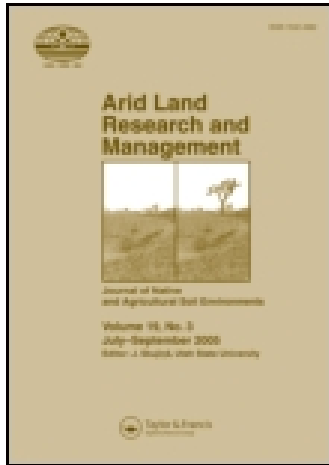


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Arid Land Research and Management

Publication details, including instructions for authors and subscription information:

<http://www.tandfonline.com/loi/uasr20>

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Published online: 24 Jun 2013.

To cite this article: Ren-tao Liu , Ha-lin Zhao , Xue-yong Zhao & Fan Zhu (2013) Effects of Cultivation and Grazing Exclusion on the Soil Macro-faunal Community of Semiarid Sandy Grasslands in Northern China, *Arid Land Research and Management*, 27:4, 377-393, DOI: [10.1080/15324982.2013.787470](https://doi.org/10.1080/15324982.2013.787470)

To link to this article: <http://dx.doi.org/10.1080/15324982.2013.787470>

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Effects of Cultivation and Grazing Exclusion on the Soil Macro-faunal Community of Semiarid Sandy Grasslands in Northern China

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Cultivation and overgrazing can cause desertification of sandy grasslands in arid and semiarid regions. However, little was known about their effects on soil macro-fauna. In this study, we described the soil macro-faunal community structure at cultivated, grazed, and ungrazed sites in the semiarid Horqin steppe, northern China. Soil bulk density was 1.61 g cm^{-3} at grazed sites, similar to that at cultivated sites. Cultivation reduced soil organic carbon and total nitrogen by 36 and 46%, respectively, compared with grazed grassland. Grazing exclusion decreased soil bulk density and pH by 16 and 3%, respectively, and increased soil moisture, soil organic carbon and total nitrogen by 75, 22, and 4%, respectively, compared with grazed grassland. Cultivated and grazed sites had similar group richness (9 and 10 groups, respectively) and Shannon's index (1.98 and 1.93, respectively); both were significantly lower there than at ungrazed sites. Simpson's index at grazed sites was 0.18, markedly higher than at ungrazed (0.12) and cultivated (0.16) sites. There were no significant differences in soil macro-faunal density among the three treatments, although the density at grazed sites was 24% higher than at cultivated sites and 30% lower than at ungrazed sites. Soil bulk density, electrical conductivity, pH, and the C/N ratio significantly affected the community structure. Specific macro-faunal groups responded to the different living conditions by adaptation to or selection of specific habitats. For example, the Thomisidae, Reduviidae, Staphilinidae, and Curculionidae responded negatively to increasing soil bulk density and all of these except the Curculionidae responded negatively to increasing soil temperature. These groups have important implications for the soil food web, and thus for the soil ecosystems. Our results suggested that cultivation and continuous

Received 5 February 2012; accepted 15 March 2013.

We thank Prof. Tóth, the journal editor, and the reviewers for their insightful comments and efforts to improve this manuscript. We also thank Geoffrey Hart (our English editor from Canada) for his assistance in English editing of the manuscript. This study was financially supported by projects of the National Natural Science Foundation of China (No. 41101050), the National Basic Research Program of China (No. 2009CB421303), and the National Science and Technology Support Program (2011BAC07B03).

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grazing negatively affected soil properties and macro-faunal diversity, whereas livestock exclusion enhanced soil macro-faunal assemblies and improved their biodiversity. Changes in these assemblies might be potential indicators of ecosystem recovery and management impacts. Supplementary materials are available for this article. Go to the publisher's online edition of Arid Land Research and Management to view the supplemental file.

Keywords community structure, cultivation, enclosure, grazing, Horqin Sandy Land, soil macro-fauna

The grassland soils of arid and semiarid regions in northern China and other parts of the world, are vulnerable to degradation caused by cultivation and overgrazing (Li et al., 2000; Li et al., 2007; Abril and Bucher, 2001; El Mouden et al., 2006). Unsustainable cultivation and overgrazing can decrease vegetation cover and soil organic matter content, and degrade the surface soils (Zida et al., 2011). Consequently, these land uses can make the land more susceptible to desertification through topsoil erosion, thus decreasing biodiversity and degrading the grassland ecosystem (Su et al., 2004).

Many authors have discussed the impacts of cultivation, overgrazing, and grazing management on degraded grassland ecosystems in semiarid and arid regions (Li et al., 2007; Zhao et al., 2007b). Grassland cultivation typically causes a degradation of soil physical, chemical, and biological properties (Saggar et al., 2001; Zhao et al., 2007a), and can lead to permanently decreased land productivity and a degraded ecosystem (Pei et al., 2008). Overgrazing can decrease soil organic C by reducing plant cover and increasing soil respiration (Cao et al., 2004; Zhao et al., 2005), and can weaken aggregate stability (resulting in increased soil erosion) through organic matter depletion and the effects of trampling by livestock (Manzano and Nívar, 2000). In contrast, livestock exclusion can promote the recovery of degraded grassland vegetation, resulting in remarkably improved surface soil properties (Su et al., 2004; Pei et al., 2008). However, no quantitative research has described the effects of cultivation, overgrazing, and grazing exclusion on the soil faunal community in these regions.

Soil fauna play important ecological roles as components of food chains and nutrient cycles, and initiate or maintain key soil processes such as the development of soil structure and the decomposition of organic matter (Coleman et al., 2004). They can improve general soil physical characteristics and soil aggregation (Zida et al., 2011). In addition, biotic interactions (e.g., between soil fauna and microbes) in the soil can regulate the structure and functioning of aboveground communities (Jiang et al., 2007). These interactions between belowground and aboveground ecosystem components strongly influence community- and ecosystem-level processes (Wardle et al., 2004). Thus, there is a great need to assess the effects of management practices on the soil faunal community, so as to provide a scientific basis for the restoration of degraded grassland ecosystems in arid and semiarid regions.

Cultivation and grazing have marked influences on soil faunal populations (Baraibar et al., 2009; Zida et al., 2011; Schon et al., 2012). Soil disturbance by processes such as tillage can destroy or damage nests and burrows, thereby decreasing faunal density (Baraibar et al., 2009). Soil fauna are therefore endangered when former grazing land is converted to agricultural uses (Schon et al., 2012). Livestock grazing not only affects the physical habitats of soil organisms through compaction by their hooves, but also affects processes such as nutrient cycling interspersed with

soil organisms, and can affect populations and community dynamics of these organisms through the effects of dung and urine returned to the soil (Bardgett et al., 2001; Bugalho et al., 2011). The combined effect of removing the pressure of livestock hooves and moisture stress caused by exposure of the soil surface in fenced exclosures can greatly promote the recovery of soil fauna (Schon et al., 2012). In previous research, soil faunal density increased as grazing intensity decreased, particularly after complete grazing exclusion (Liu et al., 2010; Baraibar et al., 2011). However, the changes in the soil macro-faunal community following cultivation, grazing, and grazing exclusion are largely unknown in arid and semiarid sandy grasslands of northern China.

The objective of the present study was to compare soil macro-faunal communities among sites subjected to short-term cultivation, continuous grazing, and grazing exclusion in the semiarid sandy grasslands of northern China. We hypothesized that the soil macro-faunal community composition would become more diverse and complex as vegetation recovery proceeded.

Materials and Methods

Study Site

The study area is in the southwestern Horqin Sandy Land (around $42^{\circ}55'$ N and $120^{\circ}42'$ E, 360 m elevation) of Inner Mongolia, northern China (Figure 1). The region has a temperate continental semiarid monsoonal climate, with annual mean precipitation of 366 mm, of which 70 to 80% falls during the summer growing season; annual mean potential evaporation is 1935 mm. Rainfall shows strong seasonal and annual variability. The annual mean temperature is 6.8°C , ranging from a maximum mean monthly temperature of 21.9°C in July to a minimum of -14.7°C in January, with a frost-free period of 130 to 150 days. The wind velocity exceeds

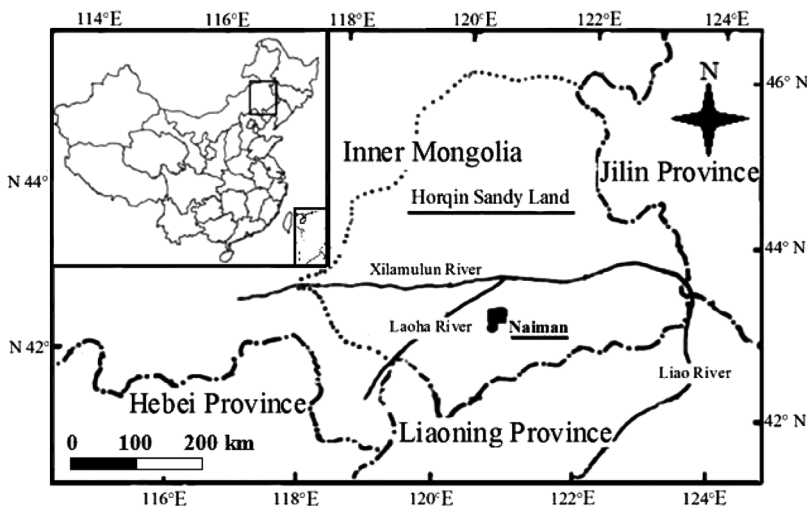


Figure 1. Location of the study site (the Horqin Sandy Land of Inner Mongolia in northern China). The black region indicates the study area near the Naiman Desertification Research Station, Chinese Academy of Sciences.

4 m s^{-1} for >200 days per year (mainly in spring and winter), which is the threshold for wind erosion. Winds stronger than 17.2 m s^{-1} occur for 17 to 24 days per year (Li et al., 2000).

The landscape of this region is characterized by dunes alternating with gently undulating lowland areas (meadow grasslands). The grassland soils are degraded sandy Chestnut soils, equivalent to Orthi-Sandic Entisols in the FAO-UNESCO system (Zhao et al., 2007a). The texture is coarse (85 to 95% sand) and loose, with a low organic matter content (0.2 to 0.5% organic C) and a depth of 20 to 50 cm. The soils are vulnerable to wind erosion and other changes (e.g., accelerated mineralization, leaching, and translocation), particularly after cultivation. The degraded grasslands are covered by grasses (e.g., *Cleistogenes squarrosa* (Trin.) Keng, *Setaria viridis* (L.) Beauv., *Phragmites australis* (Cav.) Trin. ex Steud., *Digitaria sanguinalis* (L.) Scop., *Leymus secalinus* (Georgi) Tzvel., *Pennisetum centrasiaticum* Tzvel.), forbs (e.g., *Melissilus ruthenicus* (L.) Peschkova, *Salsola collina* Pall., *Corispermum elongatum* Bunge, *Agriophyllum squarrosum* (Linn.) Moq., *Artemisia scoparia* Waldst. et Kit.), shrubs (e.g., *Caragana microphylla* Lam., *Lespedeza davurica* (Laxm.) Schindl.), and sub-shrubs (e.g., *Artemisia halodendron* Turcz. ex Bess., *Artemisia frigida* Willd. Sp. Pl.).

The grasslands in this region have been suffering from poorly managed conventional livestock grazing (i.e., overgrazing) for many years. In many places around our study sites overgrazing is still common. Many farmers grazed their sheep on the non-fenced grasslands at a density ≥ 6 sheep equivalents ha^{-1} , which is too high for grasslands under a semiarid climate (Zhao et al., 2005). To monitor the possibility of vegetation recovery after the removal of grazing pressure, a 5.2-ha patch of grassland in the extensively grazed area was fenced in 1996 to create an enclosure, and has been monitored since then as part of a grazing-exclusion experiment (Zhao et al., 2007a). We established grazing exclusion sample plots in this area of grasslands.

In addition, more than half of the original pasture area had been cultivated, but had degraded into sandy land over the last century as a result of unsustainable cultivation practices to meet a high and growing demand for food by the region's increasing population (Zhao et al., 2007a). Cultivation of these grasslands has expanded rapidly in recent years in response to economic development plans based on the use of advanced agricultural technology. Crops such as corn, wheat, and watermelons with a short growth period were planted in this area. Management typically involved a single annual tillage prior to sowing. All crop fields were cultivated to a depth of 15 cm using a horse-drawn mini-mould-board plow, seeds were sown, and then the field was harrowed and rolled. Weeds were removed by hand after the establishment of crop seedlings. Irrigation was provided during the growing seasons. The fields were left uncovered after harvesting, and were therefore highly vulnerable to wind erosion. Large areas of these cultivated grasslands have become moderately to seriously desertified as a result.

Experimental Design

In 2008, adjacent patches of about $200 \text{ m} \times 70 \text{ m}$ were selected in ungrazed and grazed grasslands (see Figure S1 in the online supplementary materials), with three replicates (henceforth, sites) under each condition. The continuously grazed sandy grasslands had a grazing pressure of about 10 sheep equivalents ha^{-1} , and had a

vegetation cover of about 24%. The ungrazed grasslands had been fenced for 12 years, and the vegetation cover had recovered to about 70%. We also selected three cultivated fields that had been planted with corn, and were located within 500 m of the grazed and ungrazed plots; these represented cultivated grasslands, with the area of 150 m × 300 m each. They were managed using the same techniques described above, and weeds were removed manually. The cropland, grazed, and ungrazed grasslands had similar landforms and soil attributes, and the groups of replicates for the three treatments were separated by 0.3 to 1.5 km. Three replicates were used for each land cover type to ensure that the observed differences resulted from the treatment rather than from site differences at the start of the study period.

At each site, we established two 30 m × 30 m sampling plots in each treatment. We randomly selected five points within each plot, and established one quadrat (1 m × 1 m) at these points. In spring (May), summer (July), and autumn (September) of 2008, we excavated a 30 cm × 30 cm × 30 cm soil sample from each quadrat, and recovered all organisms by hand to describe the soil macro-faunal community structure. All organisms that we found in our soil samples were defined as soil macro-fauna (*sensu* Coleman et al., 2004), even if they spent most of their life within the herbaceous layer.

In July 2008, five 200-cm³ soil cores were taken from each plot to determine soil bulk density and water content to a depth of 30 cm. Another five composite samples (to a depth of 30 cm) were taken from each plot to determine soil pH, electrical conductivity, organic carbon content, and soil total nitrogen content. In total, we obtained 30 soil samples for the macro-faunal population (5 quadrats × 2 plots × 3 replicates), and 60 soil samples for analysis of soil physical-chemistry properties (2 samples × 5 quadrats × 2 plots × 3 replicates) for each site type.

Data Collection and Analysis

Soil macro-fauna were stored in 75% alcohol until they could be identified in the laboratory. Identification was to the order and family level, using the keys of Zheng and Gui (1999) and Yin (2001), and organisms were classified into groups (based on families) based on their morphological features under a binocular microscope (40×). To increase our sample size for each taxonomic group sufficiently to permit statistical analysis, we combined the datasets from the three months instead of calculating changes in the macro-faunal community during the growing season.

Soil cores were oven-dried to determine soil bulk density (BD, g cm⁻³) and water content (SW, %). Soil samples were passed through a 2-mm sieve to remove plant parts and stones, and were then air-dried until steady weight. Soil pH and electrical conductivity (EC, μS cm⁻¹) were determined in a 1:1 (v/v) soil-water solution and in a 1:5 (v/v) soil-water aqueous extract, respectively. Soil organic carbon content (SOC, g kg⁻¹) was measured by the Walkley–Black K₂Cr₂O₇-H₂SO₄ oxidation method (Nelson and Sommers, 1982), and soil total nitrogen content (STN, g kg⁻¹) was determined by the Kjeldahl procedure (ISSCAS, 1978). In each quadrat, soil temperature (ST, to a depth of 30 cm, °C) was determined during the experimental period using a portable thermometer (Sato Keiryoki Mfg. Co. Ltd., Tokyo, Japan).

We calculated several diversity indices for the macro-faunal community. First, we determined the abundance of each taxonomic group (the density of each family, defined as the number of individuals m⁻²) and the group richness (number of families). We then described the diversity using Shannon's index (*H*) and Simpson's

index (D).

$$H = - \sum P_i \log_2 P_i, \quad \text{where } P_i = x_i / \sum x_i \quad (1)$$

$$D = P_i^2 \quad (2)$$

where x_i = the number of individuals in group i , and P_i = the proportion of the total number of individuals belonging to group i .

We used redundancy analysis (RDA) to identify the main soil factors that affected the faunal groups (Ter Braak and Šmilauer, 1998). Data were first analyzed by means of detrended correspondence analysis (DCA; Ter Braak and Šmilauer, 1998) using version 4.5 of the CANOCO Software (Microcomputer Power, Ithaca, NY, USA), which suggested that RDA was appropriate for further analysis (i.e., the DCA gradient length (2.41) was <4 for the macro-faunal communities). We used RDA to correlate each faunal group with the environmental variables by selecting the linear combinations of environmental variables with the smallest residual sum of squares (Kennedy et al., 2004). We used inclusive forward selection to identify the factors that explained the most variance in the species group data. Finally, the data was log-transformed to produce a normal distribution, and we used Monte Carlo permutation (999 iterations) for significance testing.

The number of individuals and the group richness did not differ significantly between the plots of a given land cover type, so we believe that the site conditions were similar between the two plots. We pooled the data in the two plots per site to obtain a single value in our statistical analysis. Analysis of variance (ANOVA) followed by multiple comparisons (using least-significant-difference [LSD] tests) was carried out using version 15.0 of SPSS for Windows (SPSS Inc, Chicago, IL, USA). Diversity indices (Shannon's H and Simpson's D) were compared by means of bootstrap methods (Solow, 1993; Magurran, 2004) using version 3.01 of the Species Diversity and Richness software (PISCES Conservation Ltd., Pennington, Lymington, UK). For all tests, statistically significant differences were assigned at $p < 0.05$.

Results

Environmental Parameters

Soil bulk density was 1.61 g cm^{-3} at the grazed sites, which was similar to the values in cropland (Table 1) but was significantly higher than the value at the ungrazed sites ($p < 0.05$). Soil water content in the soils at the grazed sites were 1.70%, and was significantly lower than in the soils of the cultivated and ungrazed sites ($p < 0.05$). Cultivation significantly increased soil pH whereas grazing exclusion significantly decreased soil pH ($p < 0.05$) compared with the value for grazed grasslands. The ungrazed and grazed sites had similar soil organic carbon and total nitrogen contents, but cultivation reduced both by 36 and 46%, respectively, in comparison to the values for grazed grassland.

Soil temperature, electrical conductivity, and the C/N ratio did not differ significantly among the treatments, though soil temperature was slightly higher and the C/N ratio was slightly lower at the grazed sites than at the cultivated and ungrazed

Table 1. Values of the environmental parameters (mean \pm S.E.; $n = 3$) for the three site types

	ST ($^{\circ}$ C)	BD (g cm^{-3})	SW (%)	pH	EC ($\mu\text{s cm}^{-1}$)	SOC (g kg^{-1})	STN (g kg^{-1})	C/N
EXG	23.01 \pm 1.01a	1.36 \pm 0.16b	2.98 \pm 0.14a	7.75 \pm 0.06c	34.89 \pm 1.79a	3.35 \pm 0.24a	0.25 \pm 0.02a	13.46 \pm 0.74a
CGG	25.35 \pm 0.15a	1.61 \pm 0.02a	1.70 \pm 0.10c	7.96 \pm 0.05b	39.56 \pm 1.94a	2.74 \pm 0.23ab	0.24 \pm 0.01a	11.41 \pm 0.83a
CCF	24.71 \pm 0.64a	1.59 \pm 0.05a	2.33 \pm 0.14b	8.25 \pm 0.03a	44.89 \pm 3.87a	1.75 \pm 0.42b	0.13 \pm 0.03b	13.29 \pm 0.27a
<i>F</i>	3.02	9.51*	25.08**	25.79**	0.37	6.82*	8.53*	3.62

Note: Values in a column followed by different letters differ significantly ($*p < 0.05$, $**p < 0.01$; one-way ANOVA followed by post-hoc LSD test). CCF, 3 years of cultivated cropland; CGG, continuously grazed sandy grassland as a control; EXG, ungrazed (exclosure) grassland fenced for 12 years; BD, soil bulk density; C/N, the ratio of soil organic carbon to total nitrogen; EC, soil electrical conductivity; pH, the soil pH (in water); SOC, soil organic carbon; ST, soil temperature; STN, soil total nitrogen; SW, soil water content.

Table 2. Density (individual m^{-2}), group richness (number of families), Shannon's index, and Simpson's index for the soil macro-faunal communities

	EXG	CGG	CCF
Density (mean \pm SE)	92 (± 4)a	71 (± 4)a	54 (± 2)a
Group richness (mean \pm SE)	17 (± 3)a	10 (± 1)b	9 (± 1)b
Shannon's index (mean \pm SE)	2.45 (± 0.02)a	1.93 (± 0.02)b	1.98 (± 0.06)b
Simpson's index (mean \pm SE)	0.12 (± 0.01)b	0.18 (± 0.01)a	0.16 (± 0.01)b

Note: Values of a parameter followed by different letters differ significantly ($p < 0.05$). $n = 3$. CCF, 3 years of cultivated cropland; CGG, continuously grazed sandy grassland as a control; EXG, ungrazed (exclosure) grassland fenced for 12 years.

sites. Soil electrical conductivity at the grazed sites ($39.56 \mu S cm^{-1}$) was between the values for ungrazed ($34.89 \mu S cm^{-1}$) and cultivated ($44.89 \mu S cm^{-1}$) sites.

Soil Macro-faunal Community Structure

The sites differed significantly in the soil macro-faunal community structure (Table 2; Table S1 in the online supplementary materials). The cropland and grazed grassland had similar group richness (9 and 10 groups, respectively) and similar values of Shannon's index (1.98 and 1.93, respectively), but both were significantly lower than those for ungrazed grassland ($p < 0.05$). Simpson's index at grazed sites was 0.18, and this was significantly higher than the values for ungrazed (0.12) and cultivated (0.16) sites ($p < 0.05$). However, the density of individuals did not differ significantly among the treatments, though cultivation decreased the density and grazing exclusion increased the density compared with the value for grazed grassland.

Redundancy Analysis

We used RDA to determine the main factors that affected the soil macro-faunal community structure (Figure 2). Axes 1 and 2 explained 33.2 and 21.3%, respectively, of the overall variance within the soil macro-faunal group data (see Table S2 in the online supplementary materials) accounting for a total of 54.5% of the total variance. The cumulative variance for the species-environment relationship for Axes 1 and 2 was 100%; thus, these axes fully accounted for the variance in the soil macro-faunal group data that could be attributed to environmental factors (here, soil properties). Species-environment correlations (Pearson's r) for both axes were greater than 0.95, indicating that the macro-faunal groups were strongly correlated with the soil parameters. Monte-Carlo tests revealed that both the first axis and all axes combined explained a significant ($p < 0.05$) amount of the variation within the data.

The ordination diagram (Figure 2) can be interpreted quantitatively: the arrow lengths indicate the proportion of the variance explained by a given factor, and arrows for individual environmental factors with similar directions show similar trends (Kennedy et al., 2004). The following parameters showed similar trends: ST and BD (negative values on axis 1 and positive values on axis 2); STN and SOC (positive on both axes); EC and pH (negative on both axes); and SW and C/N (positive on axis 1 and negative on axis 2). Figure 2 and the canonical coefficients and

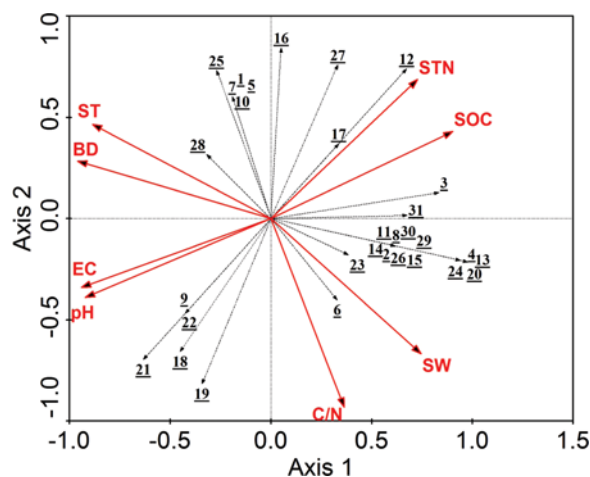


Figure 2. Redundancy analysis (RDA) ordination diagram for the macro-faunal groups, with environmental variables represented by words and macro-faunal groups represented by numbers. BD, soil bulk density; C/N, the ratio of soil organic carbon to total nitrogen; EC, soil electrical conductivity; pH, the soil pH (in water); SOC, soil organic carbon; ST, soil temperature; STN, soil total nitrogen; SW, soil water content. 1 *Leptyphantes*, 2 *Theridiidae*, 3 *Araneidae*, 4 *Thomisidae*, 5 *Lycosidae*, 6 *Gnaphosidae*, 7 *Philodromidae*, 8 *Salticidae*, 9 *Labiduridae*, 10 *Cicadellidae*, 11 *Rhopalidae*, 12 *Lygaeidae*, 13 *Reduviidae*, 14 *Pyrrhocoridae*, 15 *Anthocoridae*, 16 *Miridae*, 17 *Cydniidae*, 18 *Carabidae*, 19 *Pselaphidae*, 20 *Staphylinidae*, 21 *Aphodiidae*, 22 *Rutelidae*, 23 *Tenebrionidae*, 24 *Curculionidae*, 25 *Melolonthidae* larvae, 26 *Rutelidae* larvae, 27 *Tenebrionidae* larvae, 28 *Asilidae* larvae, 29 *Therevidae* larvae, 30 *Noctuidae* larvae, 31 *Formicidae*. (Figure available in color online.)

intraset correlations for the soil factors for axis 1 and axis 2 showed that BD, pH, EC, and the C/N ratio influenced the macro-faunal community structure most strongly. The variability in macro-faunal abundance explained by the two axes was therefore due primarily to BD, pH, and EC for axis 1, and C/N for axis 2, but the responses differed among the factors (Figure 2; supplementary Table S2).

Figure 2 also demonstrates that different groups within the macro-faunal community responded differently to the soil variables (Table 3). The macro-faunal groups whose arrows pointed in approximately the same direction in the ordination as the arrows for the environmental factors were strongly correlated with those factors (Kennedy et al., 2004; Liu et al., 2011). For example, the *Thomisidae*, *Reduviidae*, *Staphylinidae*, and *Curculionidae* were negatively correlated with soil bulk density, and all of these except the *Curculionidae* were negatively correlated with soil temperature. The *Lygaeidae* were negatively correlated with soil pH and EC but positively correlated with SOC and STN. In contrast, the *Carabidae* were negatively correlated with SOC and STN, and the *Melolonthidae* larvae were negatively correlated with the C/N ratio.

Discussion

Cultivation and overgrazing are widely recognized causes of desertification of sandy grasslands in arid and semiarid northern China. These types of land use and

Table 3. Correlation coefficients (Pearson's r) between the density of macro-faunal groups and the soil parameters

Macro-fauna	ST	BD	SW	pH	EC	SOC	STN	C/N
<i>Lepthyphantes</i>	0.449	0.355	-0.546	-0.054	-0.024	0.089	0.289	-0.631
<i>Theritidae</i>	-0.610	-0.631	0.549	-0.519	-0.535	0.498	0.343	0.351
<i>Araneidae</i>	-0.695	-0.757	0.577	-0.744	-0.756	0.727	0.580	0.270
<i>Thomisidae</i>	-0.964**	-0.997***	0.868*	-0.820*	-0.846*	0.787	0.542	0.555
<i>Lycosidae</i>	0.449	0.355	-0.546	-0.054	-0.024	0.089	0.289	-0.631
<i>Gnaphosidae</i>	-0.556	-0.535	0.552	-0.315	-0.336	0.289	0.116	0.455
<i>Philodromidae</i>	0.449	0.355	-0.546	0.573	-0.535	0.089	0.289	-0.631
<i>Salticidae</i>	-0.610	-0.631	0.549	-0.519	-0.535	0.498	0.343	0.351
<i>Labiduridae</i>	0.161	0.276	-0.003	0.573	0.560	-0.587	-0.632	0.280
<i>Cicadellidae</i>	0.449	0.355	-0.546	-0.054	-0.024	0.089	0.289	-0.631
<i>Rhopalidae</i>	-0.610	-0.631	0.549	-0.519	-0.535	0.498	0.343	0.351
<i>Lygaeidae</i>	-0.254	-0.436	0.005	-0.905*	-0.884*	0.927**	0.998***	-0.443
<i>Reduviidae</i>	-0.964**	-0.997***	0.868*	-0.820*	-0.846*	0.787	0.542	0.555
<i>Pyrrhocoridae</i>	-0.610	-0.631	0.549	-0.519	-0.535	0.498	0.343	0.351
<i>Anthocoridae</i>	-0.610	-0.631	0.549	-0.519	-0.535	0.498	0.343	0.351
<i>Miridae</i>	0.415	0.276	-0.571	-0.256	-0.221	0.298	0.524	-0.759
<i>Cydnidae</i>	-0.127	-0.218	0.003	-0.453	-0.442	0.464	0.499	-0.221
<i>Carabidae</i>	0.127	0.304	0.108	0.792	0.768	-0.818*	-0.921**	0.512
<i>Pselaphidae</i>	0.030	0.199	0.189	0.692	0.666	-0.721	-0.844*	0.552

<i>Staphylinidae</i>	-0.964**	-0.997***	0.868*	-0.820*	-0.846*	0.787	0.542	0.555
<i>Aphodiidae</i>	0.205	0.352	-0.004	0.730	0.713	-0.748	-0.804*	0.357
<i>Rutelidae^a</i>	0.161	0.276	-0.003	0.573	0.560	-0.587	-0.632	0.280
<i>Tenebrionidae</i>	-0.368	-0.347	0.374	-0.182	-0.197	0.164	0.044	0.324
<i>Curculionidae</i>	-0.790	-0.817*	0.712	-0.672	-0.693	0.645	0.444	0.455
<i>Melolonthidae</i> larvae	0.633	0.496	-0.774	-0.091	-0.049	0.141	0.427	-0.902*
<i>Rutelidae</i> larvae	-0.610	-0.631	0.549	-0.519	-0.535	0.498	0.343	0.351
<i>Tenebrionidae</i> larvae	0.289	0.137	-0.469	-0.398	-0.364	0.437	0.637	-0.716
<i>Asilidae</i> larvae	0.342	0.267	-0.420	-0.055	-0.032	0.082	0.237	-0.492
<i>Therevidae</i> larvae	-0.610	-0.631	0.549	-0.519	-0.535	0.498	0.343	0.351
<i>Noctuidae</i> larvae	-0.610	-0.631	0.549	-0.519	-0.535	0.498	0.343	0.351
<i>Formicidae</i>	-0.604	-0.644	0.518	-0.592	-0.605	0.575	0.437	0.280

Note: * Correlation was significant at $p < 0.05$ (2-tailed). ** Correlation was significant at $p < 0.01$ (2-tailed). *** Correlation was significant at $p < 0.001$ (2-tailed).

BD, soil bulk density; C/N, the ratio of soil organic carbon to total nitrogen; EC, soil electrical conductivity; pH, the soil pH (in water); SOC, soil organic carbon; ST, soil temperature; STN, soil total nitrogen; SW, soil water content.

^aSome taxonomists consider the *Rutelidae* to be a sub-family (*Rutelinae*) of *Melolonthidae*. However, because of their different morphology, and for consistency with our taxonomic sources, we have retained them as a separate family.

management have profoundly influenced soil properties (Li et al., 2007). In our study area (Table 1), accelerated soil erosion due to cultivation was a principal factor responsible for the higher soil bulk density and the lower soil organic carbon content and total nitrogen content in the cultivated soils (Lal, 2000; Lobe et al., 2001). The higher soil moisture under cultivation might be the result of irrigation, whereas the higher soil moisture at the ungrazed sites might be the result of shallow root systems and of decreased evaporation from the soil surface as a result of the higher vegetation cover. The lower soil bulk density and pH at the ungrazed sites than at the grazed sites might be the result of higher vegetation cover and the elimination of trampling by livestock, combined with the effects of the extensive shallow root systems of vegetation (Su et al., 2004; Liu et al., 2011). Grazing exclusion increased the vegetation height, canopy cover, and dry weights of the plants as the vegetation recovered, thereby not only conserving soil moisture and decreasing soil pH and bulk density but also increasing the accumulation of SOC and nutrients (Zhao et al., 2005; Pei et al., 2008).

The changes in vegetation and soil conditions caused by grazing, cultivation, and grazing exclusion can greatly affect a soil's macro-faunal community structure (Kruess and Tschardt, 2002), since the plants represent food sources and both the plants and the soil provide habitats for organisms (Verdú et al., 2007). Grazing can affect plant-associated animal communities, and particularly invertebrates (González-Megías et al., 2004; Lindsay and Cunningham, 2009), as a result of changes in both the quantity and quality of the litter entering the food web, and as a result of the poor soil habitats and high soil disturbance created by livestock trampling (Bardgett et al., 2001; Grayston et al., 2001; Parfitt et al., 2010). These factors can result in considerably lower values of density, group richness, and Shannon's diversity index (Table 2), thereby creating a series of negative feedbacks in the soil and plant degradation processes that continue to degrade the system (Bugalho et al., 2011).

Previous researchers have reported negative effects of tillage on soil arthropods (e.g., Rosas-Medina et al., 2010). Tillage directly affects soil macro-faunal populations through mechanical damage, and indirectly affects these organisms by modification of their environment via the destruction of burrows, redistribution of litter in the soil profile, and changes in soil water content and temperature (Chan, 2001). The organic matter that supports nutrient recycling through decomposition food webs is typically lost as a result of cultivation (Lobe et al., 2001), which leads to decreased biodiversity (Adl et al., 2006). In addition, the dominance of a homogeneous herbaceous cover created by monoculture vegetation in the cultivated soils may result in markedly lower density and taxonomic richness together with a lower value of the associated Shannon's diversity index (Scheu, 1992). Conversely, the spatial variability of plant cover strongly influences the diversity of soil faunal community (Babel et al., 1992). Group richness and Shannon's index were markedly lower in the cultivated soils than in the soils of the ungrazed sites.

Vegetation recovery with increasing enclosure duration can mitigate the harsh microclimate caused by the study region's arid environment, thereby creating a more beneficial environment for other organisms (Table 1; Figure 2; Liu et al., 2010). Consequently, grazing exclusion can provide a wider range of plants as food and habitats, as well as higher soil moisture, organic carbon content and total nitrogen content, together with decreased disturbances that could endanger the soil fauna (Liu et al., 2009, 2011). The development of a more diverse vegetation cover and more diverse food sources in addition to the development of better microhabitats

can markedly increase density, group richness, and Shannon's index in the soils of the ungrazed sites (Scheu, 1992). These findings agree with the results of previous studies that suggested resource availability was an important determinant of the diversity of soil organisms (Hooper et al., 2000; Wardle, 2006). As we hypothesized, the soil macro-faunal community composition would become more diverse and complex as vegetation recovery proceeded (Bugalho et al., 2011).

The markedly higher Simpson's index at the grazed sites might be related to the combination of defoliation and the addition of urea and dung during grazing, which could be beneficial for some specific soil macro-faunal groups such as root-feeding *Melolonthidae* larvae and *Tenebrionidae* larvae, and predatory *Asilidae* larvae that could adapt to the presence of these grazing organisms (Ruess and McNaughton, 1984). Only these taxonomic groups dominated the soils of the grazed sites based on their density (supplementary Table S1). Another explanation was the accumulation of an energy supply in the form of non-palatable food resources for taxonomic groups other than these dominant groups (Decaëns et al., 1998). Such faunal groups as *Melolonthidae* larvae and *Tenebrionidae* larvae feed on decaying roots produced by livestock trampling (Whitford, 2000), and were associated with a low soil C/N ratio at our sites (Table 1; Figure 2), possibly due to inputs of urea N in livestock urine. The remarkably higher abundance of *Asilidae* larvae can be explained by the fact that these animals are predatory natural enemies of *Melolonthidae* larvae (Zhang et al., 2004), and this can cause bottom-up control in the soil food web (Doblas-Miranda et al., 2008).

The distinct environments of each site type can produce different macro-faunal communities as a result of different preferences for or tolerances of different living conditions (Figure 2). This presumably reflected a strong selection of or adaptation to specific habitats by particular soil macro-fauna (Salamon et al., 2008). As we noted earlier, the high abundance of root-feeding groups such as *Melolonthidae* larvae in the soils of the grazed sites can result in faster turnover of nutrients, together with associated changes in soil microbiology (Su et al., 2004). There was a negative correlation between *Melolonthidae* larvae and soil C/N (Table 3), with a low soil C/N ratio at the grazed sites (Table 1). However, due to their root-grazing herbivory, these groups might negatively affect the root systems of the remaining grasslands (Coleman et al., 2004). Grazing might therefore result in an outbreak of these pests, further deteriorating the grassland ecosystems (Yuan, 1995).

In the cultivated soils, the higher number of predatory *Carabidae*, *Pselaphidae*, and *Aphodiidae* might be correlated with their strong mobility and their adaptation to the disturbance caused by tillage (Supplementary Table S1; Chan, 2001), even though SOC and STN were much lower at the cultivated sites than at the ungrazed and grazed sites (Table 1). The typical plant monoculture in the cultivated soils results in more homogeneous food sources that lead to a simpler faunal community composition, and thus to a simplified soil food web (Wardle, 2006). Such predatory groups as the *Carabidae* and *Pselaphidae* may also have been part of the fauna that temporarily colonizes a cultivated site in response to the periodic disturbances created by cultivation. These results suggest simplification and instability of grassland ecosystems (Doblas-Miranda et al., 2008), both of which can delay the recovery of biodiversity and the conservation of cropland.

In the soils of the ungrazed sites, the soil macro-faunal communities were characterized by the greatest numbers of herbivores (e.g., *Lygaeidae*, *Curculionidae*) and omnivores (e.g., *Formicidae*), and by most kinds of specific groups (e.g., predatory

Theridiidae, *Thomisidae*, *Salticidae*, and *Staphilinidae*; herbivorous *Rhopalidae*, *Reduviidae*, *Pyrrhocoridae*, and *Anthocoridae*). This resulted from the higher and more diverse vegetation cover and soil nutrient levels that resulted from vegetation recovery following grazing exclusion (Table 1; Liu et al., 2010). Grazing exclusion not only improved the soil properties and vegetation recovery, but also enhanced the complexity of the soil food web, thus facilitating the stability and conservation of grassland ecosystems (Bugalho et al., 2011). Further, our results will have important implications for the use of these taxa as indicators; for example, the *Lygaeidae* and the *Melolonthidae* larvae might be indicators of ungrazed and grazed grassland, respectively. However, a more extensive assessment would be required across a broader range of soil environments before these taxa can be used as indicators, since the integrated effects of various soil factors can strongly affect macro-faunal abundance (Thomas et al., 2004; Schon et al., 2008).

Conclusions

In the semiarid Horqin Sandy Land, cultivation and continuous grazing have decreased the soil organic carbon and total nitrogen contents, and have increased soil alkalinity and bulk density. After livestock exclusion, the soil pH and bulk density decreased, and soil organic carbon and total nitrogen increased markedly. These changes resulted in a considerably higher density of individuals, group richness, and Shannon's index for the soil macro-faunal community in the soils of ungrazed sites than in the soils of grazed and cultivated sites. Soil bulk density, pH, EC, and the soil C/N ratio were the main factors that affected the macro-faunal community structure. Several soil macro-faunal groups responded to the different living conditions at the three sites based on their ability to adapt to or their preference for specific habitats. These findings will have important implications for the soil ecosystems created by different forms and intensities of land management.

Overall, our results suggested that cultivation and continuous grazing adversely affected soil properties and macro-faunal diversity, whereas livestock exclusion promoted vegetation recovery and the development of richer macro-faunal assemblies with improved biodiversity. Thus, grazing exclusion appears to be a good option for promoting the recovery of ecosystem functioning in these degraded sandy grasslands. Our results also suggest that the degraded grasslands should not continue to be overgrazed or converted into cropland; both grazing and cultivation may be possible, but at lower intensity and only if accompanied by measures such as moving grazing animals to new pastures before they damage the vegetation excessively, and sowing cover crops to protect cultivated soils after harvesting.

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