Effectiveness of afforested shrub plantation on ground-active arthropod communities and trophic structure in desertified regions

Rentao Liu [a,b,*], Fan Zhu [a], Yosef Steinberger [b]

[a] Key Laboratory for Restoration and Reconstruction of Degraded Ecosystem in Northwestern China of Ministry of Education, Ningxia University, Yinchuan 750021, China
[b] Mina & Everard Goodman Faculty of Life Sciences, Bar Ilan University, Ramat Gan 52900, Israel

ARTICLE INFO

ABSTRACT

In desertified regions, the mobile sand land, and afforested shrubland of different ages (i.e., 6, 15, 24 and 36 years) in addition to the naturally restored grassland were selected in order to evaluate their consequences on taxonomic and trophic structures, and the diversity of ground-active arthropods. The taxa-specific responses to the land conversion were found within functional groups. The activity density of predator Labiduridae and Carabidae, and herbivore Tenebrionidae decreased, while that of herbivore Melolonthidae and omnivore Formicidae increased markedly ($p < 0.05$) from the mobile sand land to the afforested shrubland, regardless of shrubland ages. The taxa richness of herbivores also increased markedly ($p < 0.05$) from the mobile sand land to the afforested shrubland, regardless of shrubland ages. However, there was little effect of land conversion on the activity density and richness of detritivores due to the lack of litter on the surface of soils. Different functional groups of ground-active arthropods responded in a different way to the land conversion. The total density and Simpson index decreased, while the taxa richness and Shannon index increased markedly ($p < 0.05$) from the mobile sand land to afforested shrubland, regardless of shrubland ages. It was suggested that the afforested shrub plantations of different ages had similar implications on the trophic structure and diversity indices as was close to the naturally restored grassland (except for Shannon index). The Shannon index in the grazed afforested shrubland of 24 years that was close to the naturally restored grassland was markedly ($p < 0.05$) lower compared with the other afforested plantations. In conclusion, the afforested shrubland was an option for the stabilization of mobile sand land and biodiversity recovery of degraded ecosystems, regardless of shrub ages. Furthermore, the grazing management on afforested shrubland could improve the effectiveness of this recovery process.

© 2014 Elsevier B.V. All rights reserved.

1. Introduction

In arid and semi-arid areas, the sandy grassland is susceptible to land degradation and has undergone severe desertification primarily due to heavy grazing combined with climate change (Li et al., 2004; Zhao et al., 2005). Desertification in addition to drought which is directly related to the degradation of vegetation cover, soil degradation, and nutrient depletion, is causing great ecological concerns about the management on desertification reversion (Mekuria et al., 2007; Nyssen et al., 2004; Zhao et al., 2010). The widespread afforestation program has been reported as an effective management for the stabilization of mobile sand land and the recovery of degraded arid ecosystems, and plays an important role in rehabilitating desertified ecosystems (Jiang et al., 2007; Su and Zhao, 2003). In addition, the rate of transition from an afforested plantation to natural vegetation depends on the aboveground and belowground processes (Wardle et al., 2004). It was found

that the afforested shrubs aboveground acting as "fertile island" indicated an improvement of soil properties and herbaceous vegetation in mobile sand land (Zhao et al., 2007). Correspondingly, these changes in environmental variables acting as living conditions and food sources for soil animals might have fundamental effects on the composition, abundant distribution and trophic structure of soil biota community belowground.

In the desertified regions, the perennial desert shrubs selected as the afforested plantations were found to become "hot spots", acting as a pronounced biological "hub" in determining soil community (Cushman et al., 2010; J.L. Liu et al., 2012; Liu et al., 2012a,b). The improved shrubland habitat maintained significantly higher abundance and group richness diversity of soil arthropods compared with the mobile sand land (Liu et al., 2013). This was ascribed to the multiple functions that the shrubs served as primary producers and providers of resources as well as modulators of the physical environment (e.g., microclimate, soil nutrient and water availability) beneath their canopies (Shachak et al., 2008; Wang et al., 2011). These ecological functions were combined to create a favorable microhabitat with relatively mild microclimates, secure oviposition sites, high resource availability and shelter, thereby resulting in a greater degree of arthropod

* Corresponding author at: Key Laboratory for Restoration and Reconstruction of Degraded Ecosystem in Northwestern China of Ministry of Education, Ningxia University, Yinchuan 750021, China. Tel./fax: +86 951 4967201.
E-mail addresses: nxuliu2012@126.com, liubarilanu@gmail.com (R. Liu).

activity and colonization in areas under shrubs in comparison to the open spaces between shrubs (Titus et al., 2002; Liu et al., 2011). The activities of soil organisms played key roles in various ecosystem functions such as soil structure dynamics and decomposition of organic matter (Lavelle and Spain, 2001). As reported, the shrub patches in mobile sand land facilitated soil arthropod assembly and acted as "Arthropod Island" that was directly correlated with the food web structure and ecological function (Zhao and Liu, 2013). Shrub plantation age indicated significant influences on the diversity of ground arthropods though that depended on seasonal changes (Liu et al., 2013). However, the recovery effectiveness and extent of ground-active arthropods along the chronosequence of afforested shrub plantations remained unclear, relative to the naturally restored grassland.

The goal of this study was to describe the changes of taxonomic and trophic structures, and biological diversity of ground-active arthropods in the mobile sand land along a chronosequence of afforested shrublands (6, 15, 24 and 36 years) relative to the naturally restored grassland.

2. Materials and method

2.1. Ethics statement

A scientific research and collecting permit were obtained for the study from the Science and Technology Department of Yanchi County, Ningxia in 2013. No specific permits were required for the described field study. Our study did not involve endangered or protected species. Voucher specimens were deposited in the Key Laboratory for Restoration and Reconstruction of Degraded Ecosystems in Northwestern China of Ministry of Education, Ningxia University.

2.2. Study area

The study was conducted in northern Yanchi county (37°04′–38°10′ N and 106°30′–107°41′E), which is located at the southwest fringe of Mu Us sandy land in the Ningxia Hui autonomous region, China (Fig. 1). The study region is a typical agro-pasture transition zone
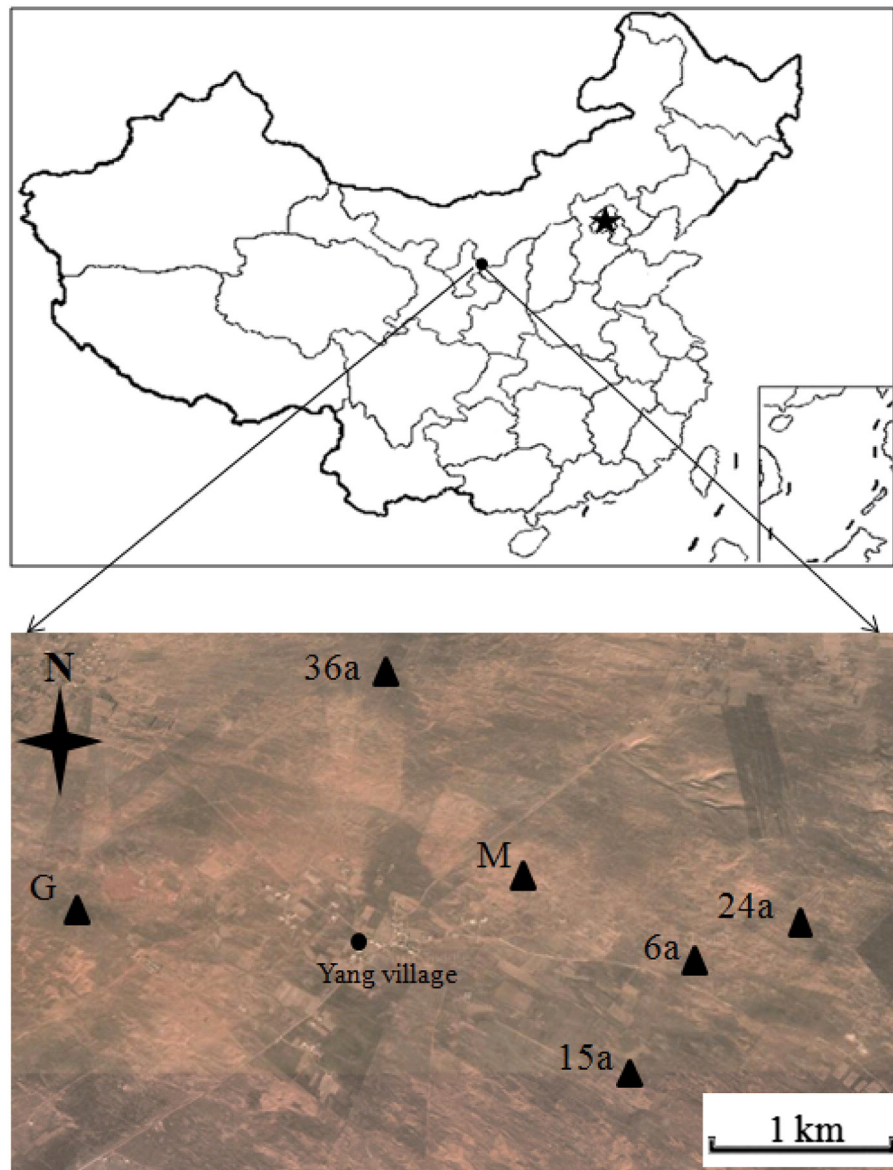
Imagery ©2014 DigitalGlobe, Map data ©2014 Autonavi, Google

Fig. 1. Location of study area (●) and sampling sites (▲) in northwestern China.

comprising a transition from typical steppe to desert steppe and sandy grassland that was susceptible to desertification. The elevation ranges from 1295 to 1951 m above sea level (1600 m on average) from North to South, and the region has a temperate continental semi-arid monsoonal climate. Mean annual precipitation is 292 mm, with about 70% of the total precipitation occurring from June to September. Mean annual potential pan-evaporation is 2710 mm. Mean annual temperature is 7.5 °C, and the lowest and highest monthly mean temperatures are −8.7 °C in January and 22.4 °C in July, respectively. Mean annual wind velocity is 2.8 m s^{-1} and prevailing winds are mainly northwest in April and May. Sand dust blowing at wind velocities over 5.0 m s^{-1} occurs 323 times per year, on average. Wind erosion often occurs from April to mid-June before the rainy season arrives (climate data from Yanchi Meteorological Station, 1976 to 2010). The main soil types of the study sites are Orthi-sandic Entisols and Arenosols (i.e., sandy soils) (World Reference Base for Soil Resources), which are poor in fertility with loose structure and are very susceptible to wind erosion (Xu et al., 2012).

Over the past several decades, the study region has undergone severe desertification and is now a key area for desertification research in China (Liu et al., 2013). Desertification started in the 1950s and reached a peak in the late 1970s, caused by disturbance of the sandy grasslands from extensive fuel wood gathering and overgrazing. During the desertification peak, mobile, semi-mobile and semi-fixed sand sheets together made up 33% of the total land area in this county. In order to curb desertification and alleviate its detrimental effects, the local government implemented a conservation program in the 1970s that selected strong adaptive but native species to sandy grasslands for the afforested plantations. The selected afforested shrub *Caragana* plantations, both artificial and natural, accounted for an area of ca. 24.3% of the county and had a mean density of 915–3015 plant hm^{-2}. Through several decades of effort, the percentage of severely desertified land covered by sands in this county dropped by 64.6% from 1989 to 2003. During the stabilization of mobile sand land, an intermingled artificial shrubland was established within sandy grasslands in Ningxia, northwestern China.

2.3. Experimental design

The retrospective approach (Bhojvaid and Timmer, 1998; Su and Zhao, 2003) was adopted in this study because of the availability of closely located *Caragana koushinskii* plantations established 6, 15, 24, and 36 years ago on sandy soils (Fig. 1). It was assumed that the soils of each site were similar prior to planting shrubs in the mobile sand land. The plantations, therefore, provided a time series of differing lengths of shrub occupancy on similar sites. Changes in ground-active arthropod diversity and in environmental factors were measured by comparing the shrubland of different ages with the mobile sand land, taking the adjacent naturally restored grassland as a reference that was under exclosure for 23 years. Within each treatment site, three replicate 10 ha plots 200 m apart from each other were selected as the sampling sites.

At the center of each sampling plot, four points were selected and set with about 20 m away from each other. Because the afforested shrubs was arranged in belt (the details see Liu et al., 2013), the shrubs within every two belts were marked as sampling points (beneath the shrub canopy). The long distance between the two selected shrub belts (>24 m, one every two belts) was selected in order to ensure trap sampling independence.

Since the sampling points from one of the three plots in the afforested shrubland of 24 years were destroyed by heavy grazing during the sampling period, this plot was deleted from the data analysis. Totally, the data from 17 replicate plots were obtained for this study, i.e., 3 replicate plots for mobile sand land, 6, 15, and 36 years of shrubland, and naturally restored grassland respectively, and 2 replicate

ones for 24 years of shrubland. All the field work was carried on in August, 2013.

2.4. Ground-active arthropod sampling

At each sampling point, a pitfall trap (7 cm in diameter, 10 cm in depth) filled with approximately 70 ml of 70% ethanol solution were buried, with the opening of the trap level with the ground surface. Traps were checked every three days during the sampling period and fresh ethanol solution was added when needed. Specimens were preserved in 75% ethanol and identified to family and order level using relevant literatures (Yin, 2001; Zheng and Gui, 2004). Totally, for each replicate plot, four traps were collected for the ground-active arthropod determination.

2.5. Plant and soil property determination

At each sampling point close to traps mentioned above, herbaceous vegetation was sampled for the determination of plant density (individuals m^{-2}), richness, and height (cm), by using a quadrat of 50 cm × 50 cm area. For the afforested shrubland site, shrub height (cm) and crown diameter (m^2) were measured by using a tape. Similarly, a soil core (stainless steel cylinders with a volume of 100 cm^3) was collected from 0 to 10 cm soils in order to determine soil bulk density (BD, g cm^{-3}). Nearby, composite soil samples were obtained for the determination of soil physic-chemistry analysis, including soil particle-size distribution (0.10 < coarse sand content < 2 mm [CS], 0.05 mm < fine sand content < 0.10 mm [FS], clay plus silt content <0.05 mm [CPS], %), temperature (ST, °C), water content (SW, %), soil pH, and total nitrogen content (TN, g kg^{-1}) and organic carbon content (OC, g kg^{-1}).

Soil particle-size distribution was determined by the pipette method in a sedimentation cylinder using sodium hexametaphosphate as the dispersing agent (ISSCAS, 1978). Soil bulk density and water content were determined gravimetrically by drying the soil samples at 105 °C for 48 h. Soil pH was determined in a 1:1 soil–water slurry. Soil temperature was determined at depths of 0–10 cm during the sampling phase using a portable thermometer (Sato Keiryoki Mfg. Co. Ltd., Tokyo, Japan). Soil organic carbon was measured using the $K_2Cr_2O_7$–H_2SO_4 oxidation method of Walkey and Black, and soil total nitrogen followed the Kjeldahl procedure (UDK 140 Automatic Steam Distilling Unit, Automatic Titroline 96, Italy).

Values of variables from these sources, such as herbaceous vegetation characteristics (i.e., plant density, species richness and height) and soil physico-chemistry properties (i.e., soil coarse sand content, fine sand content, clay plus silt content, soil temperature, water content, bulk density, pH, and total nitrogen and organic carbon content) were used to explore the relative contributions of different environmental factors to arthropod composition in the current study.

2.6. Data analysis

Within each replicate plot, the trap contents from the four sampling points were pooled together as one sample in order to enrich the data used for the multivariate analysis of variance. Then the abundance of each taxonomical group was obtained for each plot. Based on the total abundance (number of individuals per traps) and taxonomic richness (the total number of taxonomic groups recorded per traps), the Shannon index and Simpson index were calculated.

All statistical analyses were performed through replicate sites using SPSS 16.0 for Windows (SPSS Inc., Chicago, Illinois). Least significant difference (LSD) tests and Post Hoc Multiple comparisons were used to test differences between means for each class within each factor. Before applying parametric tests, we tested for normality and homogeneity of variances. For all tests, statistically significant differences were assigned at $p < 0.05$.

Ordination techniques were used to determine the relative contribution of the measured environmental variables to the community composition (Lepš and Šmilauer, 2003). The taxa abundance data were first analyzed by detrended correspondence analysis (DCA) to decide between linear and unimodal response model for arthropod community data. The length of the first DCA ordination axis was 3.02 (for taxonomic group data), suggesting that redundancy analysis (RDA) was an appropriate approach (i.e., length of gradient < 4). Thus, we used RDA to correlate each arthropod group with the environmental variables by selecting the linear combinations of environmental variables that gave the smallest residual sum of squares. DCA and RDA were carried out using CANOCO software for Windows 4.5 (Microcomputer Power, Itha-ca, USA).

Analysis was performed with default settings. Forward selections were performed to test which factor(s) had significant influence on the arthropod community structure. The selection procedures were stopped when the factor to be added was not significant anymore (Li et al., 2014). Before RDA, a Hellinger transformation was applied to remove the issues of double-zeros in the data matrix and improve the analysis. The data and Monte Carlo reduced model tests with 499 unre-stricted permutations were used to statistically evaluate the significance of the first canonical axis and of all canonical axes combined. In order to meet the requirements of the Monte Carlo reduced model test, all taxonomic group data were square root-transformed.

3. Results

3.1. Composition and abundance of taxonomical groups

A total of 37 taxonomic groups from 8 orders and 34 families plus additional three larval groups of ground-active arthropods were collected in the sampling sites during the study period (Table 1). 9, 13, 16, 14, 15 and 19 taxonomical groups were captured in the mobile sand land, the shrubland of 6, 15, 24 and 36 years, and naturally restored grassland, respectively. The overall ground-active arthropod assemblage was found to be dominated in terms of abundance by Labiduridae (Dermaptera, 12.85% of the total), Tenebrionidae (Coleoptera, 48.96%), and Formicidae (Hymenoptera, 11.61%) in addition to subdominant Carabidae and Melolonthidae (Coleoptera, 6.11% and 8.56% respectively), which together comprised 88.09% of the total number of individuals. Only seven (additional two families Elateridae and Curculionidae, Coleoptera) of the 37 taxonomical groups had the number of individuals exceeding 1% of the total. However, the other 30 taxonomical groups made up only 8.73% of the totals.

Within ground-active arthropod assemblage, the seven main taxonomical groups responded differently to changes in habitat types (Table 1). Three taxonomical groups (i.e., Family Labiduridae, Carabidae and Tenebrionidae) showed a consistent decrease in abundance from the mobile sand land to the afforested shrublands, with

Table 1

The abundant distribution (mean ± SE, individuals trap^{-1}) of taxonomical groups of ground-active arthropods in each treatment site. M, 6a, 15a, 24a, 36a and G underlined mean mobile sand land, afforested shrubland of 6, 15, 24 and 36 years, and natural grassland, respectively. SE following the sites represents standard error of mean ($n = 3$ except for 24a [$n = 2$]). Predators Pr, herbivores Ph, and detritivores De, omnivores Om.

Taxa	Trophic	M	6a	15a	24a	36a	G	F
Phalangudae	Pr	0	0.6 ± 0.3	0.6 ± 0.6	0	1.0 ± 0.6	0	1.15^{ns}
Araneidae	Pr	0	0	0	0.5 ± 0.5	0	0	1.94^{ns}
Liocranidae	Pr	0	0	1.0 ± 1.0	0.5 ± 0.5	0.6 ± 0.6	0.1 ± 0.1	0.59^{ns}
Philodromidae	Pr	0	0	0	0.5 ± 0.5	0.6 ± 0.3	0.6 ± 0.3	1.87^{ns}
Gnapphosidae	Pr	0.3 ± 0.3	0.3 ± 0.3	0.6 ± 0.3	0	0	0	1.16^{ns}
Clubionidae	Pr	0	0	0	0	0	0.1 ± 0.1	0.91^{ns}
Thomisidae	Pr	0	0	0.3 ± 0.3	0	0	0.1 ± 0.1	0.86^{ns}
Lycosidae	Pr	0	0.3 ± 0.3	0	0.5 ± 0.5	0.6 ± 0.3	0	1.47^{ns}
Salticidae	Pr	0.3 ± 0.3	0	0	0	0	0	0.91^{ns}
Tettigoniidae	Ph	0	0	0	0	0	0.1 ± 0.1	3.62*
Aphididae	Ph	0	0	0	0	0	0.1 ± 0.1	0.91^{ns}
Adult Scutelleridae	Ph	0	0	0	0	2.6 ± 1.7	0.1 ± 0.1	2.02^{ns}
Adult Pentatomidae	Ph	0	0	0	0.5 ± 0.5	0	0	1.94^{ns}
Adult Miridae	Ph	0	0	0	0.5 ± 0.5	0	0	1.94^{ns}
Adult Lygaeidae	Ph	0	0	0.6 ± 0.6	0	1.6 ± 0.8	0.2 ± 0.1	1.90^{ns}
Adult Pyrrhocoridae	Ph	0	0	0	0	0	0.1 ± 0.1	0.91^{ns}
Adult Labiduridae	Pr	38.7 ± 23.7a	0.6 ± 0.3b	0 ± 0b	0 ± 0b	1.0 ± 0.5b	0 ± 0b	2.37*
Adult Carabidae	Pr	6.6 ± 1.2a	0.6 ± 0.6b	3.3 ± 3.3b	2.0 ± 0.0b	6.0 ± 2.1ab	1.2 ± 0.2b	5.35*
Adult Teneberionidae	Ph	99.0 ± 52.3a	16.3 ± 4.4b	8.6 ± 2.8b	19.5 ± 1.5b	15.0 ± 3.5b	1.7 ± 0.5b	2.56*
Adult Meloidae	Ph	0	0	0.3 ± 0.3	0	0	0	0.91^{ns}
Adult Chrysomelidae	Ph	0	0	0	0	0	0.1 ± 0.1	0.91^{ns}
Adult Buprestidae	Ph	0.3 ± 0.3	0	0	0	0	0	0.91^{ns}
Adult Elateridae	Ph	0 ± 0b	3.0 ± 3.0a	1.3 ± 1.3a	0 ± 0b	0.6 ± 0.6a	0 ± 0b	2.32*
Adult Melonthidae	Ph	0.3 ± 0.3b	12.7 ± 4.3a	5.3 ± 2.6b	1.5 ± 0.5b	5.6 ± 1.7ab	1.8 ± 0.5b	3.73*
Adult Glaphyridae	Ph	0	2.3 ± 1.9	0.6 ± 0.6	0	0	0	1.22^{ns}
Adult Histeridae	De	0	0.6 ± 0.6	0	0	0	0	0.91^{ns}
Adult Silphidae	De	0	0	0.3 ± 0.3	0	0	0.3 ± 0.3	0.71^{ns}
Adult Curculionidae	Ph	0	2.0 ± 1.1	0.3 ± 0.3	1.0 ± 1.0	2.0 ± 2.0	0	0.83^{ns}
Adult Dermestidae	De	1.0 ± 0.6	0	0.6 ± 0.3	0	0	0	2.36^{ns}
Adult Cassidae	Ph	0	0	0	0.5 ± 0.5	0	0	1.94^{ns}
Larval Chrysomelidae	Ph	0	0	0	0	0	0.1 ± 0.1	3.62*
Larval Meloidae	Ph	0	0	0	0	0	0.1 ± 0.1	3.62*
Larval Lepidoptera	Ph	0	0	0	0.5 ± 0.5	0.3 ± 0.3	0.1 ± 0.1	1.04^{ns}
Chalcidoidea	Ph	0	0	0	0	0.3 ± 0.3	0	0.91^{ns}
Sphecidae	Ph	0	0.3 ± 0.3	1.6 ± 0.3	0.5 ± 0.5	0	0.1 ± 0.1	7.25**
Pompilidae	Ph	0.3 ± 0.3	0	0	0	0	0	0.91^{ns}
Formicidae	Om	0 ± 0b	8.3 ± 3.8ab	8.0 ± 6.5ab	2.5 ± 1.5ab	4.0 ± 2.1ab	14.5 ± 3.1a	1.95*

ns means no significant differences.

* $p < 0.05$.

** $p < 0.01$.

gains of no changes as the afforested plantation aged that was close to the naturally restored grassland. Whereas, two taxonomical groups (i.e., Family Melolonthidae and Formicidae) showed a consistent increase in abundance from the mobile sand land to the afforested shrublands, with gains of no changes as the afforested plantation aged that was considerably lower in comparison to the naturally restored grassland. The other two taxonomical groups (i.e., Family Elateridae and Curculionidae) were found only in the afforested shrublands, whereas no individuals were found in the mobile sand land and naturally restored grassland.

3.2. Abundance and richness of functional groups

Due to only one taxonomical group (i.e., Formicidae) captured for the omnivores, the taxa density and richness was omitted here. The activity density and taxa richness of the other three functional groups (i.e., predators, herbivores and detritivores) was differentially affected by the afforested shrubland in the mobile sand land (Table 1; Fig. 2a). Compared to the mobile sand land, total predator and herbivore activity density decreased significantly ($p < 0.05$), by over 78% and 63% respectively in the afforested shrubland but that was close to the naturally restored grassland (Fig. 2a). Whereas, no significant ($p > 0.05$) differences in total detritivores activity were found between all these treatment sites including the naturally restored grassland. Likewise, compared to the mobile sand land, the taxa richness of herbivores was found to be marginally ($p = 0.08$) higher in the afforested shrubland but that was also close to the naturally restored grassland. Whereas, no significant ($p > 0.05$) differences in taxa richness of either predators or detritivores were found between all these treatment sites including the naturally restored grassland (Fig. 2b).

3.3. Community indices of ground-active arthropods

The manually afforested shrubland had a marked effect on the total abundance ($p < 0.01$), taxa richness ($p = 0.07$), and the Shannon index ($p < 0.001$) and Simpson index ($p < 0.01$) (Fig. 3). Total abundance and Simpson index showed a consistent decrease from the mobile sand land to the afforested shrublands, with gains of no significant ($p > 0.05$) changes as the shrublands aged though that was close to the naturally restored grassland. In contrast, the group richness and Shannon index showed a consistent increase from the mobile sand land to the afforested shrublands. Noticeably, the taxa richness showed similar values as the shrubland aged that was also close to the naturally restored grassland, whereas the Shannon index in the shrubland of 24 years that was close to the naturally restored grassland showed significantly ($p < 0.05$) lower values in comparison to the other three shrubland of 6, 15 and 36 years.

3.4. Contribution of environmental properties to community structure of ground-active arthropods

RDA showed that the first two axes explained 73.4% (48.3% by Axis 1 and 11.2% by Axis 2) of the variation in the community composition by environmental parameters (Table 2; Fig. 4). The first axis (Monte Carlo permutation test: $F = 3.74$, $p = 0.02$) showed a clear environmental gradient of increased soil temperature (correlation coefficient, $r = 0.72$, $p < 0.05$) and decreased plant species richness ($r = -0.67$, $p < 0.05$), which separated the ground-active arthropod community between the mobile sand land with higher soil temperature and the afforested shrubland sites in addition to naturally restored grassland with higher plant species richness (Fig. 4).

Although all examined 12 environmental variables explained 82.5% of the variation in the community composition, pRDA showed that of these 12 variables, only the conditional effects of soil temperature and pH were significant in the Monte Carlo permutation test ($F = 5.51$, $p < 0.01$ in ST and $F = 3.71$, $p = 0.01$ in pH). Of the total 82.5% variation explained by the RDA, 27% was caused by ST, 15% by pH and the rest (40.5%) by variables that were not significant in the Monte Carlo permutation test (Table 2). The RDA graph also showed that three taxonomical groups (i.e., family Labiduridae, Carabidae and Tenebrionidae) were characteristic of the mobile sand land, whereas two taxonomical groups (i.e., family Melolonthidae and Formicidae) were characteristic of the afforested shrubland in addition to naturally restored grassland (Fig. 4).

4. Discussion

In desertified regions, the afforested shrubs in the mobile sand land could act as "fertilize island" and caused habitat heterogeneity (Sarig and Steinberger, 1994), which had strong influences on the arthropod composition and abundance distribution (J.L. Liu et al., 2012; Liu et al., 2012a,b). In this study, three contrasting response assemblies were identified from the responses of the seven main ground-active arthropod families to changes in habitat types. For example, three arthropod families (i.e., Labiduridae, Carabidae and Tenebrionidae) and two arthropod families (i.e., Melolonthidae and Formicidae) showed consistent but contrasting changes (the first group in decrease, while the second in increase) in abundance from the mobile sand land to the afforested shrubland, while the rate of changes, whether increase or decrease, was similar between the shrubland of different ages. However, the two families (i.e., Elateridae and Curculionidae) were found only in the afforested shrublands. More importantly, there were distinctive responses of these three taxonomical groups to the shrubland and naturally restored grassland. These family-specific responses indicated that the microhabitat preferences of ground-active arthropods could be expressed at the family level, and that some of these main families can be used as reliable bioindicators of environmental change generated by habitat changes as resulted from the afforested shrub plantations
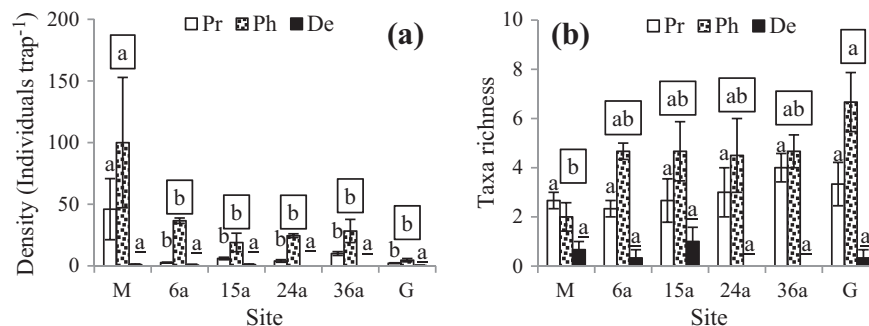

Fig. 2. The activity density (a) and taxa richness (b) of three functional groups (mean ± SE, predators [Pr], herbivores [Ph] and detritivores [De]) of ground-active arthropods in each treatment site. Values with different letters (within boxes and underlined referring to Ph and De, respectively) are significantly different at $p < 0.05$.
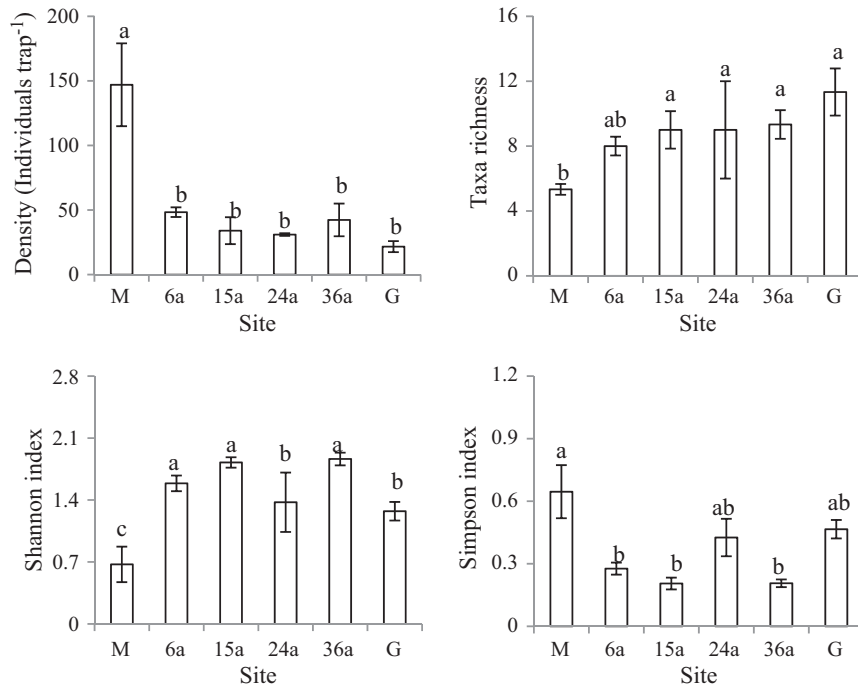
Fig. 3. Total abundance (mean ± SE), taxa richness, Shannon index and Simpson index of ground-active arthropods in each treatment site. Values with different letters are significantly different at $p < 0.05$.

(Heino and Soininen, 2007; Vieira et al., 2012). As confirmed, the Biplot of the redundant analysis (RDA) showed that three families (i.e., Labiduridae, Carabidae and Tenebrionidae) preferred the mobile sand land with high temperature and low vegetation cover (i.e., obstacles) (J.L. Liu et al., 2012; Liu et al., 2012a,b). For instance, the tenebrionid assemblage has been known to be xerophilic and characteristic of desert steppe sites with higher soil temperature and coarsed sand texture (Li et al., 2014; Ren and Yu, 1999). Additionally, the dominant family Tenebrionidae had the strong capability of dispersals and could move freely for food resources without obstacles in mobile sand land (J.L. Liu et al., 2012; Liu et al., 2012a,b). In contrast, the two families Melolonthidae and Formicidae showed remarkably positive response to the afforested shrubland and naturally restored grassland. Especially in the naturally restored grassland, there were considerably higher abundance relative to either of the afforested

shrubland. This result was correlated with the diverse food resources provided by the recovery of vegetation cover, particularly in the naturally restored grassland with high herbaceous diversity (Liu et al., 2014).

Fig. 4. Biplot of the first two RDA axes for showing the association of ground-active arthropod community composition with environmental variables. M, 6a, 15a, 24a, 36a and G underlined mean mobile sand land, afforested shrubland of 6, 15, 24 and 36 years, and naturally restored grassland, respectively. The capitals mean environmental parameters: CS coarse sand content, FS fine sand content, CPS clay plus silt content, ST soil temperature, SW soil water content, BD bulk density, TN total nitrogen content, OC organic carbon content, PD herbaceous plant density, PS plant species richness, VC plant height. The numbers close to the arrows mean the taxonomical groups: 1 Phalangudae, 2 Araneidae, 3 Liocranidae, 4 Philodromidae, 5 Gnapphosidae, 6 Clubionidae, 7 Thomisidae, 8 Lycosidae, 9 Salticidae, 10 Tettigoniidae, 11 Aphididae, 12 Scutelleridae, 13 Pentatomidae, 14 Miridae, 15 Lygaeidae, 16 Pyrrhocoridae, 17 Labiduridae, 18 Carabidae, 19 Adult Teneberionidae, 20 Meloidae, 21 Chrysomelidae, 22 Buprestidae, 23 Elateridae, 24 Melolonthidae, 25 Glaphyridae, 26 Histeridae, 27 Silphidae, 28 Curculionidae, 29 Dermestidae, 30 Cassidae, 31 Larval Chrysomelidae, 32 Larval Meloidae, 33 Larval Lepidoptera, 34 Chalcidoidea, 35 Sphecidae, 36 Pompilidae, 37 Formicidae.

Table 2
Redundancy analysis for quantifying the conditional effects of the environmental factors on the ground-active arthropod community composition. CS coarse sand content, FS fine sand content, CPS clay plus silt content, ST soil temperature, SW soil water content, BD bulk density, pH, TN total nitrogen content, OC organic carbon content, PD herbaceous plant density, PS plant species richness, VC plant height.

Factors	Inertia conditional effects	F	P	MCR (%)
ST	0.27	5.51	0.002**	27
pH	0.15	3.71	0.006**	15
CS	0.06	1.42	0.186	6
PS	0.05	1.33	0.214	5
FS	0.04	1.15	0.266	4
BD	0.04	1.15	0.326	4
SW	0.04	1.10	0.306	4
VC	0.04	1.12	0.324	4
PD	0.03	0.71	0.600	3
CPS	0.02	0.39	0.836	2
OC	0.02	0.36	0.814	2
TN	0.06	1.42	0.272	6
Total				82

The inertia from the conditional effects is used to calculate the multivariate correlation ratio (MCR), which represents the proportion of variation explained by the individual environmental factors. The sum of all eigenvalues = 1.000.
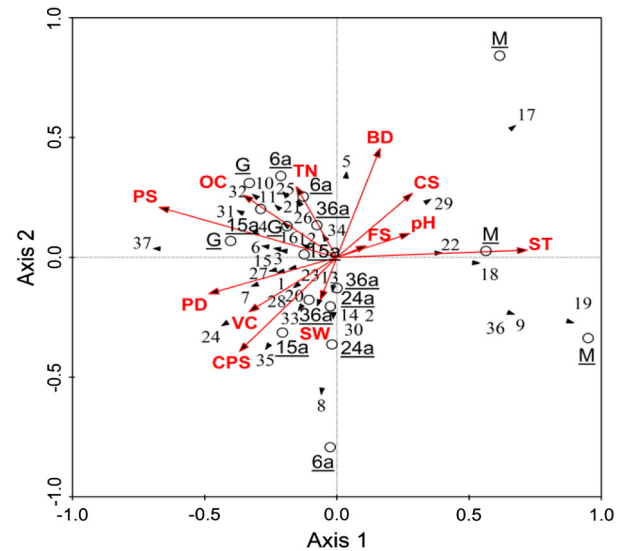
The habitat types due to the afforested shrubland in mobile sand land also differed across trophic groups. The results of the multi-trophic studies support this result. These shifts largely reflected the changes in the availability of resources in each habitat (Wardhaugh et al., 2012). For instance, the afforested shrubland had negative effect on the density of herbivores and predators as both functional groups had significantly lower frequency of activity and density in the afforested shrubland compared to the mobile sand land. In our study, the decrease in herbivore activity density in the afforested shrubland sites was mainly caused by the decreased activity density of herbivorous Tenebrionidae beetles (i.e., Blaps and Anatolica), as mentioned above by the altered abiotic conditions such as soil temperature and soil texture. In addition, the mono-shrub (C. koushinskii) selected as the afforested plantations was another important reason. According to J.L. Liu et al. (2012) and Liu et al. (2012a,b), a high diversity of shrub species was important for the maintenance of a diverse tenebrionid beetle assemblage because of food sources from different shrub species. Contrastingly, the simplified food resources provided by the monophyletic C. koushinskii shrubs and more obstacles from the recovery vegetation cover could limit the movement of tenebrionid beetles and decrease the abundance distribution in the afforested shrubland (J.L. Liu et al., 2012; Liu et al., 2012a,b). The decrease in predator activity density in the afforested shrubland sites was mainly caused by the decreased activity density of predator Labiduridae and Carabidae. Besides the altered abiotic conditions discussed above, another reason was that the potential prey from the decreased herbivore density declined correspondingly during this conversion process. This was a reflective of "bottom-up control" on food web structure in the study systems (Liu et al., 2011).

In contrast, the afforested shrubland had positive effect on the richness of herbivores as this functional group had considerably higher taxa richness in the afforested shrubland compared to the mobile sand land. The increase in herbivore taxa richness in the afforested shrubland sites was mainly caused by the increased taxa richness of diverse herbivorous arthropod families. Along with the stabilization of mobile sand land by the afforested shrub plantations, the improved herbaceous vegetation and soil properties (J.L. Liu et al., 2012; Liu et al., 2012a,b) was an attractive of diverse ground-active herbivorous families dwelling in the soils of afforested shrubland (Liu et al., 2013). The considerably higher herbivore activity richness in the naturally restored grassland in comparison to the afforested shrubland further confirmed this result (J.L. Liu et al., 2012; Liu et al., 2012a,b). Compared with the mobile sand land the herbivore taxa richness was two times more in the afforested shrubland, even three times more in the naturally restored grassland (Fig. 1). The diverse herbivorous arthropod families and increased herbivore taxa richness acting as primary consumers could maintain the development and stabilization of food web structure on the one hand, whereas on the other hand, this increased herbivore taxa richness enhanced the risk of pest exposure on forage grass in our system (Li et al., 2014). More attention should be paid to the future conservation and management on this afforested shrub plantations. However, the detritivorous activity density and richness showed no changes between each treatment site including the mobile sand land and the afforested shrubland of different ages, indicating a very low value in this study. This result was correlated with the lack of litter on the ground that was a necessity for detritivorous arthropods (Lavelle and Spain, 2001).

Interestingly, the activity density of each functional group was similar between the afforested shrubland of different ages, and a similar pattern was followed by the taxa richness of these functional groups. Although the soil properties and herbaceous vegetation changed much along the chronosequences of afforested shrubland (J.L. Liu et al., 2012; Liu et al., 2012a,b), the similar responses of these two functional groups to the shrubland of different ages might attribute to the mono-shrub plantations that dominated the desertified grassland and showed similar consequences on the activity density and richness of functional groups. Furthermore, the activity density and richness of all these

functional groups (except the richness of herbivores as discussed above) in the afforested shrubland was close to that in the naturally restored grassland. It was suggested that the afforested shrubland might have the similar ecological implications on the food web structure in comparison to the naturally restored grassland in desertified regions. Whereas this conclusion should be attained with caution, since the afforested shrubland differed much from the naturally restored grassland on many disciplines such as soil ecological process, soil water flux, forage grass stock and so on (Bhojvaid and Timmer, 1998; Cushman et al., 2010; Wang et al., 2011).

Together, the observed strong shifts in the taxonomic and trophic composition of ground-active arthropods following the conversion of mobile sand land to afforested shrubland, could significantly alter ground-active arthropod community structure. In this study, the decreased total abundance and Simpson index of ground-active arthropods were mainly caused by the decreased predator and herbivore activity density acting as the dominated arthropod groups during this conversion process. Whereas, the increased taxonomical richness and Shannon index of ground-active arthropods were mainly caused by the increased herbivore taxa richness. The taxonomical richness and Shannon index showed a similar trend to the herbivore taxa richness from the mobile sand land to the afforested shrubland of different ages. On the one hand, the multivariate analysis revealed that changes in the community structure were largely mediated by changes in environment parameters (i.e., soil temperature, pH and herbaceous richness) because a major part (83%) of the variation in the community structure was explained by variations in the examined nine soil physicochemical variables, including soil texture, temperature, bulk density, water content, total nitrogen, organic carbon content, and pH, and potential availability of food resources of herbaceous vegetation (plant density, species richness and height). This was consistent with the findings of previous studies in other ecosystems demonstrating that human-induced changes in soil pH, soil texture, soil moisture and temperature, and resource availability together played important roles in structuring soil faunal communities (Byrne et al., 2008; Huhta and Hanninen, 2001; Lindberg et al., 2002; Zhao et al., 2011). In addition, the results from pRDA analysis further elucidated that soil temperature were the most important factor determining the ground-active arthropod composition as it contributed to the major part (27%, MCR) of the total variation explained by the RDA. A study of the response of ground-active arthropods to the shrub patches in other arid desert ecosystems also found that ground-active arthropod assemblages responded strongly to changes in soil temperature conditions (Perner and Malt, 2003; Liu et al., 2014). Previous studies suggested that soil temperature was one of the most limiting factors of the ecological processes that affected the behavior, egg laying and daytime refuge of soil animals in arid and semiarid ecosystems (Austin et al., 2004; J.L. Liu et al., 2012; Liu et al., 2012a,b; Noy-Meir, 1973). Therefore, the effect of the afforested shrubland on the ground-active arthropod community was strongly mediated by the effect on soil temperature microhabitats resulting from the change of aboveground vegetation type (i.e., shrub canopy) and biophysical properties (Zhao and Jackson, 2014).

Nevertheless, the Shannon index was significantly lower in the afforested shrubland of 24 years compared with the other three shrubland habitat (i.e., shrubland of 6, 15 and 36 years). The 24 year-old shrubland was being subjected to grazing pressure during the study period in practice though it was previously under exclosure from livestock grazing. Livestock grazing could negatively affect arthropods (Kruess and Tscharntke, 2002; Liu et al., 2013), and sufficiently grazing pressure was expected to produce negative, even detrimental, effects on arthropod assemblages even after the grazing has stopped (Holmquist et al., 2013). The trampling by livestock was detrimental to the egg survival, and fecund, oviposition and life cycles of ground-active arthropods, and the time was long (Lenoir and Lennartsson, 2010). Therefore, this relationship was intriguing and deserved more attention for the management of manually afforested plantations in arid regions. Noticeably,

the Shannon index of ground-active arthropods in the afforested shrubland of 24 years was close to that in the naturally restored grassland, both suggesting significantly lower values in comparison to the other three shrubland habitats (i.e., shrubland of 6, 15 and 36 years). It was suggested that the grazing management could improve the effectiveness of afforested shrubs in the mobile sand land on arthropod diversity recovery relative to the naturally restored grassland. A growing number of empirical studies demonstrated positive diversity-stability relationships (Ives and Carpenter, 2007). These results implied that the manually afforested shrubland in the mobile sand land, especially through grazing management, was an effective and option strategy for biological diversity conservation, desertification control and the recovery process of desertified grassland ecosystems.

5. Conclusions

Our study provided several insights into the consequences of the manually afforested shrubland in the mobile sand land on the activity taxonomic and trophic groups, and community diversity of ground-active arthropods through altered microclimate, soil properties and resource distribution. The first insight was that the conversion of mobile sand land to afforested shrubland resulted in major changes in the activity taxonomic and trophic structure, and facilitated the herbivore richness and diversity of ground-active arthropods with important consequences for the ecological process, function and services of soil ecosystems. The second insight was that no differences were found in the activity taxonomic and trophic structures and community diversity between the afforested shrubland of different ages (except for the Shannon diversity in the shrubland of 24 years). It was suggested that mono-shrub plantations afforested in the mobile sand land had similar effects on ground-active arthropods, regardless of shrub ages. The third insight was that the livestock grazing in the afforested shrubland was found to indicate negative influences on the ground-active arthropod diversity recovery. The removed grazing in the afforested shrubland could improve the effectiveness of arthropod diversity recovery relative to the naturally restored grassland. All these findings highlighted the importance of detailed knowledge of understanding how the afforested shrub plantations affected the functional structure and diversity of ground-active arthropods, and could help develop more effective management strategies on the management of shrubland, desertification control, and the recovery and conservation of desertified grassland ecosystem.

Acknowledgements

References

Austin, A.T., Yahdjian, L., Stark, J.M., Belnap, J., Porporato, A., Norton, U., Ravetta, D.A., Schaeffer, S.M., 2004. Water pulses and biogeochemical cycles in arid and semiarid ecosystems. Oecologia 141, 221–235.

Bhojvaid, P.P., Timmer, V.R., 1998. Soil dynamics in an age sequence of Prosopis juliflora planted for sodic soil restoration in India. For. Ecol. Manag. 106, 181–193.

Byrne, L.B., Bruns, M.A., Kim, K.C., 2008. Ecosystem properties of urban land covers at the aboveground–belowground interface. Ecosystems 11, 1065–1077.

Cushman, J.H., Waller, J.C., Hoak, D.R., 2010. Shrubs as ecosystem engineers in a coastal dune: influences on plant populations, communities and ecosystems. J. Veg. Sci. 21, 821–831.

Heino, J., Soininen, J., 2007. Are higher taxa adequate surrogates for species level assemblage patterns and species richness in stream organisms? Biodivers. Conserv. 137, 78–89.

Holmquist, J.G., Schmidt-Gengenbach, J., Haultain, S.A., 2013. Effects of a long-term disturbance on arthropods and vegetation in subalpine wetlands: manifestations of pack stock grazing in early versus midseason. PLoS One 8, e54109.

Huhta, V., Hanninen, S.M., 2001. Effects of temperature and moisture fluctuations on an experimental soil microarthropod community. Pedobiologia 45, 279–286.

Institute of Soil Sciences, Chinese Academy of Sciences (ISSCAS), 1978. Physical and Chemical Analysis Methods of Soils. Shanghai Science Technology Press, Shanghai, China.

Ives, A.R., Carpenter, S.R., 2007. Stability and diversity of ecosystems. Science 317, 58–62.

Jiang, D.M., Li, Q., Liu, F.M., Jiang, Y., Liang, W.J., 2007. Vertical distribution of soil nematodes in an age sequence of Caragana microphylla plantations in the Horqin Sandy Land, Northeast China. Ecol. Res. 22, 49–56.

Kruess, A., Tscharntke, T., 2002. Contrasting responses of plant and insect diversity to variation in grazing intensity. Biodivers. Conserv. 106, 293–302.

Lavelle, P., Spain, A., 2001. Soil Ecology. Kluwer Scientific Publications, Amsterdam, the Netherlands.

Lenoir, L., Lennartsson, T., 2010. Effects of timing of grazing on arthropod communities in semi-natural grasslands. J. Insect Sci. 10, 1–24.

Lepš, J., Šmilauer, P., 2003. Multivariate Analysis of Ecological Data Using CANOCO. Cambridge University Press, UK, Cambridge, pp. 43–75.

Li, F.R., Zhao, L.Y., Zhang, H., Zhang, T.H., Shirato, Y., 2004. Wind erosion and airborne dust deposition in farmland during spring in Horqin sand land of eastern Inner Mongolia, China. Soil Tillage Res. 75, 121–130.

Li, F.R., Liu, J.L., Sun, T.S., Jin, B.W., Chen, L.J., 2014. Converting natural vegetation to farmland alters functional structure of ground-dwelling beetles and spiders in a desert oasis. J. Insect Conserv. 18, 57–67.

Lindberg, N., Bengtsson, J., Persson, T., 2002. Effects of experimental irrigation and drought on the composition and diversity of soil fauna in a coniferous stand. J. Appl. Ecol. 39, 924–936.

Liu, R.T., Zhao, H.L., Zhao, X.Y., Drake, S., 2011. Facilitative effects of shrubs in mobile sand on soil macro-faunal community in Horqin sand land of Inner Mongolia, Northern China. Eur. J. Soil Biol. 47, 316–321.

Liu, J.L., Li, F.R., Liu, C.A., Liu, Q.J., 2012a. Influences of shrub vegetation on distribution and diversity of a ground beetle community in a Gobi desert ecosystem. Biodivers. Conserv. 21, 2601–2619.

Liu, R.T., Chai, Y.Q., Xu, K., Zhu, F., 2012b. Variations of ground vegetation and soil properties during the growth process of artificial sand-fixing Caragana koushinskii plantations in desert steppe. Chin. J. Appl. Ecol. 23, 2937–2942.

Liu, R.T., Zhao, H.L., Zhao, X.Y., 2012c. Influences of grazing exclusion on soil macro-invertebrate diversity in degraded sandy grassland (Inner Mongolia, China). Pol. J. Ecol. 60, 375–385.

Liu, R.T., Zhao, H.L., Zhao, X.Y., Zhu, F., 2013. Effects of cultivation and grazing exclusion on the soil macro-faunal community of semiarid sandy grasslands in northern China. Arid Land Res. Manag. 27, 377–393.

Liu, R.T., Chai, Y.Q., Xu, K., Zhu, F., 2014. Seasonal changes of ground vegetation characteristics under artificial Caragana koushinskii plantations with age in desert steppe. Acta Ecol. Sin. 34, 500–508.

Mekuria, W., Veldkamp, E., Haile, M., Nyssen, J., Muys, B., Gebrehiwot, K., 2007. Effectiveness of exclosures to restore degraded soils as a result of overgrazing in Tigray, Ethiopia. J. Arid Environ. 69, 270–284.

Noy-Meir, I., 1973. Desert ecosystems: environment and producers. Annu. Rev. Ecol. Evol. Syst. 4, 25–52.

Nyssen, J., Poesen, J., Moeyersons, J., Deckers, J., Haile, M., Lang, A., 2004. Human impact on the environment in the Ethiopian and Eritrean Highlands—a state of the art. Earth-Sci. Rev. 64, 273–320.

Perner, J., Malt, S., 2003. Assessment of changing agricultural land use: response of vegetation, ground-dwelling spiders and beetles to the conversion of arable land into grassland. Agr. Ecosyst. Environ. 98, 169–181.

Ren, G.D., Yu, Y.Z., 1999. The Darking Beetles From Deserts and Semideserts of China (Coleoptera: Tenebrionidae). Hebei University Publishing House, Baoding (in Chinese).

Sarig, S., Steinberger, Y., 1994. Microbial biomass response to seasonal fluctuation in soil salinity under the canopy of desert halophytes. Soil Biol. Biochem. 26, 1405–1408.

Shachak, M., Boeken, B., Groner, E., Kadmon, R., Lubin, Y., Ne'Eman, G., Perevolotsky, A., Shkedy, Y., Ungar, E.D., 2008. Woody species as landscape modulators and their effects on biodiversity patterns. Bioscience 58, 209–221.

Su, Y.Z., Zhao, H.L., 2003. Soil properties and plant species in an age sequence of Caragana microphylla plantations in the Horqin Sandy Land, North China. Ecol. Eng. 20, 223–235.

Titus, J.H., Nowak, R.S., Smith, S.D., 2002. Soil resource heterogeneity in the Mojave Desert. J. Arid Environ. 52, 23–37.

Vieira, L.C., Oliveira, N.G., Brewster, C.C., Gayubo, S.F., 2012. Using higher taxa as surrogates of species-level data in three Portuguese protected areas: a case study on Spheciformes (Hymenoptera). Biodivers. Conserv. 21, 3467–3486.

Wang, X.P., Wang, Z.N., Berndtsson, R., Zhang, Y.F., Pan, Y.X., 2011. Desert shrub stemflow and its significance in soil moisture replenishment. Hydrol. Earth Syst. Sci. 15, 561–567.

Wardhaugh, C.W., Stork, N.E., Edwards, W., 2012. Feeding guild structure of beetles on Australian tropical rainforest trees reflects microhabitat resource availability. J. Anim. Ecol. 81, 1086–1094.

Wardle, D.A., Bardgett, R.D., Klironomos, J.N., Setälä, H., van der Putten, W.H., Wall, D.H., 2004. Ecological linkages between aboveground and belowground biota. Science 304, 1629–1633.

Xu, D.M., Xu, X.Z., Xie, Y.Z., Wang, K., 2012. Dynamics of sandy desertification and detection of sandy land/steppe boundary: vegetation and soil properties. Pol. J. Ecol. 60, 251–263.

Yin, W.Y., 2001. Pictorial Keys to Soil Animals of China. Science Press, Beijing.

Zhao, K., Jackson, R.B., 2014. Biophysical forcings of land-use changes from potential forestry activities in North America. Ecol. Monogr. 84, 329–353.

Zhao, H.L., Liu, R.T., 2013. The "bug island" effect of shrubs and its formation mechanism in Horqin Sand Land, Inner Mongolia. Catena 105, 69–74.

Zhao, H.L., Zhao, X.Y., Zhou, R.L., Zhang, T.H., Drake, S., 2005. Desertification processes due to heavy grazing in sandy rangeland, Inner Mongolia. J. Arid Environ. 62, 309–319.

Zhao, H.L., Zhou, R.L., Su, Y.Z., Zhang, T.H., Zhao, L.Y., Drake, S., 2007. Shrub facilitation of desert land restoration in the Horqin sand land of Inner Mongolia. Ecol. Eng. 31, 1–8.

Zhao, X.Y., Luo, Y.Y., Wang, S.K., Huang, W.D., Lian, J., 2010. Is desertification reversion sustainable in northern China? A case study in Naiman county, part of a typical agro-pastoral transitional zone in Inner-Mongolia, China. Glob. Environ. Res. 14, 63–70.

Zhao, J., Wang, X.L., Shao, Y.H., Xu, G.L., Fu, S.L., 2011. Effects of vegetation removal on soil properties and decomposer organisms. Soil Biol. Biochem. 43, 954–960.

Zheng, L.Y., Gui, H., 2004. Insect Classification. Nanjing Normal University Press, Nanjing.