume *Lespedeza dahurica*. In the 10EX, some annual and perennial grasses (*S. viridis*, *Cleisogenes squarrosa*, and *Phragmites australis*) and forbs including *A. scoparia*, *S. collina*, and *Chenopodium glaucum* were major species.

Soil particle size distribution and bulk density

Particle size distribution showed more silt and very fine sand and less fine and medium sand in the top 15 cm of soils under the non-grazed exclosures compared with soils under continuous grazing. Comparing the 10EX and 5EX treatments, there was only a slight difference in particle size distribution,

Table 1 Ground cover characteristics at the three grazing treatments. Within ground cover and litter category means ±SD with the same letters are not significantly different ($P < 0.05$). Each treatment × cover component was measured at 90 points. Area that live vegetation and litter overlapped is included in the percent of live vegetation cover.

	Percent ground cover (%)			Height (cm) of litter	
Treatment	Bare ground	Live vegetation	Litter	Standing litter	Fallen litter
10EX	19.2 ± 11.4a	40.8 ± 9.6a	40.4 ± 8.8a	44.8 ± 12.3a	1.7 ± 0.4a
5EX	37.0 ± 14.3b	28.6 ± 6.9b	34.5 ± 10.2a	29.9 ± 5.2b	0.9 ± 0.2b
CG	65.4 ± 4.8c	24.3 ± 3.2b	10.3 ± 2.1b	13.2 ± 4.2c	0.3 ± 0.1c

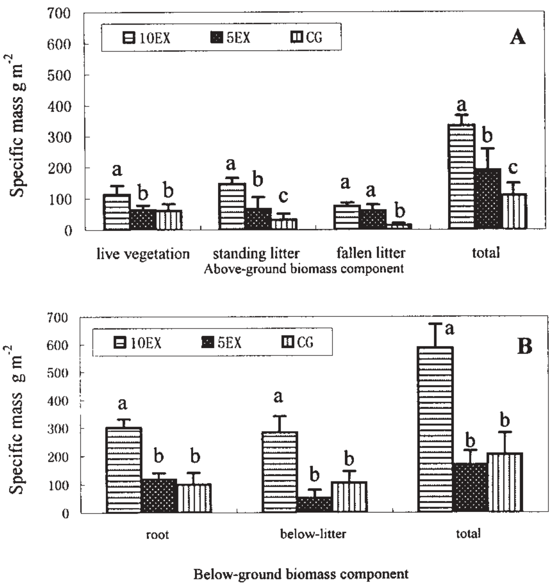

Fig. 1 Specific mass of **A**, above-ground and **B**, below-ground biomass components (0–15 cm) among the three grazing treatments (bar = ±1 SD). For an individual biomass component, treatment means with the same letter are not significantly different ($P < 0.05$, $n = 3$).

with a significant higher silt content and a slight higher very fine sand content in the 10EX site (Table 2). Despite a narrow range (1.39–1.58 g cm^{-3}) due to the homogeneous sandy texture, soil bulk density in the continuous grazed site was significantly higher than that in the 5EX and 10EX sites, with the lowest value being found in 10EX (Table 2). Bulk density was also significantly higher in 5EX than in 10EX ($P < 0.05$).

Above- and below-ground biomass

Live vegetation, various forms of litter, and root biomass were strongly affected by exclosure and grazing practices (Fig. 1). Live vegetation, and standing and fallen litter biomass were lowest in the grazed site, intermediate in 5EX, and highest in 10EX. After exclosure for 5 years, the total amount of above-ground biomass had increased by 78%, and after exclosure for 10 years, this increase was 312% compared with the continuous grazed grassland.

Root biomass and litter mass incorporated into the soil was significantly higher in the 10EX than in the continuous grazed site (Fig. 1). However, in the 5EX treatment, no significant change was found for root biomass ($P = 0.519$), and the amount of litter incorporated into soil showed a slight decrease compared with the grazed site.

Overall, exclosure of degraded sandy grassland resulted in a significant increase in the amount of total organic material including above- and below-ground biomass. Compared with the grazed site, this increase was 2.9 times in the 10EX site (920 g m^{-2}) and 1.6 times in the 5EX site (361 g m^{-2}), respectively.

Organic C in plant components

Carbon concentration in roots, live vegetation, and various forms of litter showed a different variation among different sites (Table 2). Carbon concentration in live vegetation and standing litter were similar in the three study sites. Under continuously grazed conditions, roots had significantly higher C concentrations, and, in contrast, fallen litter and litter incorporated into soil had significantly lower C concentrations than under exclosure conditions. In particular, C concentration in litter incorporated into soil was strongly affected by grazing, being 41–45% lower in the grazed site than in the 5EX and 10EX sites.

Carbon storage per area in plant components showed the same trend as their biomass among treatments (Fig. 2), although C concentrations in plant components varied. Total C storage of plant components in 10EX was on average 317 g m^{-2}, with an average of 133 g m^{-2} above-ground C storage and an average of 184 g m^{-2} below-ground C storage. There were 53% and 69% increases compared with those in 5EX (total: 134 g m^{-2}; above: 78 g m^{-2}; below: 56 g m^{-2}) and in the grazed site (total: 98 g m^{-2}; above: 43 g m^{-2}; below: 55 g m^{-2}), respectively.

Table 2 Particle size distribution and bulk density of soils (0–15 cm) under the different treatments. Within columns, means ± SD with the same letters are not significantly different ($P < 0.05$, $n = 3$).

	Particle size distribution (%)						Bulk density
Site	2–0.5 mm	0.5–0.25 mm	0.25–0.1 mm	0.1–0.05 mm	0.05–0.002 mm	<0.002 mm	(g cm^{-3})
10EX	0.1 ± 0.06a	9.0 ± 5.1b	33.8 ± 7.5b	46.8 ± 7.9a	7.4 ± 2.4a	2.9 ± 1.3a	1.39 ± 0.10c
5EX	0.1 ± 0.04a	14.9 ± 3.1a	36.7 ± 5.6b	39.9 ± 6.7b	5.7 ± 1.1b	2.7 ± 0.9a	1.44 ± 0.09b
CG	0.1 ± 0.06a	17.5 ± 5.8a	46.8 ± 8.4a	30.6 ± 4.0c	3.2 ± 1.2c	2.0 ± 0.7a	1.58 ± 0.05a

Soil organic C

Soil organic C concentration in the 0–15 cm depth was highest in the 10EX site and lowest in the CG site, and there were significant differences among the three sites (Table 3). In the CG site and 5EX site, the spatial variability of soil organic C was higher than that in the 10EX site. When C content was expressed as on an areal basis, C storage in the 0–15 cm depth exhibited a similar pattern to that of C concentration, but the differences were narrowed (Fig. 2). A significant difference for organic C storage was observed between the 10EX site and the CG site (P = 0.013), however, there were no statistically significant differences between 10EX and 5EX (P = 0.067) and between 5EX and CG (P = 0.26).

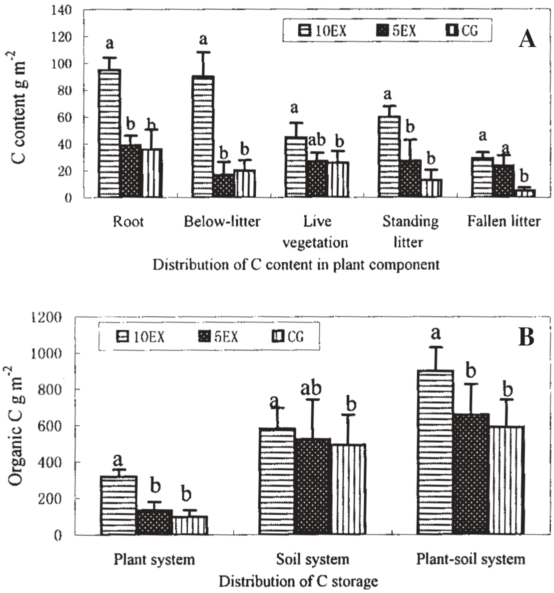

Fig. 2 Distribution of **A**, C storage in components of plant system and **B**, total C storage in plant-soil system among the three grazing treatments (bar = ±1 SD). For an individual component, treatment means with the same letter are not significantly different (P < 0.05, n = 3).

DISCUSSION

Livestock grazing is often regarded as one of the main causes of vegetation and soil degradation/desertification in the semi-arid Horqin sandy steppe (Li et al. 2000; Zhao et al. 2000). Vegetation cover, primary productivity, and area of bare soil are often used to assess the spatial extent and degree of desertification (Zhu & Wang 1992; de Soyza et al. 1998). The present study indicated that bare ground increased considerably, while standing and fallen litter and live vegetation cover decreased under continuous grazing. As a consequence of increased bare ground, more of the soil surface was exposed directly to wind erosion under strong winds due to the decrease in surface roughness (Li et al. 2000). Further, reduced live vegetation and standing litter height also accelerated wind erosion in winter and spring. Although wind erosion was not measured in this study, it was confirmed by the characteristics in soil particle size distribution, with a lower content in fine size fractions (<0.1 mm) in the CG site (36%) than in the 5EX site (48%) and in the 10EX site (57%) (Table 2).

Continuous grazing resulted in a significant increase in soil bulk density. The change of bulk density is considered as an important early indicator of ecosystem degradation (Rubio & Bochet 1998), because it leads to further alteration of soil properties related directly to plant growth, such as soil water infiltration and retention (Salihi & Norton 1987). Increased bulk density could feed back to other soil-plant processes that continue to deteriorate the system (Manzano & Návar 2000). In this study, higher bulk density in the grazed site presumably resulted from the compaction due to livestock trampling. Also, it resulted indirectly from a decrease in the below-ground biomass.

Continuous grazing decreased ecosystem C sequestration. Vegetative cover and litter accumulation decreased under continuous grazing conditions. The increase of bare ground and

Table 3 Organic C concentrations in plant components and soils (0–15 cm) in the three study sites. Within category means ± SD with the same letters are not significantly different (P < 0.05, n = 3).

Site	Organic C (g kg^{-1}) Roots	Live vegetation	Standing litter	Fallen litter	Litter incorporated into soil	Soil organic C (g kg^{-1})
10EX	314 ± 8b	392 ± 22b	409 ± 17a	390 ± 16a	315 ± 17a	2.79 ± 0.10a
5EX	329 ± 24b	425 ± 9a	409 ± 8a	389 ± 14a	333 ± 30a	2.42 ± 0.15b
CG	355 ± 16a	400 ± 14a	400 ± 13a	359 ± 21b	185 ± 18b	2.10 ± 0.18c

reduction of litter height accelerated wind erosion and allowed parts of the litter to be blown away in winter and spring. Trampling by livestock reduced litter particle size and created better litter-soil contact, facilitating more rapid decomposition by soil microorganisms (Shariff et al. 1994) in the grazed site than in the EX sites. All the above factors contributed to a decrease in plant component C storage under continuous grazing. The impact of grazing on soil organic C was attributed in part to the sharp decrease of litter accumulation and input into soil. On the other hand, accelerated wind erosion, due to reduced vegetative cover and litter accumulation, led to the loss of soil organic C because much of the soil organic C in a grassland ecosystem is concentrated near the soil surface where it is also more susceptible to loss or redistribution by wind (Schuman et al. 2002). The data shows that there were significant differences in C storage in various plant components and in soil between 10EX and CG, but no significant differences were found in total below-ground plant C and soil C storage between 5EX and CG (Fig. 2). The lack of statistical differences in C storage between 5EX and CG sites was in part probably explained by the high spatial variation of the parameters examined. On the other hand, consecutive droughts during the recent 5 years of exclosure resulted in a very slow restoration of vegetation despite grazing being excluded. The magnitude of potential for C sequestration was related to the rainfall regime (Lal 2000).

The greatest impact of exclosure on this sandy grassland ecosystem arose from the significant increase of combined live vegetation and cover. This was of practical significance in the erosion-prone Horqin sandy steppe since soil erosion by wind was sharply mitigated in this ecosystem when vegetation cover is over approximately 60% (Zhao 1998). Vegetation restoration under exclosure increased litter accumulation, decreased soil erosion by wind, and trapped wind-blown fine materials and dust. This contributed to C sequestration at an ecosystem level. From the detected differences in total plant-soil system C storage between sites, it was possible to estimate that after 10 years of livestock exclosure, C gain in the plant-soil system was on average 31 g m^{-2} yr^{-1}, and soil C sequestration was 9.2 g m^{-2} yr^{-1}.

In terms of the distribution of C in the plant-soil system, 65, 79, and 83% of plant-soil C was in soil at 0–15 cm depth in the 10EX site, in the 5EX and in the CG site, respectively (Fig. 2). This suggests that although inputs of litter C increased after exclosure, recycling of above-ground plant C to the soil was restricted when grazing was excluded and C was immobilised in plant litter accumulating on the soil surface. With increased exclosure time, a build-up of litter on the surface may decrease soil temperature and soil water content, which will in turn affect plant residue and litter decomposition rates and thus C and nutrient cycling (Naeth et al. 1991; Reeder et al. 2001). Therefore, further study is required on the time scale for non-grazed exclosure.

Lal (2000) suggested that total potential of soil C sequestration through restoration of degraded soils and desertification control in semi-arid regions may range from 30 to 120 kg $ha^{-1}yr^{-1}$. Realisation of this potential requires the development and implementation of appropriate land use and soil/vegetation management practices. Sandy grassland is an important component of land resources in north China and is prone to desertification. Non-grazing exclosure has been recognised as essential to vegetation recovering in degraded sandy land (Zhao 1998). Nowadays, stopping livestock grazing in degraded sandy grassland for protecting and improving eco-environment is being implemented widely in north China. It is expected that exclosure of degraded sandy grasslands on a large scale will potentially act as a substantial sink for atmospheric C.

CONCLUSIONS

Continuous grazing in this erosion-prone sandy grassland is very detrimental to vegetation and soil, resulting in a marked increase in bare ground and a significant decrease in soil-plant system C storage. Vegetation restoration and litter accumulation under non-grazed exclosure increased ground cover, protected soil from wind erosion, and increased soil organic matter, thus sequestering atmospheric C. It was concluded that degraded sandy grassland could contribute to significant C sequestration at ecosystem level with the implementation of restoration practices. From a perspective of ecological restoration and land use as well as C and nutrients recycling which affects ecosystem functioning, there was a need for further research concerning time scale for exclosure.

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REFERENCES

Abril, A.; Bucher, E. H. 2001: Overgrazing and soil carbon dynamics in the western Chaco of Argentina. *Applied Soil Ecology 16*: 243–249.

Bruce, J. P.; Frome, M.; Haites, E.; Janzen, H.; Lal, R.; Paustian, K. 1999: Carbon sequestration in soils. *Journal of Soil and Water Conservation 54*: 382–389.

Chinese Soil Taxonomy Cooperative Research Group, Institute of Soil Science, Academic Sinica, 1995: Chinese soil taxonomy (revised proposal). Beijing, China, Chinese Agricultural Science and Technology Press.

de Soyza, A. G.; Whitford, W. G.; Herrick, J. E.; Van Zee, J. W.; Havstad, K. M. 1998: Early warning indicators of desertification: examples of tests in the Chihuahuan Desert. *Journal of Arid Environments 39*: 101–112.

Frank, A. B.; Tanaka, D. L.; Hofmann, L.; Follett, R. F. 1995: Soil carbon and nitrogen of Northern Great Plains grasslands as influenced by long-term grazing. *Journal of Range Management 48*: 470–474.

Franzluebbers, A. J.; Stuedemann, J. A.; Schomberg, H. H.; Wilkinson, S. R. 2000: Soil organic C and N pools under long-term pasture management in the Southern Piedmont USA. *Soil Biology & Biochemistry 32*: 469–478.

Institute of Soil Sciences, Chinese Academy of Sciences (ISSCAS) 1978: Physical and chemical analysis methods of soils. Shanghai, Shanghai Science Technology Press. Pp. 7–59. (In Chinese)

Lal, R. 2000: Carbon sequestration in drylands. *Annals of Arid Zone 39*: 1–10.

Li, S. G. 1996: Influences of environmental factors on sandy land vegetation. *In*: Liu, X. M.; Zhao, H. L.; Zhao, A. F. *ed*. Wind-sandy environment and vegetation in the Horqin sandy land. Beijing, China, Science Press. Pp. 217–294. (In Chinese)

Li, S. G.; Harazono, Y.; Oikawa, T.; Zhao, H. L.; Chang, X. L. 2000: Grassland desertification by grazing and the resulting micrometeorological changes in Inner Mongolia. *Agricultural and Forest Meteorology 102*: 125–137.

Manley, J. T.; Scheman, G. E.; Reeder, J. D.; Hatt, R. H. 1995: Range land soil carbon and nitrogen responses to grazing. *Journal of Soil and Water Conservation 50*: 294–298.

Manzano, M. G.; Návar, J. 2000: Process of desertification by goats overgrazing in the Tamaulipan thornscrub (*matorral*) in north-eastern Mexico. *Journal of Arid Environments 44*: 1–17.

Naeth, M. A.; Bailey, A. W.; Pluth, D. J.; Chanasyk, D. S.; Hardin, R. T. 1991: Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland ecosystems of Alberta. *Journal of Range Management 44*: 7–12.

Nelson, D. W.; Sommers, L. E. 1982: Total carbon, organic carbon and organic matter. *In*: Page, A. L.; Miller, R. H.; Keeney, D. R. ed. Methods of soil analysis. Part 2. 2nd ed. *Agronomy 9*: 539–577.

Reeder, J. D.; Franks, C. D.; Milchunas, D. G. 2001: Root biomass and microbial processes. *In*: Follett, R. F.; Kimble, J. M.; Lal, R. *ed*. The potential of US grazing lands to sequester carbon and mitigate the greenhouse effect. Boca Raton, FL, Lewis Publishers. Pp. 139–166.

Reeder, J. D.; Schuman, G. E. 2002: Influence of livestock grazing on C sequestration in semi-arid mixed-grass and short-grass rangelands. *Environmental Pollution 116*: 457–463.

Rubio, J. L.; Bochet, E. 1998: Desertification indicators as diagnosis criteria for desertification risk assessment in Europe. *Journal of Arid Environments 39*: 113–120.

Salihi, D. O.; Norton, B. E. 1987: Survival of perennial grass seedling under intensive grazing in semi-arid rangelands. *Journal of Applied Ecology 24*: 145–151.

Schuman, G. E.; Janzen, H. H.; Herrick, J. E. 2002: Soil carbon dynamics and potential carbon sequestration by rangelands. *Environmental Pollution 116*: 391–396.

Shariff, A. R.; Biondini, M. E.; Grygiel, C. E. 1994: Grazing intensity effects on litter decomposition and soil nitrogen mineralization. *Journal of Range Management 47*: 444–449.

Wang, T. 2000: Land use and sandy desertification in north China. *Journal of Desert Research 20*: 103–108.

Wienhold, B. J.; Hendrickson, J. R.; Karn, J. F. 2001: Pasture management influences on soil properties in the Northern Great Plains. *Journal of Soil and Water Conservation 56*: 27–34.

Zhang, Q.; Zhao, X.; Zhao, H. L. 1998: Grasslands in desert areas in China. Beijing, Meteorological Press. Pp. 1–2. (In Chinese)

Zhao, H. L. 1998: Exclosure results and its evaluation of degenerated grassland in Horqin sandy land. *Journal of Desert Research 18*: 46–50. (In Chinese with English abstract)

Zhao, H. L.; Zhao, X. Y.; Zhang, T. H. 2000: Causes, processes and countermeasures of desertification in the interlocked agro-pastoral area of north China. *Journal of Desert Research 20(Suppl.1)*: 22–28. (In Chinese with English abstract)

Zhu, Z. D.; Chen, G. T. 1994: Sandy desertification in China. Beijing, Science Press. 250 p. (In Chinese)

Zhu, Z. D.; Wang, T. 1992: Theory and practice on sandy desertification in China. *Journal of Quaternary Science 2*: 97–106. (In Chinese with English abstract)