ABSTRACT: In arid and semi-arid areas, heavy grazing combined with climate change cause land degradation (e.g., desertification). Grazing management is essential for ecosystem recovery and desertification control in these areas, including Northern China's Horqin Sandy Land. However, the recovery of soil faunal community during grazing exclusion is unknown. We examined plant and soil macro-invertebrate community structure together with soil properties in three treatments in a representative degraded Horqin sandy grassland: exclosure for 15 and 10 years (15EX and 10EX) and long-term continuous grazing (CG). The vegetation cover and height increased significantly and soil bulk density decreased significantly along the gradient from CG to 15EX, but there were no significant differences in soil pH, electrical conductivity, organic carbon and total nitrogen. Soil macro-invertebrate abundance, group richness and diversity increased along the gradient from CG to 15EX, with significant differences in invertebrate abundance and group richness between CG and 15EX; there was no significant differences between CG and 10EX. There were no significant differences in soil macro-invertebrate diversity and evenness between these three treatments. These results suggested that grazing exclusion for at least 15 years might be necessary for the recovery of these fauna. The vegetation height and the soil electrical conductivity, organic carbon, and total nitrogen determined the distribution and community structure of soil macro-invertebrates. Some faunal groups lived in specific habitats due to strong adaptation to different management practices. For example, the Thomisidae, Philodromidae, Salticidae, and Rhopalidae tended to live in habitats with tall vegetation. The Lygaeidae, Miridae, Teneberionidae, and Linyphiidae adapted to live in soil with low soil organic carbon and nitrogen (ungrazed grassland).

KEY WORDS: soil macro-fauna, grazing management, sandy grassland, community diversity, exclosure, Horqin Sandy Land.

1. INTRODUCTION

Land degradation processes such as desertification in arid and semi-arid areas result from various factors, including climatic variation and unsustainable human activities (UNEP 1992, Gad and Abdel 2000). Livestock grazing is a dominant human activity in many semi-natural and managed rangelands (Soderstrom et al. 2001) and is the main cause of land degradation and sandy desertification in many arid and semi-arid areas (Okin et al. 2001, Zhao et al. 2005). Grazing management is increasingly aimed at the restoration of grassland ecosystems and desertification control (Mekuria et al.)
Ren-Tao Liu et al. 2007, Parfittta et al. 2010). Bugalho et al. (2011) reported that controlled grazing can be beneficial to biomass accumulation and seed production, and may therefore mitigate desertification of rangeland (Su et al. 2005, Zhao et al. 2005). Grazing exclusion can lead to the restoration of soil fertility, vegetation biomass and composition, soil fauna, and water storage (Mekuria et al. 2007, Parfitta et al. 2010).

When grazing management alters plant diversity and the availability of key resources, the diversity and abundance of invertebrates is likely to change (Siemann et al. 1998, García et al. 2010). Reduced grazing pressure can increase species richness and the abundance of phytophagous insects and their parasitoids (Parfitta et al. 2010, Bugalho et al. 2011), whereas intense grazing can reduce arthropod species diversity and abundance (Gardner et al. 1997, Mills and Adl 2011). However, the relationship between soil macro-invertebrate community diversity and grazing management in arid and semi-arid areas is poorly understood. Soil macro-invertebrates are an essential part of the soil ecosystem, so research on their relationships with the recovery process is necessary to improve our understanding of the factors that govern biological restoration in sandy grasslands and to guide the development of effective desertification control policies.

The objectives of the present study were to quantify the impacts of grazing and grazing exclusion on macro-invertebrate community and to assess the effects of short- and long-term exclosure on macro-invertebrate diversity in the Horqin Sandy Land.

2. STUDY AREA

The study area is located in the southwestern part (42°55’N and 120°42’E, 360 m elevation) of the Horqin Sandy Land, in Inner Mongolia, northern China (Fig. 1). The region has a temperate continental semiarid monsoonal climate. The annual mean pre-

Fig. 1. Location of the study area (the Horqin Sand Land in Inner Mongolia). The black region indicates the study area near the Naiman Desertification Research Station.
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Precipitation is 366 mm, of which 70–80% falls during the summer growing season, versus an annual mean potential evaporation of 1935 mm. The annual mean temperature is 6.8°C, with a maximum mean monthly temperature of 21.9°C in July and a minimum of –14.7°C in January. The annual frost-free period ranges from 130 to 150 days. The annual average wind speed is 3.4 m s⁻¹, with a mean of 4.3 m s⁻¹ during the primary aeolian erosion season (the spring, when vegetation cover is low). Significant wind erosion often occurs from April to mid-June, before the rainy season arrives. Two sandy soil types predominate in the Horqin Sandy Land, with sandy Kastanozems under the original grasslands, generally in the west and present as remnants elsewhere, and with more widespread Arenosols in the desertified central and eastern areas (Cheng et al. 1997, Su et al. 2003).

The study area sustained disorganized but unsustainable and continuous livestock grazing for many years, and significant desertification had been observed by the early 1990s (Zhao et al. 2007). At that time, a restoration project was initiated and grazing exclosures were established using fences constructed with concrete pilings and barbed wire. In the exclosures, all grazing by domestic herbivores was excluded with the goal of allowing the natural vegetation to recover. The dominant plant species were grasses such as Cleistogenes squarrosa (Trin.) Keng, Setaria viridis (L.) Beauv., Phragmites australis L., Digitaria ciliaris (Retz.) Koeler, Leymus chinensis (Trin.) Tzvel, and Pennisetum centrasiaticum Tzvel, accompanied by legumes and forbs such as Mellissitus ruthenicus (L.) C.W. Chang, Salsola collina Pall., Corispermum elongatum Bunge, Agriophyllum squarrosum (L.) Moq., and Artemisia scoparia Waldst. et Kit. The vegetation also included a few shrubs, such as Caragana microphylla Lam., Lespedeza davurica (Laxm.) Schindl. var. sessilis V. H. Vassil., and sub-shrubs such as Artemisia halodendron Turcz. ex Besser and Artemisia frigida Willd. (Li et al. 2000).

3. MATERIALS AND METHODS

3.1. Field sampling

The field sites were established in an open, flat, and degraded sandy grassland covering 50 ha, at the long-term observation site of the Naiman Desertification Research Station, operated by the Chinese Academy of Sciences (Fig. 1). Three sites were selected for sampling: (1) 15EX, a 0.8-ha exclosure that has not been grazed for 15 years (38 × 200 m); (2) 10EX, a 1.5-ha exclosure that has not been grazed for 10 years (75 × 200 m); and (3) CG, a 1-ha continuously grazed (6–10 or more than 10 sheep units per ha; Zhao et al. 2005) grassland adjacent to the 15EX and 10EX sites.

At each site, we established two (30 × 30 m) plots at the time of peak standing crop in August 2008. Five sampling points were chosen randomly within each plot. In total, there were 30 samples in this investigation. At the centre of each sampling point, we excavated 30 × 30 × 30 cm soil samples, and recovered all macro-organisms from these samples by hand sorting to investigate soil macro-invertebrate community structure.

Near the soil samples, we obtained additional soil samples using a cylindrical 100-cm³ stainless steel soil auger for the analysis of soil physico-chemical properties. We obtained two soil cores at each sampling point to determine the soil bulk density (BD). At each sampling site, we established five randomly located 1 × 1 m quadrates adjacent to the sampling points to investigate the number of plant species, their density (individuals m⁻²), and the overall ground vegetation cover (VC, %), as well as the mean vegetation height (VH, cm). Vegetation cover was estimated visually using sample images for VC values ranging from 0 to 100% in 10% intervals. Vegetation height was the mean height of all plants inside the quadrates.

Soil macro-invertebrates were stored in 75% alcohol in the field and brought back to the laboratory for identification. They were identified at the order or family level according to the key of Yin (2000), and were classified into groups on the base of morphological features under a binocular magnifying glass (40×). The larvae and adults were counted separately because of their different functions in the soil during these different parts of their life history.

Soil samples were passed through a 2-mm sieve to remove plant parts and other debris, and were then air-dried. Soil pH and electrical conductivity (EC, μs cm⁻¹) were determined.
in 1:1 (v/v) soil water solution and in 1:5 (v/v) soil water aqueous extract, respectively. The subsamples were then finely ground to pass a 0.25-mm sieve, and analyzed for soil organic carbon (SOC, g kg\(^{-1}\)) using the Walkley-Black dichromate oxidation procedure (Nelson and Sommers 1982) and for soil total nitrogen (STN, g kg\(^{-1}\)) using the Kjeldahl procedure (UDK140 Automatic Steam Distilling Unit, Automatic Titroline 96, Italy) (ISSCAS 1978).

### 3.2. Data analyses

We calculated several indices for the soil invertebrate community indices. First, we determined the abundance of each taxonomic group (individuals m\(^{-2}\)) and calculated the group richness (number of taxonomic groups at the order or family level). We described the diversity using Shannon’s index (\(H^\prime\)) and its associated evenness index (\(E\); Southwood and Henderson 2000):

\[
H^\prime = –\Sigma P_i \log_2 P_i
\]

where: \(P_i = x_i/\Sigma x_i\)

\[
E = H^\prime/\ln S
\]

where: \(x_i = \) the number of individuals in the group category \(i\) and \(S = \) number of group categories, and \(P_i = \) the proportion of the total number of individuals belonging to the group category \(i\).

All statistical analyses were carried out using version 15.0 of SPSS for Windows (SPSS Inc, Chicago, IL, USA), with multiple comparisons and analysis of variance (ANOVA) used to determine the differences among the treatments (Sokal and Rohlf 1995). Before applying parametric tests, we tested normality for homogeneity of variances. For non-normal distributions, we log\(_{10}\)-transformed the data before analysis. For all tests, statistically significant differences were assigned to \(P < 0.05\).

Redundancy analysis (RDA) was used to examine the main abiotic factors (BD, pH, EC, SOC and STN) and biotic factors (VC and VH) affecting the individual macro-invertebrate groups. Data were first analyzed by detrended correspondence analysis (DCA) using version 4.5 of the CANOCO Software (Microcomputer Power, Ithaca, NY, USA), which suggested RDA as an appropriate approach for further analysis (length of gradient < 4 for macro-invertebrate communities). We used RDA to correlate each macro-invertebrate group with the environmental variables by selecting the linear combinations of environmental variables that gave the smallest residual sum of squares (Kennedy et al. 2004).

### 4. RESULTS

#### 4.1. Environmental variables

There were no significant differences among the three treatments in soil pH, EC, SOC and STN (\(P > 0.05\)) but BD decreased significantly (\(P < 0.05\); from 1.61 to 1.37 g cm\(^{-3}\)) with increasing duration of exclosure. VC was significantly higher at the 15EX site than in the other two treatments (\(P < 0.05\)), and VH was significantly higher in both the exclosure treatments (\(P < 0.05\); Table 1). VH in the exclosures increased to at least 1.6 times the value in CG after 10 and 15 years.

At the 15EX site, some annual and perennial grasses (\(C.\) squarrosa, \(S.\) viridis, and \(P.\) australis) and some forbs (\(S.\) collina, \(A.\) scoparia, and \(Chenopodium\) \(glaucum\) \(L.\)) were the major species. At the 10EX site, the major species.

<table>
<thead>
<tr>
<th>Parameters</th>
<th>BD (g m(^{-3}))</th>
<th>pH</th>
<th>EC</th>
<th>SOC (g kg(^{-1}))</th>
<th>STN (g kg(^{-1}))</th>
<th>VC (%)</th>
<th>VH (cm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>15EX</td>
<td>1.37±0.07b</td>
<td>8.34±0.03a</td>
<td>41±5a</td>
<td>2.75±0.20a</td>
<td>0.22±0.02a</td>
<td>88.75±4.27a</td>
<td>23.75±4.32a</td>
</tr>
<tr>
<td>10EX</td>
<td>1.46±0.10b</td>
<td>8.28±0.06a</td>
<td>44±5a</td>
<td>4.48±0.33a</td>
<td>0.27±0.03a</td>
<td>58.06±9.24b</td>
<td>28.15±8.83a</td>
</tr>
<tr>
<td>CG</td>
<td>1.61±0.03a</td>
<td>8.35±0.01a</td>
<td>42±3a</td>
<td>3.35±0.23a</td>
<td>0.25±0.02a</td>
<td>43.67±7.36b</td>
<td>14.67±5.39b</td>
</tr>
</tbody>
</table>
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Species were S. collina, A. scoparia, S. viridis, and L. davurica var. sessilis. The dominant plant species at the CG site were the forbs S. collina and A. scoparia.

4.2. Community structure of soil macro-invertebrates

We collected a total of 337 individuals from 40 macro-invertebrate groups by hand sorting. Most of the groups had a very low abundance: 33 of the macro-invertebrate groups had fewer than 10 individuals, and the abundance of the other 7 groups ranged from 11 to 70 individuals. In addition, the soil macro-invertebrate community was dominated by insects, with the most numerous individuals groups in the Staphilinidae (Coleoptera), larval Melolonthidae (Coleoptera) and Formicidae (Hymenoptera). The dominant groups at the 15EX site were the Staphilinidae, larval Melolonthidae and Formicidae; at the 10EX site, they were the Carabidae (Coleoptera) and Formicidae, and at the CG site, they were the larval Melolonthidae, larval Tenebrionidae (Coleoptera), larval Asilidae (Diptera), and Formicidae.

Grazing exclusion significantly affected the abundance of the soil macro-invertebrates ($F = 24.88, P < 0.05$). Their abundance increased significantly ($P < 0.05$) after exclosure for 15 years compared with the other sites (Fig. 2A). However, the abundance at the 10EX site did not differ significantly from that at the CG site. There were also remarkable influences of grazing exclusion on the group richness, which was significantly higher at the 15EX than at the CG site (Fig. 2B), but there was no significant difference between the 10EX and CG sites or between the 10EX and 15EX sites.

The community diversity, as measured by Shannon’s index, increased with increasing duration of exclosure, but the differences

---

Table 2. Results of the redundancy analysis (RDA). Values are for axes 1 and 2, which are plotted in the RDA diagram in Fig. 4. The highest canonical coefficients and correlations (values > 0.9 or < −0.9) are highlighted with *.

<table>
<thead>
<tr>
<th>Axis</th>
<th>1</th>
<th>2</th>
</tr>
</thead>
<tbody>
<tr>
<td>Eigenvalue</td>
<td>0.422</td>
<td>0.252</td>
</tr>
<tr>
<td>Cumulative percentage of variance:</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Species data</td>
<td>42.2</td>
<td>67.4</td>
</tr>
<tr>
<td>Species-environment relationship</td>
<td>62.6</td>
<td>99.8</td>
</tr>
<tr>
<td>Summary of Monte Carlo test:</td>
<td>For all axes:</td>
<td></td>
</tr>
<tr>
<td>$F$ ratio</td>
<td>2.91</td>
<td>3.71</td>
</tr>
<tr>
<td>$P$ value</td>
<td>0.05</td>
<td>0.04</td>
</tr>
<tr>
<td>Species-environment correlations</td>
<td>0.994</td>
<td>0.997</td>
</tr>
</tbody>
</table>

Correlations ($r$):

| BD  | -0.538 | 0.842 |
| pH  | 0.664  | 0.747 |
| EC  | -0.982* | -0.187 |
| SOC | -0.932* | -0.360 |
| STN | -0.989* | -0.142 |
| VC  | 0.794  | -0.607 |
| VH  | -0.137 | -0.990* |

Coefficients ($c$):

| BD  | -0.535 | 0.840 |
| pH  | 0.660  | 0.745 |
| EC  | -0.976* | -0.187 |
| SOC | -0.927* | -0.359 |
| STN | -0.984* | -0.142 |
| VC  | 0.789  | -0.606 |
| VH  | -0.136 | -0.987* |
among sites were not significant \((P > 0.05)\); Fig. 3A). The maximum diversity occurred at the 15EX site \((2.37 \pm 0.36)\) and the minimum diversity occurred at the CG site \((1.93 \pm 0.02)\). To ensure that these differences were not simply due to the observed changes in the group richness, we also calculated the evenness \((i.e., the relative Shannon index)\). The evenness decreased with increasing duration of exclosure, but the differences among sites were not significant \((P > 0.05);\) Fig. 3B). The evenness \((0.74)\) was slightly lower at the 15EX site than at the other sites, but still represented 74% of its maximum possible value.

4.3. Main environmental factors that affected the distribution of the soil macro-invertebrate groups

We used RDA to identify the main factors that affected the macro-invertebrate community structure in the three management treatments, with rankings based on the relative abundance of the macro-invertebrate groups (Table 2; Fig. 4). Table 2 showed that axes 1 and 2 were significant and explained 42.2 and 25.2% of the overall variance, respectively, within the macro-invertebrate group data, together accounting for 67.4% of the total variance. The species-environment relationship for axes 1 and 2 accounted for 99.8% of the total variance, indicating that together, these axes accounted for the bulk of the variance in the macro-invertebrate group data that could be attributed to environmental factors (abiotic and biotic properties). Species-environment correlations for both axes were \(> 0.99\), indicating that the macro-invertebrate group data were strongly correlated with the environmental parameters. Monte-Carlo significance tests revealed that both the first axis and all axes combined explained a signifi-
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Fig. 4. Redundancy analysis (RDA) ordination diagram for the soil invertebrate group data for the environmental variables (pH, BD, STN, EC, SOC, VH, and VC) and for the faunal groups (numbers). Environmental variables: BD – soil bulk density; EC – electrical conductivity; SOC – soil organic carbon; STN – total soil nitrogen; VC – vegetation cover; VH – vegetation height.


Fig. 4 showed the RDA plot for the macro-invertebrate group data in relation to both, the abiotic and biotic axes. The macro-invertebrate group arrows that pointed in approximately the same direction as an environmental factor arrow indicated a strong positive correlation (the longer the macro-invertebrate group arrow, the stronger the relationship; Kennedy et al. 2004). Arrows for soil EC, SOC, STN, and VH were longer than those for BD, pH, and VC, indicating that EC, SOC, STN, and VH accounted for a greater proportion of the variance in macro-invertebrate community structure than BD, pH, and VC. EC, SOC, STN, and VH had the strongest influences on the macro-invertebrate community, and these influences were significant. The canonical coefficients and intraset correlations for the environmental factors for each axis indicated that axis 1 primarily represented a soil EC, SOC, and STN gradient and that axis 2 represented a VH gradient.

Fig. 4 also demonstrates that different macro-invertebrate groups within the community responded differently to specific abiotic and biotic properties. For example, the abundance of Linyphiidae, Lygaeidae, Miridae, and Tenebrionidae was inversely correlated with soil EC, SOC, and STN, whereas the abundance of the Thomisidae, Philodromidae, Salticidae, and Rhopalidae was positively correlated with VH. The abundance of the Oxyopidae, Gnaphosidae, Gryllophilidae, Anthocoridae, Cyniidae, Staphilinidae, Pselaphidae Latreille, Aphodiidae Leach, Elateridae Leach, Carabidae, larval Elateridae Leach, larval Coccinellidae, larval Curculionidae, larval Meloidae, adult Noctuidae, larval Noctuidae, larval Pyralidae, Formicidae, Tenthredinidae, and mites was positively correlated with VC, whereas the abundance of the Cicadellidae was negatively correlated with VC. The abundance of the Araneidae, Lycosidae, larval Melolonthidae, larval Tenebrionidae, and larval Aslidae was positively correlated with soil pH.
5. DISCUSSION AND CONCLUSION

Grazing management has played an important role in vegetation restoration and soil improvement in the semi-arid steppes of the Horqin Sandy Land (Li et al. 2000, Zhao et al. 2005). In the present study, VC and VH increased significantly with increasing duration of exclosure, but the soil properties generally did not increase except for BD. It was reported that VC was often used to assess the spatial extent and degree of desertification (Zhu and Wang 1992, de Soyza et al. 1998). In the Horqin Sandy Land, previous research suggested that the establishment of grazing exclosures could remarkably benefit the restoration of degraded steppes within a time as short as 10 or 15 years (Zhao et al. 2005, Su et al. 2003, 2005). In the present study, grazing exclusion significantly increased VC and decreased BD compared to levels at the CG site. The higher BD at the grazed site presumably resulted directly from compaction due to livestock trampling and indirectly from a decrease in belowground biomass (Zhao et al. The changes in vegetation and soil conditions that occur under grazing management can have remarkable influences on the soil invertebrate community structure (Kruess and Tscharntke 2002), since plants are food sources and the soil habitats they create provide refuges for soil fauna (Kruess and Tscharntke 2002, Verdú et al. 2007). We found significant increases in soil macro-invertebrate abundance after 15 years of grazing exclusion. Group richness was also significantly higher at the 15EX site than at the CG site. Soil macro-invertebrate diversity (Shannon’s index) also increased and macro-invertebrate evenness decreased with increasing duration of exclosure, though the differences were not significant. There was a relatively stable and higher VC and VH in the exclosures, and this is likely to provide habitat for a large and diverse invertebrate fauna (Wang et al. 2006). Increases in VC with increasing exclosure duration might increase the diversity of phytophagous soil fauna because most soil fauna feed on plants (Rosenzweig and Abramski 1993, Siemann 1998) and because the plants mitigate the harsh microclimate of arid environments, creating a more beneficial environment for more kinds of soil fauna (Liu et al. 2009). Plants can also influence the soil biota directly by providing a source of energy (i.e., carbon) in their root exudates and as a result of inputs into the detrital pool through root turnover, and contribute indirectly by altering soil structure and hydrology, thereby influencing the microclimate experienced by soil organisms (Wolters et al. 2000). In contrast, the significantly increased BD in the continuously grazed grassland could lead to negative feedbacks in soil and plant processes that would continue to degrade the system (Manzano and Návar 2000), thereby negatively influencing soil invertebrates. As a result, there was a lower soil macro-invertebrate abundance, group richness, and diversity at the CG site than in the two exclosures.

We found no significant differences in soil macro-invertebrate abundance and group richness between the 10EX and CG sites. This suggested that 10 years of exclosure was insufficient to significantly improve the degraded grassland. One of the main reasons for this finding might be the lack of a significant difference in VC between the 10EX and CG sites (Rosenzweig and Abramski 1993, Siemann 1998, Wang et al. 2006). Furthermore, there were no significant differences in soil macro-invertebrate diversity and evenness between the three treatments. This indicated that even 15 years of grazing exclusion had a limited effect on these biological characteristics of the degraded grassland.

Nevertheless, the changes in environmental conditions that developed under the grazing management practices resulted in the selection of certain macro-invertebrate groups that were adapted to the habitats created by these practices (Lv et al. 2007, Verdú et al. 2007, Liu et al. 2009). Ungrazed grasslands might provide a higher resource heterogeneity than grazed grasslands, since VH was significantly greater, and this would create a more complex plant architecture (Forbes et al. 2005, Daryanto and Eldridge 2010), which would attract species in the Thomisidae, Philodromidae, Salticidae, and Rhopalidae. In addition, the differences in VC might result in differences in microclimate (Halffiter and Arellano 2002) and might enhance resource heterogeneity (Kruess and Tscharntke 2002);
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These differences affected the distribution of the Oxyopidae, Gnaphosidae, Gryllotalpidae, Cicadellidae, Anthocoridae, Cydnidae, Staphilinidae, Pselaphidae Latreille, Aphodiidae Leach, Elateridae Leach, larval Carabidae, larval Elateridae Leach, larval Coccinellidae, larval Curculionidae, larval Melolonthidae, adult Noctuidae, larval Noctuidae, larval Pyralidae, Formicidae, and Tenthredinidae (Dennis et al. 2001, Robson et al. 2009). Unfortunately, we did not measure key environmental parameters such as soil moisture content, temperatures above and within the vegetation, and soil temperatures that would be potentially significant factors that determine the suitability of the sites for different taxonomic groups. The relationships between the macro-invertebrates and these factors should be the subject of future research.

In our study ecosystem, differences in soil conditions would produce differences in habitat and resource (root and litter) quality. Several soil properties (EC, SOC, and STN) appeared as main factors that explained the observed significant differences in the distributions of certain macro-invertebrate groups, such as the Lygaeidae, Miridae, Tenebrionidae, and Linyphiidae; the abundance of these groups was correlated with low soil EC, SOC, and STN in grasslands after 15 years of grazing exclusion (Doblas-Miranda et al. 2009). In contrast, the Araneidae, Lycosidae, larval Melolonthidae, larval Tenebrionidae, and larval Asilidae tended to live in habitats with high pH. These differences in macro-invertebrate distribution become important because they also affect the trophic structure of the species assemblages associated with different microhabitats, and these associations have potential functional consequences for the ecosystem (Doblas-Miranda et al. 2009, Liu et al. 2009).

In conclusion, we found that continuous grazing in this degraded sandy grassland was detrimental to the vegetation and soil, resulting in a significant decrease in VC and VH. The exclosure protected the soil and vegetation, allowing an increase in VC and VH that was accompanied by increased soil macro-invertebrate abundance, group richness, and diversity, particularly after 15 years of grazing exclusion. While there was a limited effect on soil macro-invertebrate diversity and evenness even after 15 years. Furthermore, we found clear associations between certain groups of soil macro-invertebrates and certain combinations of environmental conditions, and these associations were affected by the management practices. Our results suggested that the implementation of long-term management such as grazing exclusion would improve soil macro-invertebrate diversity, thereby promoting the restoration of degraded ecosystems in the Horqin Sandy Land.

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6. REFERENCES


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