CHANGES IN CARABID (COLEOPTERA: CARABIDAE) ASSEMBLAGES AND DIVERSITY PATTERNS BETWEEN 1997 AND 2014 IN A DESALINIZED, INTENSIVELY CULTIVATED AGRICULTURAL LANDSCAPE OF NORTHERN CHINA

XIN ZHANG

College of Agricultural Resources and Environmental Sciences, China Agricultural University, Beijing 100094, China

College of Architecture and Art, Hebei University of Architecture, Zhangjiakou 075000, China

PANLONG WU,YUNHUI LIU, YIN HAN, XUZHU ZHANG, PIAOPIAO DAI, ZHENRONG YU

College of Agricultural Resources and Environmental Sciences, China Agricultural University, Beijing 100094, China

Liuyh@cau.edu.cn

AND

JAN C. AXMACHER

UCL Department of Geography, University College London, London, UK

ABSTRACT

Reclamation of salinity-affected land for intensive agricultural production represents a highly promising pathway towards feeding our increasing global population. Nonetheless, biodiversity and ecosystem service responses to agricultural intensification in desalinized landscapes remain poorly understood. In our study, we analyzed long-term diversity responses of carabids as important pest control agents to agricultural intensification in desalinized landscapes by comparing data from 1997 and 2014, and we analyzed the potential role of field-margins as beetle refuge habitats. Despite agricultural intensification, carabid species richness increased significantly following desalinization, with this increase being chiefly limited to field margins. Carabid assemblages also showed a dramatic temporal species turnover, leading towards a species-rich homogeneous community dominated by generalists. Therefore, we believe that desalinization triggered positive regional diversity responses despite simultaneous agricultural intensification, with semi-natural field margins playing an increasingly important role as local diversity hotspots. Nonetheless, the highly uniform composition of the generalist beetle assemblages and its potential implications for ecosystem functioning require further scrutiny, and the targeted management of semi-natural habitats appears crucial to optimize farmland biodiversity and associated ecosystem services in desalinized agricultural landscapes.

Key Words: desalinization, biodiversity, ground beetle, landscape change

Feeding an increasing global human population requires a substantial increase in the arable land area or in agricultural productivity (FAO 2015). As salinized land covers about 7% of the world’s land surface (Li et al. 2014), desalinization of potential agricultural land could be an effective measure to increase the arable land area and associated food production.

Earlier studies indicated that transformations of natural habitats into agricultural fields are highly problematic, since both natural habitat loss and agricultural intensification are well established as main causes for the loss of global biodiversity and ecosystem services, which are essential to sustainable production (MEA 2005; Norris 2008). Salinized land is commonly characterized by a low diversity and productivity, because high soil salinity is known to severely limit plant growth (Dagar 2003; Wu et al. 2015). Therefore, promoting vegetation and productively in the desalinized areas would potentially support increases in overall farmland diversity because of directly and indirectly links between plant species diversity/productivity and diversity of invertebrate taxa (Kareiva, 1983; Siemann,1998), despite having very minor negative implications for biodiversity conservation but a local loss of a small number of salinity-specialist species (Ladã¡Nyi et al., 2016). The contrasting effects desalinization and increasingly intensive agricultural land-use have on species assemblages in desalinized areas that are subsequently experiencing intensive agricultural management are poorly understood (Liu *et al.*, 2006). Agricultural intensification can potentially prevents or slow down adjustments of species assemblages to the improved environmental conditions that follow desalinization.

In intensified agricultural landscapes, semi-natural habitats can provide important additional food resources, refuge, shelter and winter habitats. They therefore are increasingly established and managed to alleviate some of the negative impacts intensive agricultural production has on agricultural biodiversity (Marshall and Moonen 2002; Haaland *et al*. 2011) and on the ecological services provided by agricultural landscapes for example in Europe (Marshall and Moonen 2002; Haaland *et al*. 2011).To date, most studies looking at the role of field margins and similar semi-natural habitats in agricultural landscapes relied on spatial snapshot comparisons between these habitats and the surrounding, heavily managed agricultural fields, while the long-term temporal variations in field-margin assemblages have been widely neglected (Jung et al., 2008; Cole et al., 2012). Addressing this knowledge gap, long-term changes in the role semi-natural habitats play in harboring diverse species assemblages in increasingly heavily managed agricultural areas with dramatically increased inputs of agro-chemicals form a second focus of our study.

China represents a country that faces dramatic challenges in feeding its increasing population on a very limited arable land area. About 5% of the country’s terrestrial land (3.6×10^7 ha) is taken up by saline soils (Wang *et al*. 2011). Of this, 1.3×10^7 ha could potentially be (re-)claimed for cultivation (Li *et al*. 2014; Wang *et al*. 2011). This would increase China’s total arable land area by 10%. In the North China Plain that represents one of the main agriculture regions in China and provides more than 75% of the national wheat and 35% of the national maize harvest, about 10% of the land area has been affected by high levels of soil salinity (Xin and Li, 1990). During the 1970s and 1980s, China made great progress in managing and decreasing salinity of potential agricultural land. In the process, large areas were desalinized to allow for a greatly enhanced production of agricultural crops, and an intensive management regime became established across desalinized regions to meet the country’s increasing food demand (Wang *et al*. 2011; Xin and Li 1990).

In 1997, we investigated the diversity and species composition of carabid beetles at field margins and in fields of three formerly salinity-affected districts desalinized in 1973, 1978 and 1982, respectively. Carabidae (Coleoptera) were chosen because they can be sampled in a highly standardized approach using pitfall traps, are easy to preserve and to identify, react sensitively to environmental change and are a taxon providing important pest control functions in agricultural landscapes (Lövei and Sunderland 1996; Thiele 2012). We recorded a very low carabid diversity across the desalinized landscape, with a slightly elevated carabid diversity at field margins, and a significant effect of plant richness, soil salt content and nitrogen content on the composition of carabid assemblages encountered at field margin habitats (Liu *et al*. 2006). In 2014, seventeen years after the initial study, and after the study region had experienced significant agriculture intensification that also dramatically altered the agricultural landscape pattern, we re-sampled the carabid assemblages of fields and margins at the three desalinized districts, aiming to establish the carabid community response patterns to the agricultural intensification experienced across the desalinized landscape. We focused on the following three questions in the reclaimed desalination landscape: (1) How has biodiversity changed over long time-periods of agricultural intensification? (2) What role do extensively managed field margins play in supporting landscape-scale ground beetle diversity following this agricultural intensification? (3) How do environmental conditions at field margins affect the carabid species composition at these habitats over time?

MATERIAL AND METHODS

Study Area and Site Selection. The research was conducted at Quzhou county (36°36'-36°58'N, 114°50'-115°13'E) in Hebei province at the center of the North China Plain, characterized by a temperate semi-humid continental monsoon climate with an annual mean temperature of 14.1 °C (ranging from 13.0 to 15.4 °C) and an average annual precipitation of ~483 mm (ranging from 219 to 792 mm; National Meteorological Bureau, period 1994-2014). Before the 1980’s, high levels of soil salinity caused by shallow saline groundwater represented serious problems for the region's agricultural production.

Salinity-affected agricultural areas in Quzhou county were therefore desalinized in three stages, involving different villages or ‘experimental districts’ (EDs). This process started in 1973 with ED1 (Zhangzhuang village), followed 1978 by ED2 (Wangzhuang village) and 1982 by ED3 (Situan village). A series of measures, including digging ditches to improve drainage, planting shelterbelts to reduce evapotranspiration, irrigating with fresh water, were taken to improve the leaching of soil salt. Organic manure was simultaneously applied to improve soil fertility (Xin and Li 1990). After successful desalinization, crop cultivation on the land gradually intensified, eventually resulting in a homogenous vegetation and soil conditions across all EDs (Table 1), and in productivity levels comparable to areas unaffected by soil salinity. In 2014, cultivated land covered >71% of the county’s land area, with very low proportions of semi-natural habitats like field margins and woodland remaining in the agricultural landscape (accounting for 1.24% of the total area in 1986 and 1.05% in 2000). The increase in agricultural production in the wider study area is exemplified by the increase in annual nitrogen fertilizer inputs in Hebei province from 0.78×10^5 t in 1987 to 1.52×10^5 t in

2012, and by the average yield of winter wheat increasing from 3080 kg/ha in 1987 to 5559 kg/ha in 2012 (NBSPRC 2015).

In 1997, carabid beetles were sampled at a total of 30 plots selected at 10 sites in ED3, with one plot at each site situated within a field margin (FM), 10 m inside the field (‘near field margin’ -NFM) and 30 m inside the field (‘far from field margin’ - FFM). In addition, carabids were sampled at another 10 plots each located at field margins in ED1 and ED2 (Liu *et al*. 2006).

In 2014, we sampled a similar array of plots located in the vicinity of the original sites, since some of the original plots had been transformed to settlements or other non-cultivated land-uses. Following the original sampling layout, 30 plots were established in ED3, with one plot each situated within a field margin (FM) and 10 m inside the field (NFM), while the FFM plots were located only 20 m inside the field due to the smaller field sizes encountered in 2014. In addition, another 10 plots each were again established at field margins in ED1 and ED2.

Sites were selected to represent the respective dominant land-use types in the region. In 1997, selected sites included 24 winter wheat/summer maize rotational systems as well as 2 vegetable and 2 cotton fields, while only winter wheat/summer maize was selected in 2014, as the sowing area of vegetables and cotton fields in ED3 was very small in that year.

Carabid Sampling. Carabids were sampled using pitfall traps. Sampling plots consisted of arrays of five pitfall traps placed 5 m apart in a straight line within the field margin habitats at each plot, and in ED3 also in a straight line inside the field parallel to the field margin. All pitfall traps operated for 5 days each month from May to October in both 1997 and 2014. Traps were partly filled with 15% salt water to preserve the specimens, and some detergent was added to break the water surface tension. Overall, 10 plots containing 50 traps were placed in the selected field margins of each experimental district, and another 10 plots with 50 traps were placed 10 m and 20 m/30 m away from the field margins in fields of ED3, respectively, resulting in a total of 50 study plots containing 250 pitfall traps.

Recording of Environmental Variables. In September of both 1997 and 2014, site conditions within the field margins, including the total number of plant species, the soil salt content and the soil nitrogen content, were recorded (Table 1). The soil salt content was measured as conductivity, while soil nitrogen was recorded as alkali-soluble soil N in 1997 and as total soil N in 2014. Both alkali-soluble and total soil N increase with increasing nitrogen fertilizer input (Wang *et al*. 2010) and can hence be used in comparisons to analyses the relative exposition of the fields and field margins to fertilizers.

Data Analysis. Individual-based rarefaction and extrapolation (R/E) curves for carabid were calculated and plotted to compare the species richness between sites. This approach allows a standardized analysis of α-diversity without the discarding of data in large samples (Krebs 1989). Rarefying to a standardized small sample size or extrapolation to a large sample size allows direct comparisons of the estimated species richness for standardized sample sizes (Colwell et al. 2012). We calculated R/E curves using iNEXT (iNterpolation/EXTrapolation) (Colwell et al. 2012; Hsieh et al. 2016), an R package (R Core Team 2015).

Non-linear multidimensional scaling (NMDS) of the chord-normalized expected species shared (CNESS)-index (Trueblood *et al*. 1994) was used to analyze the dissimilarity between communities at different sites using PAST (Paleontological Statistics) to calculate the NMDS plots (Hammer *et al*. 2001). The CNESS index represents a probability-based measure of dissimilarity between samples for a pre-determined sample size. A variation in the respective sample-size parameter *m* allows a shift in emphasis in the analysis from the most dominant species (*m*=1, expressing the probability of two individuals randomly sampled from two different samples/plots to represent the same species) to the overall similarity between samples, considering both common and rare species. In our study, the similarity was calculated for *m*=1 and for the largest common sample size for all plots. The CNESS dissimilarity matrix was calculated using COMPAH (Gallagher 1998). In 1997, data of three randomly selected plots had to be pooled together for this ordination analysis in order to obtain a sufficient number of individuals allowing for meaningful analyses. To maintain consistency in our analytical approach, data were again pooled in sets of three randomly selected plots for the ordination analysis in 2014.

Redundancy analysis was used as constrained ordination technique to explore correlations between environmental parameters of the field margin and the composition of carabid assemblages. This analysis was computed using Canoco5 (ter Braak and Smilauer 2012). Prior to the analysis, the species matrix was modified using the Hellinger transformation to optimize the use of the constrained ordination with community composition data containing many zeros (Legendre and Gallagher 2001). The environmental variables were log-transformed to ensure normality and then standardized using z-transformation. We used stepwise selection to select the appropriated subset of the predictors for the RDA. All environmental variables having a significant influence at a significance level of 95% were selected (Lepš and Šmilauer 2003).

.

RESULTS

Carabid Species Alpha-Diversity and Composition. A total of 540 individuals representing 19 species were captured across all plots in 1997, while 4930 individuals representing 34 species were sampled in 2014 (Appendix A). Only 8 species were found in both years, while 10 species were found only in 1997, and 25 species were solely encountered in 2014 (*Harpalus* sp. 1 in 1997 and *Harpalus* sp. 2 in 2014 were excluded from this comparison, because we could not verify whether they represented the same species). Rarefaction and extrapolation (R/E) curves showed a great increase in the carabid species richness between 1997 and 2014, as the diversity of carabid assemblages from all sampling plots was much greater in 2014 when compared to 1997 (Fig. 1a).

The composition of both dominant species (CNESS, *m*=1) (Fig. 2a) and the entire beetle assemblages (CNESS, *m*=25) (Fig. 2b) differed strongly between the two sampling years, indicating a significant change in the composition of the carabid assemblages between the two sampling events.

Differences Between Carabid Diversity and Composition at Field Margins and Fields in ED3. Differences between assemblages sampled in fields and field-margins increased between the two sampling years at ED3. In 1997, 128 individuals representing 17 species were found at the 10 field margin plots, while 194 individuals also representing 17 species were captured at the 20 plots located within the fields. In 2014, 749 individuals representing 28 species and 1458 individuals representing 22 species were recorded from field margins and inside the fields, respectively (Appendix A). In 2014, 6 species were uniquely encountered at the field margin, while this was only true for one species in 1997 (Appendix A). Rarefaction and extrapolation (R/E) also showed that the diversity of carabid assemblages at field margins were not significantly different to in-field assemblages in 1997 (Fig. 1b), while diversity was significantly larger at the field margins in 2014 (Fig. 1c). Overall, the diversity of field margin assemblages increased greatly from 1997 to 2014 (Fig. 1d), while in-field diversity remained widely stable (Fig. 1e).

The composition of dominant species (CNESS, *m*=1) showed an overall greater differentiation than that between field and field-margin assemblages in both respective sampling years (Fig. 2a). A very similar pattern emerged for the composition of the entire beetle assemblages (CNESS, *m*=25) (Fig. 2b). However, field and field-margin plots were more closely aggregated in 2014 than in 1997 (Fig. 2b), indicating a homogenization of the carabid species composition across the fields and field margins.

Changes of Field-Margin Assemblages and Their Responses to Environmental Variables. In 1997, 346 individuals representing 19 species were found in field margins across all the experimental districts, while 3472 individuals representing 34 species were found in field margins in 2014. Again, rarefaction and extrapolation (R/E) showed a significantly regional increase in carabid diversity at field margins in 2014 when compared to 1997 across all three experimental districts (Fig. 1f).

Additionally, a strong turnover in dominant species (CNESS *m*=1, Fig. 3a) and the entire assemblages (CNESS *m*=17, Fig. 3b) occurred between sampling years, with the differentiation of assemblages according to experimental districts being much weaker than the differentiation between years. Dominant species in the field margins showed greater similarities between experimental districts in 2014 than in 1997, again indicating a homogenization in the distribution of dominated species at these habitats.

In 1997, the RDA indicated that Alkali-soluble N was the only environmental variable that showed a significant, albeit small, correlation with the composition of local carabid assemblages (pseudo-F=1.9, *p*=0.02), explaining 6.91% of the total variation in the species composition (Fig. 4). Six of the eleven species present at field margins in 1997, but missing from 2014 margin samples, including *Chlaenius sericimicans* Chaudoir, 1876 (S7), *Harpalus eous* Tschitscherine, 1901 (S21), *Harpalus aogashimensis* Habu, 1957 (S15), *Harpalus vicarius* Harold, 1878 (S31), *Peronomerus nigrinus* Bates, 1873 (S36) and *Pterostichus sp.* (S40), were all negatively linked to alkali-soluble N, indicating that their diversity was negatively affected by increasing soil N contents (Fig. 4).

In 2014, none of the three analyzed environmental predictor variables included in the RDA - plant diversity, soil salt content and total soil N - showed any significant correlations with the composition of local carabid assemblages.

DISCUSSION

Agricultural intensification is a well-known trigger of general declines in biodiversity (Donald *et al*. 2001; Kleijn *et al*. 2009), including in carabid assemblages. However, some authors have suggested that agricultural intensification does not always lead to losses in biodiversity (Tscharntke *et al*. 2005) and needs to be seen holistically in the context of overall changes in environmental conditions across the respective agricultural landscape. In some cases, the higher productivity in an intensified agricultural landscape could potentially sustain a greater abundance and even a greater diversity of organisms in comparison to a more pristine landscape (Söderström *et al*. 2001; Clough *et al*. 2011). In landscapes naturally affected by high levels of salinity, plant growth, biodiversity and crop production are strongly constrained, and our results show that significant effects are also apparent in the ground beetle assemblages. Even under increasing agricultural intensification, plant diversity and crop productivity appear to increase greatly over time once the salinity-related constraints are removed. These changes appear to provide an enhanced direct supply of food sources for invertebrate herbivores, with cascading effects through the food-chain (Murdoch *et al*. 1972; Siemann 1998), potentially explaining the significant increase in carabid diversity between sampling years in our study. The restriction of the associated increases in carabid diversity chiefly to field margin habitats can then be seen as reflecting the much lower exposure of these habitats to intensive agricultural management, in combination with the highly significant increases in plant diversity at these habitats when compared to the agricultural fields.

However, our study explicitly does not demonstrate that agricultural intensification did not have a negative effect on local carabid diversity and assemblage composition, despite the observed increase in species richness. In-field carabid diversity remained very low despite successful desalinization, with the strong changes in the species composition of in-field communities not leading to significant species richness increases in these habitats. Furthermore, agricultural intensification is commonly reported to be associated with shifts towards an increased dominance of habitat generalists and extremely stress-tolerant carabid species that are often characterized by short life cycles and small body sizes (Burel *et al*. 1998; Liu *et al*. 2012). It can be expected that the observed increase in biodiversity across the investigated landscape is chiefly limited to such generalist and stress-tolerant open-field species with distinct traits that allow them to persist within the wider agricultural landscape (Burel *et al*. 1998; Tscharntke *et al*. 2012a). This is confirmed by the fact that 20 of the 25 carabid species uniquely observed in 2014 were common habitat generalist species encountered in at least two distinctly different habitat types such as cultivated land, semi-natural habitats or forests (Liu *et al*. 2006; Yu *et al*. 2006a; Liu *et al*. 2012; Liu *et al*. 2007; Liu *et al*. 2010; Liu *et al*. 2015; Yu *et al*. 2004; Yu *et al*. 2006b; Yu *et al*. 2010). The lack of any significant correlations between the carabid species composition at the field margins and our recorded environmental variables in 2014 further supports the assumption that the carabid assemblages currently encountered across this agricultural landscape are widely composed of generalist species that do not respond strongly to changes in plant diversity or soil salt and soil nitrogen contents. Meanwhile, our results also provide a strong indication that at least some of the species that had disappeared from our 2014 samples did so in response to the environmental changes related to agricultural intensification, since these species showed a strong sensitivity to high nitrogen fertilizer contents as reflected by the respective ordination plots.

Overall, our results conform with the commonly observed trend towards a biotic homogenization, suggesting that human disturbances favor widespread ecological generalist species at the detriment of specialist species (McKinney and Lockwood 1999). In the context of the wider agricultural landscape, our study highlights that the importance of semi-natural field margins strongly increases with increasing agricultural intensification, with these habitats forming significant diversity hotspots for carabids in the landscape that limit the effects of landscape simplification associated with agriculture intensification. This diversity, even if representing a highly homogenized assemblage that lacks a strong spatial differentiation across the wider agricultural landscape, still contains a wide range of different traits, with beetles of widely ranging size (such as *Tachys* sp. with body size of 2-5 mm as well as *Carabus smaragdinus* Fischer, 1823 that exceeds 30 mm in length) and known food requirements (herbivores, carnivores, and omnivores) recorded at the margins in 2014.

The current work also adds crucial insights into the long-term effects agricultural intensification has on biodiversity. To date, most studies investigating the effects of environmental change on biodiversity mainly use space-for-time substitution approaches, with results obtained from spatial landscape gradients used to infer species' responses on temporal scales (Burel *et al*. 1998; Pickett 1989; Bonthoux *et al*. 2013).

In addition, our study demonstrates that the degree of change in species richness and α-diversity provides a highly incomplete view of the effects of agricultural intensification on species-rich taxa (Dormann *et al*. 2007), as it for example fails to highlight the homogenization of the community composition, and the associated potential negative effects on ecosystem functioning and ecosystem service provisions that have wide-spread implications also for food security (Tscharntke *et al*. 2012b; Bommarco *et al*. 2013). A thorough understanding of the specific species traits and functional roles that were favored by the agricultural intensification and the long-term consequences of the resulting shifts in species traits and functional groups for ecosystem functioning are crucial to effectively conserve both biodiversity and food security (Tscharntke *et al*. 2012a; Tscharntke *et al*. 2012b; Bommarco *et al*. 2013; Clavel *et al*.

2011). While the key role of semi-natural habitats such as field margins as refuge habitats for many generalist invertebrate species in the agricultural landscapes and their resulting importance for ecosystem services provided across the agricultural landscape is undisputable, we need to further our understanding how factors such as the overall landscape structure and connectivity affects this role, and how we can therefore optimize the management of these habitats in the landscape to strongly support agricultural diversity and services (Haaland *et al*. 2011; Tscharntke *et al*. 2012a).

CONCLUSION

Following desalinization and increasingly intensive agricultural management over >20 a, carabid species richness significantly increased in field margin habitats. Substantial temporal changes in their species composition resulted in a homogenized community of generalist species encountered across the agricultural landscape. The effects of this homogenization, associated with the loss of low nutrient specialist species and their specific traits and ecosystem functioning, clearly require further scrutiny. Furthermore, it must be noted that assemblages sampled within cultivated fields remained very species-poor despite the desalinization and the associated strong shift in the species composition. Overall, our results confirm that desalinization and subsequent intensive agricultural management could represent an efficient approach to expand the arable land area and increase overall agricultural yield, with relatively minor negative implications for agricultural biodiversity, as long as semi-natural habitats are promoted and managed favourably within the resulting intensively managed agricultural landscape matrix.

ACKNOWLEDGMENTS

The authors are greatly indebted to the National Natural Science Foundation of China (41271198 and 31470514). We are also very grateful to Dr Hongbin Liang from the Institute of Zoology at the China Academy of Sciences for his great help with the determination of carabid specimens caught in 2014

REFERENCES CITED

Bommarco, R., D. Kleijn, and S. G. Potts. 2013. Ecological intensification: harnessing ecosystem services for food security. Trends in Ecology & Evolution 28(4):230-238.

Bonthoux, S., J. Y. Barnagaud, M. Goulard, and G. Balent. 2013. Contrasting spatial and temporal responses of bird communities to landscape changes. Oecologia 172(2):563-574.

Burel, F., J. Baudry, A. Butet, P. Clergeau, Y. Delettre, D. Le Coeur, F. Dubs, N. Morvan, G. Paillat, and S. Petit. 1998. Comparative biodiversity along a gradient of agricultural landscapes. Acta Oecologica 19(1):47-60.

Clavel, J., R. Julliard, and V. Devictor. 2011. Worldwide decline of specialist species: toward a global functional homogenization? Frontiers in Ecology and the Environment 9(4):222-228.

Clough, Y., J. Barkmann, J. Juhrbandt, M. Kessler, T. C. Wanger, A. Anshary, D. Buchori, D. Cicuzza, K. Darras, and D. D. Putra. 2011. Combining high biodiversity with high yields in tropical agroforests. Proceedings of the National Academy of Sciences 108(20):8311-8316.

Colwell, R. K., A. Chao, N. J. Gotelli, S. Y. Lin, C. X. Mao, R. L. Chazdon, and J. T. Longino. 2012. Models and estimators linking individual-based and sample-based rarefaction, extrapolation and comparison of assemblages. Journal of Plant Ecology 5(1):3-21.

Dagar, J. C. 2003. Biodiversity of Indian saline habitats and management and utilization of high salinity tolerant plants with industrial application for rehabilitation of saline areas. Desertification in the third millennium. Swets and Zeitlinger Publishers, Lisse:151-172.

Den Boer, P. J. 1970. On the significance of dispersal power for populations of carabid-beetles (Coleoptera, Carabidae). Oecologia 4(1):1-28.

Donald, P. F., R. E. Green, and M. F. Heath. 2001. Agricultural intensification and the collapse of Europe's farmland bird populations. Proceedings of the Royal Society of London B: Biological Sciences 268(1462):25-29.

Dormann, C. F., O. Schweiger, I. Augenstein, D. Bailey, R. Billeter, G. De Blust, R. DeFilippi, M. Frenzel, F. Hendrickx, and F. Herzog. 2007. Effects of landscape structure and land-use intensity on similarity of plant and animal communities. Global

Ecology and Biogeography 16(6):774-787.

Ernoult, A., Y. Tremauville, D. Cellier, P. Margerie, E. Langlois, and D. Alard. 2006. Potential landscape drivers of biodiversity components in a flood plain: Past or present patterns? Biological Conservation 127(1):1-17.

FAO. 2015. Food and Agriculture Organization of the United Nations Statistics Division. Available at: http://faostat3.fao.org/home/E (accessed.12th May 2015).

Fischer, M., O. Bossdorf, S. Gockel, F. Hänsel, A. Hemp, D. Hessenmöller, G. Korte, J. Nieschulze, S. Pfeiffer, and D. Prati. 2010. Implementing large-scale and long-term functional biodiversity research: The Biodiversity Exploratories. Basic and Applied Ecology 11(6):473-485.

Fukami, T., and D. A. Wardle. 2005. Long-term ecological dynamics: reciprocal insights from natural and anthropogenic gradients. Proceedings of the Royal Society of London B: Biological Sciences 272(1577):2105-2115.

Gallagher, E. 1998. Compah96. URL (24 September 2004): www.es.umb.edu/faculty/edg/files/pub/COMPAH.EXE.

Haaland, C., R. E. Naisbit, and L. F. BERSIER. 2011. Sown wildflower strips for insect conservation: a review. Insect Conservation and Diversity 4(1):60-80.

Hammer, Ø., D. A. T. Harper, and P. D. Ryan. 2001. PAST: Paleontological Statistics Software Package for education and data analysis. Palaeontolia Electronica 4.

Hsieh, T. C., K. H. Ma, and A. Chao. 2016. iNEXT: An R package for interpolation and extrapolation of species diversity (Hill numbers). Methods in Ecology and Evolution. http://chao.stat.nthu.edu.tw/wordpress/software_download/(Accessed, March, 13, 2018)

Kleijn, D., F. Kohler, A. Báldi, P. Batáry, E. D. Concepción, Y. Clough, M. Diaz, D. Gabriel, A. Holzschuh, and E. Knop. 2009. On the relationship between farmland biodiversity and land-use intensity in Europe. Proceedings of the Royal Society of London B: Biological Sciences 276(1658):903-909.

Krebs, C. J. 1989. Ecological methodology. Harper & Row, New York.

La Sorte, F. A., T. M. Lee, H. Wilman, and W. Jetz. 2009. Disparities between observed and predicted impacts of climate change on winter bird assemblages. Proceedings of the Royal Society of London B: Biological Sciences:rspb20090162.

Legendre, P., and E. D. Gallagher. 2001. Ecologically meaningful transformations for ordination of species data. Oecologia 129(2):271-280.

Lepš, J., and P. Šmilauer. 2003. Multivariate analysis of ecological data using CANOCO. Cambridge university press, Cambridge.

Li, J. G., L. J. Pu, M. F. Han, M. Zhu, R. S. Zhang, and Y. Z. Xiang. 2014. Soil salinization research in China: Advances and prospects. Journal of Geographical Sciences 24(5):943-960.

Liu, Y. H., J. C. Axmacher, C. L. Wang, L. T. Li, and Z. R. Yu. 2012. Ground Beetle (Coleoptera: Carabidae) Assemblages of Restored Semi-natural Habitats and Intensively Cultivated Fields in Northern China. Restoration Ecology 20(2):234-239.

Liu, Y. H., J. C. Axmacher, C. L. Wang, L. T. Li, and Z. R. Yu. 2010. Ground beetles (Coleoptera: Carabidae) in the intensively cultivated agricultural landscape of Northern China–implications for biodiversity conservation. Insect Conservation and Diversity 3(1):34-43.

Liu, Y. H., J. C. Axmacher, L. T. Li, C. L. Wang, and Z. R. Yu. 2007. Ground beetle (Coleoptera: Carabidae) inventories: a comparison of light and pitfall trapping. Bulletin of Entomological Research 97(6):577-584.

Liu, Y. H., M. C. Duan, X. Z. Zhang, X. Zhang, Z. R. Yu, and J. C. Axmacher. 2015. Effects of plant diversity, habitat and agricultural landscape structure on the functional diversity of carabid assemblages in the North China Plain. Insect Conservation and Diversity 8(2):163-176.

Liu, Y. H., Z. R. Yu, W. B. Gu, and J. C. Axmacher. 2006. Diversity of carabids (Coleoptera, Carabidae) in the desalinized agricultural landscape of Quzhou county, China. Agriculture, Ecosystems & Environment 113(1):45-50.

Lövei, G. L., and K. D. Sunderland. 1996. Ecology and behavior of ground beetles (Coleoptera: Carabidae). Annual Review of Entomology 41(1):231-256.

Magurran, A. E., S. R. Baillie, S. T. Buckland, J. M. Dick, D. A. Elston, E. M. Scott, R. I. Smith, P. J. Somerfield, and A. D. Watt. 2010. Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. Trends in Ecology & Evolution 25(10):574-582.

Marshall, E. J. P., and A. C. Moonen. 2002. Field margins in northern Europe: their functions and interactions with agriculture. Agriculture, Ecosystems & Environment 89(1):5-21.

McKinney, M. L., and J. L. Lockwood. 1999. Biotic homogenization: a few winners replacing many losers in the next mass extinction. Trends in Ecology & Evolution 14(11):450-453.

MEA. 2005. Millenium ecosystem assessment. Island Press, Washington.

Metzger, J. P., A. C. Martensen, M. Dixo, L. C. Bernacci, M. C. Ribeiro, A. M. G. Teixeira, and R. Pardini. 2009. Time-lag in biological responses to landscape changes in a highly dynamic Atlantic forest region. Biological Conservation 142(6):1166-1177.

Murdoch, W. W., F. C. Evans, and C. H. Peterson. 1972. Diversity and pattern in plants and insects. Ecology 53(5):819-829.

NBSPRC. 2015. National Bureau of Statistics of the People’s Republic of China. Available at: http://data.stats.gov.cn. (Accessed: 12th May 2015).

Norris, K. 2008. Agriculture and biodiversity conservation: opportunity knocks. Conservation Letters 1(1):2-11.

Pickett, S. T. A. 1989. Space-for-time substitution as an alternative to long-term studies. [pp. 110-135]. Long-term studies in ecology. Springer, New York. 110-135 pp.

Siemann, E. 1998. Experimental tests of effects of plant productivity and diversity on grassland arthropod diversity. Ecology 79(6):2057-2070.

Söderström, B. O., B. Svensson, K. Vessby, and A. Glimskär. 2001. Plants, insects and birds in semi-natural pastures in relation to local habitat and landscape factors. Biodiversity & Conservation 10(11):1839-1863.

Team, R. C. 2014. R: A language and environment for statistical computing.

ter Braak, C. J. F., and P. Smilauer. 2012. Canoco reference manual and user's guide: software for ordination, version 5.0. Microcomputer Power.

Thiele, H. U. 2012. Carabid beetles in their environments: a study on habitat selection by adaptations in physiology and behaviour. Springer Science & Business Media.

Trueblood, D. D., E. D. Gallagher, and D. M. Gould. 1994. Three stages of seasonal succession on the Savin Hill Cove mudflat, Boston Harbor. Limnology and Oceanography 39(6):1440-1454.

Tscharntke, T., A. M. Klein, A. Kruess, I. Steffan-Dewenter, and C. Thies. 2005. Landscape perspectives on agricultural intensification and biodiversity-ecosystem service management. Ecology Letters 8(8):857-874.

Tscharntke, T., J. M. Tylianakis, T. A. Rand, R. K. Didham, L. Fahrig, P. Batary, J. Bengtsson, Y. Clough, T. O. Crist, and C. F. Dormann. 2012a. Landscape moderation of biodiversity patterns and processes-eight hypotheses. Biological Reviews 87(3):661-685.

Tscharntke, T., Y. Clough, T. C. Wanger, L. Jackson, I. Motzke, I. Perfecto, J. Vandermeer, and A. Whitbread. 2012b. Global food security, biodiversity conservation and the future of agricultural intensification. Biological Conservation 151(1):53-59.

Wang, J. L., X. J. Huang, T. Y. Zhong, and Z. G. Chen. 2011. Review on sustainable utilization of salt-affected land. Acta Geographica Sinica 66(5):673-684.

Wang, J., J. L. Lv, M. G. Xu, Y. H. Duan, B. R. Wang, and J. Huang. 2010. The variation characteristics of nitrogen in red soil under long-term different fertilization. Soil and Fertilizer Sciences in China:1-6.

Wu, Y. P., Y. Zhang, Y. M. Bi, and Z. J. Sun. 2015. Biodiversity in saline and non-saline soils along the Bohai sea Coast, China. Pedosphere 25(2):307-315.

Xin, D. H., and W. J. Li. 1990. Integrated reclamation and development of a low production salinity area. Beijing Agricultural University Press, Beijing.

Yu, X. D., T. H. Luo, and H. Z. Zhou. 2004. Carabus (Coleoptera: Carabidae) assemblages of native forests and non-native plantations in Northern China. Entomologica Fennica 15(3):129-137.

Yu, X. D., T. H. Luo, and H. Z. Zhou. 2006a. Habitat associations and seasonal activity of carabid beetles (Coleoptera: Carabidae) in Dongling Mountain, North China. Entomologica Fennica 17(2):174.

Yu, X. D., T. H. Luo, and H. Z. Zhou. 2010. Distribution of ground-dwelling beetle assemblages (Coleoptera) across ecotones between natural oak forests and mature pine plantations in North China. Journal of Insect Conservation 14(6):617-626.

Yu, Z. R., Y. H. Liu, and J. C. Axmacher. 2006b. Field margins as rapidly evolving local diversity hotspots for ground beetles (Coleoptera: Carabidae) in northern China. The Coleopterists Bulletin 60(2):135-143.

Ladã¡Nyi, Z., Blanka, V., Ãron JÃ³zsef DeÃ¡k, Rakonczai, J., & MezåSi, G. 2016. Assessment of soil and vegetation changes due to hydrologically driven desalinization process in an alkaline wetland, hungary. Ecological Complexity 25, 1-10.

Jung, M. P., Kim, S. T., Kim, H., & Lee, J. H. 2008. Biodiversity and community structure of ground-dwelling spiders in four different field margin types of agricultural landscapes in korea. Applied Soil Ecology, 38(2), 185-195.

Cole, L. J., Brocklehurst, S., Mccracken, D. I., Harrison, W., & Robertson, D. 2012. Riparian field margins: their potential to enhance biodiversity in intensively managed grasslands. Insect Conservation & Diversity, 5(1), 86–94.

Kareiva, P. 1983. Influence of vegetation texture on herbivore populations: resource concentration and herbivore movement.Variable Plants and Herbivores in Natural and Managed Systems (ed. by R.F. Denno and M.S. McClure), pp. 259–289.Academic Press, New York City, New York.

Siemann, E. 1998. Experimental tests of effects of plant productivity and diversity on grassland arthropod diversity. Ecology,79, 2057–2070.

R Core Team. 2015. R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. URL http://www.R-project.org/.

Appendix A Distribution of carabid species at field margins (field margin of 3rd ED) and at field in Quzhou in 1997 and 2014 and the records of their habitats and distribution in the Northern of China

Code	**Species**	**1997**			**2014**			**Habitats**	**Distribution records in North of China**
		FM	**CF**	**Total**	**FM**	**CF**	**Total**		
S1	*Amara communis* Panzer, 1797 **				3		3	field margin, grassland, plantation forest, forest, vegetable field	Chongli, Yanqing, Quzhou, Fangshan
S2	*Asaphidion semilucidum* Motschulsky, 1862 **				41(23)	16	57	field margin, cereal field, plantation forest	Quzhou, Shunyi, Anyang
S3	*Anisodactylus signatus* Panzer, 1797 **				52		52	field margin, cereal field	Shunyi, Quzhou, Dongbeiwang
S4	*Carabus brandti* Faldermann, 1835 **				4(4)		4	field margin, grassland, plantation forest, forest, cereal field, bushwood, windbreak	Haidian, Yanqing, Chongli, Miyun, Quzhou, Donglingshan, Shunyi, Fangshan, Anyang
S5	*Carabus smaragdinus* Fischer, 1823 **				3(2)	2	5	orchard, field margin, grassland, plantation forest, forest, cereal field, bushwood, windbreak	Haidian, Yanqing, Chongli, Donglingshan, Fangshan, Quzhou, Miyun, Anyang
S6	*Chlaenius micans* Fabricius, 1792	4(3)		4	1772(351)	785	2557	cereal yield, orchard, field margin, plantation forest, vegetable field, grassland	Haidian, Yanqing, Quzhou, Shunyi, Miyun, Dongbeiwang, Anyang
S7	*Chlaenius sericimicans* Chaudoir, 1876 *	5	3	8				field margin, cereal field	Quzhou

S8	*Chlaenius virgulifer* Chaudoir, 1876	2(1)	1	3	3(3)	5	8	field margin, cereal field, plantation forest, forest	Miyun, Shunyi, Quzhou
S9	*Curtonotus giganteus* Mostchulsky 1844**				11(3)		11	field margin, grassland, plantation forest, forest, cereal field, windbreak	Shunyi, Dongbeiwang, Miyun, Quzhou
S10	*Diplocheila* sp.**				1(1)		1	field margin	Quzhou
S11	*Dolichus halensis* Schaller, 1783	18(10)	11	29	121(13)	20	141	field margin, cereal field, vegetable field, orchard, plantation forest, grassland, windbreak, forest	Haidian, Yanqing, Chongli, Fangshan, Shunyi, Miyun, Quzhou
S12	*Dyschirius sp.* **				65(16)	72	137	field margin, cereal field	Quzhou
S13	*Diplocheila zeelandica* L. Redtenbacher, 1867 **				3		3	field margin	Quzhou
S14	*Harpalus amputatus* Say, 1830 **				2(1)		2	field margin, cereal field	Shunyi, Quzhou
S15	*Harpalus aogashimensis* Habu, 1957 *	5(2)	2	7				field margin, cereal field	Quzhou
S16	*Harpalus bungii* Chaudoir, 1844 **				165(47)	22	187	field margin, grassland, plantation forest, forest, vegetable field, orchard, bushwood, windbreak	Haidian, Yanqing, Chongli, Shunyi, Donglingshan, Miyun, Quzhou, Dongbeiwang, Anyang
S17	*Harpalus chasanensis* Lafer, 1989 **				1(1)		1	field margin, plantation forest, windbreak	Shunyi, Quzhou, Miyun
S18	*Harpalus calceatus* Duftschmid, 1812 **				2		2	cereal field, vegetable field, grassland, plantation forest, field margin, orchard	Yanqing, Chongli, Shunyi, Fangshan, Quzhou
S19	*Harpalus corporosus* Motschulsky, 1861 **				11(8)	1	12	field margin, grassland, plantation forest, forest, vegetable field, orchard, bushwood, windbreak	Yanqing, Chongli, Shunyi, Fangshan, Quzhou, Miyun, Anyang

S20	*Harpalus davidi* Tschitscherine, 1897 **				3		3	field margin, grassland, cereal field, orchard	Chongli, Quzhou
S21	*Harpalus eous* Tschitscherine, 1901 *	17(2)	3	20				field margin, cereal field, plantation forest	Chongli, Miyun, Quzhou
S22	*Harpalus griseus* Panzer, 1796	75(25)	47	122	223(18)	96	319	field margin, grassland, plantation forest, vegetable field, orchard	Haidian, Yanqing, Chongli, Fangshan, Shunyi, Miyun, Quzhou, Dongbeiwang
S23	*Harpalus pallidipennis* A. Morawitz, 1862 **				206(131)	110	316	field margin, grassland, plantation forest, cereal field, vegetable field, orchard, windbreak	Haidian, Chongli, Dongbeiwang, Miyun, Anyang, Shunyi, Fangshan, Quzhou
S24	*Harpalus pastor* Motschulsky, 1844 **				70(11)	15	85	field margin, grassland, plantation forest, cereal field, vegetable field, orchard, bushwood, forest	Yanqing, Chongli, Shunyi, Fangshan, Quzhou, Miyun, Dongbeiwang, Donglingshan
S25	*Harpalus roninus* Bates, 1873 **				13(4)	6	19	cereal field, field margin	Chongli, Yanqing, Haidian, Shunyi, Quzhou
S26	*Harpalus simplicidens* Schauberger, 1929	63(22)	61	124	117(2)	24	141	field margin, plantation forest, cereal field, vegetable field, orchard	Haidian, Yanqing, Chongli, Shunyi, Miyun, Quzhou, Fangshan, Dongbeiwang
S27	*Harpalus sinicus* Hope, 1845	11(6)	3	14	2(1)	3	5	field margin, orchard, plantation forest, vegetable forest	Haidian, Yanqing, Chongli, Quzhou, Dongbeiwang
S28	*Harpalus* sp. 1	1	6	7				field margin, cereal field	Quzhou
S29	*Harpalus* sp. 2				5(2)	2	7	field margin, cereal field	Quzhou
S30	*Harpalus tridens* A.Morawitz, 1862 *	42(19)	27	69				field margin, cereal field	Dongbeiwang, Quzhou
S31	*Harpalus vicarius* Harold, 1878 *	4(2)		4				field margin	Quzhou

S32	*Lesticus magnus* Motschulsky,1860 *	16(4)	3	19				field margin, cereal field, vegetable field, plantation forest	Yanqing, Shunyi, Dongbeiwang, Quzhou
S33	*Microlestes* sp.**				103(24)	78	181	field margin, cereal field	Quzhou
S34	*Patrobus flavipes* Motschulsky, 1844*	6(3)	2	8				field margin, cereal field	Quzhou
S35	*Panagaeus davidi* Fairmaire, 1887 **				3		3	cereal field, field margin, forest	Shunyi, Quzhou, Donglingshan
S36	*Peronomerus nigrinus* Bates, 1873 *	17(9)	4	21				cereal field, field margin	Quzhou
S37	*Poecilus nitidicollis* Motschulsky, 1844 **				126(8)	46	172	cereal field, field margin	Quzhou
S38	*Pterostichus haptoderoides* Tschitscherine, 1889 **				75(6)	9	84	cereal field, field margin, grassland	Chongli, Quzhou
S39	*Pterostichus microcephalus* Motschulsky, 1860	20(2)	4	24	118(19)	26	144	cereal field, field margin, grassland, orchard, plantation forest, bushwood	Haidian, Yanqing, Chongli, Miyun, Quzhou, Donglingshan, Shunyi, Anyang, Dongbeiwanng
S40	*Pterostichus* sp. *	14(5)	3	17				cereal field, field margin	Quzhou
S41	*Pterostichus sulcitarsis* A. Morawitz, 1862 **				6(2)	6	12	cereal field, field margin	Quzhou
S42	*Scarites acutidens* Chaudoir, 1855	5(2)	1	6	2(2)		2	cereal field, field margin, vegetable field, bushwood, forest, plantation forest, grassland	Chongli, Donglingshan, Quzhou
S43	*Scarites rectifrons* Bates 1873*	21(11)	13	34				cereal field, field margin	Quzhou

S44	*Scarites terricola* Bonelli, 1813 **				40(2)	18	58	field margin, grassland, plantation forest, cereal field, vegetable field, orchard, windbreak	Haidian, Yanqing, Shunyi, Miyun, Quzhou, Dongbeiwang, Anyang
S45	*Tachys* sp.**				100(44)	96	196	cereal field, field margin	Quzhou
Total abundance		346(128)	194	540	3472(749)	1458	4930		

*Species present in 1997 only; **Species present in 2014 only.

Hebei province: Quzhou (36° 36′ –36° 58′ ″ N, 114° 50′ –115° 13′ E); Chongli (40° 47′ –41° 17′ N, 114° 47′ –115° 34′ E).

Beijing: Dongbeiwang (40°28′ '–41°05′ N, 115825′ –117°30′ E); Shunyi (40° 14′ 21.66″ –40° 13′ 55.35″ N, 116° 36′ 0.49″ –116° 34′ E); Miyun (40° 21′ –40° 31′ N, 116° 41′ –116° 49′ E); Fangshan (39° 43′ –39° 49′ N, 115° 35′ –115° 46′ E); Yanqing (40° 16'-40° 47'N, 115°44'-116°34'E); Haidian (39° 53′ -40° 09′ N, 116° 03′ -116° 23′ E); Donglingshan (40° 18′ N, 115° 44′ E). Henan province：Anyang (36° 12′ ﹣36° 7′ N, 114° 4′ ﹣114° 14′ E).

Habitat and distribution information were derived from Liu *et al*, 2006, 2007, 2010,2012,2015; Yu *et al*, 2004,2006a,2010; Yu *et al*, 2006b; Warren-Thomas *et al*. ,2014, and also our unduplicated data.

Data in bracket indicated the numberof individuals presented in the third Experimental District

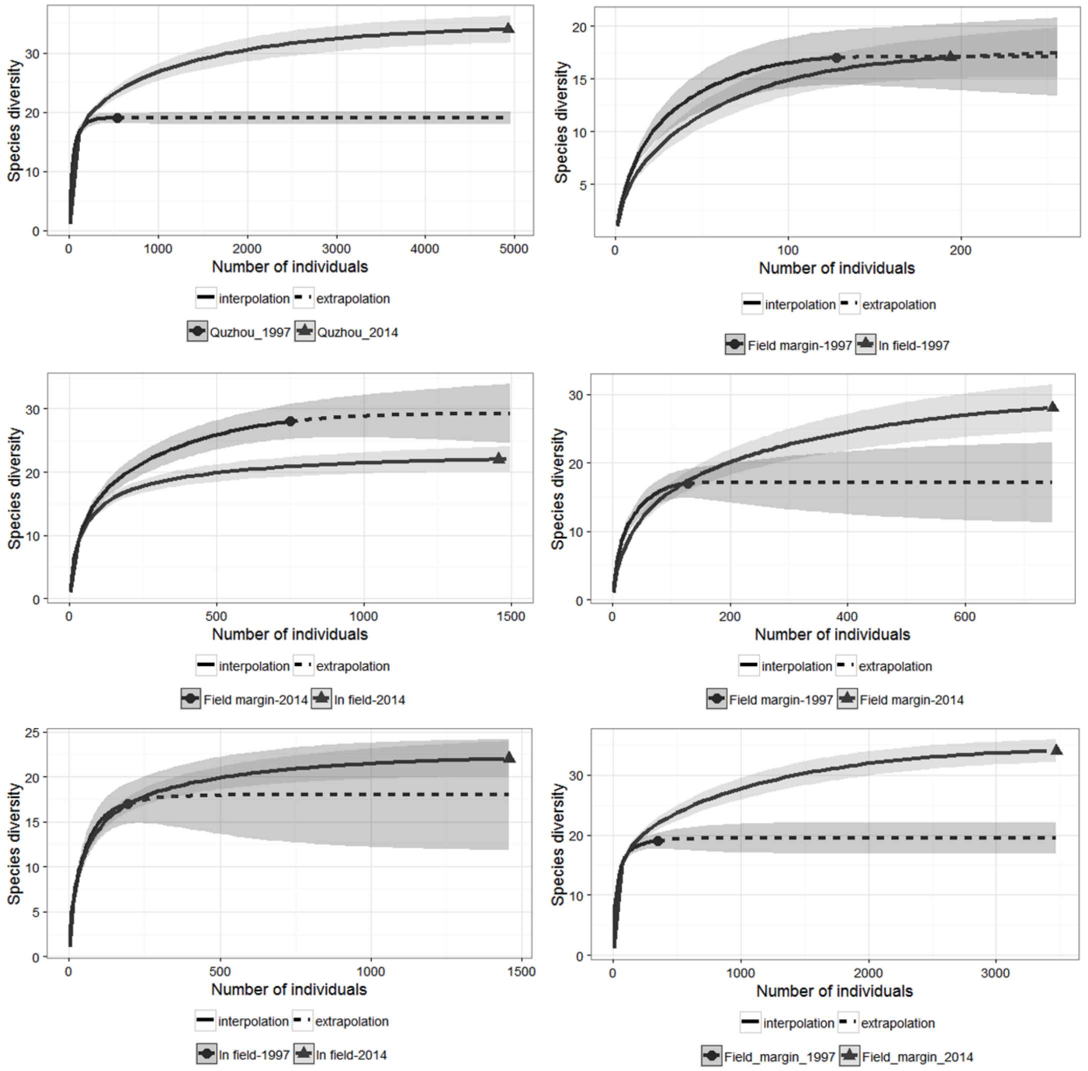

Fig. 1. Rarefaction and extrapolation (R/E) curves of carabid assemblages at agriculture landscape. a) Comparing carabid samples from all plots in Quzhou in 1997 and 2014, b) Comparing carabid samples from field margin plots and fields plots at the 3rd Experimental district in 1997, c) carabid samples from all field margin plots and fields plots at the 3rd Experimental district in 2014, d) Comparing carabid samples from all field margin plots at the 3rd Experimental district between 1997 and 2014, e) Comparing carabid samples from all fields at 3rd Experimental district between 1997 and 2014, f) Comparing carabid samples from all field margins across all the three Experimental districts between 1997 and 2014.

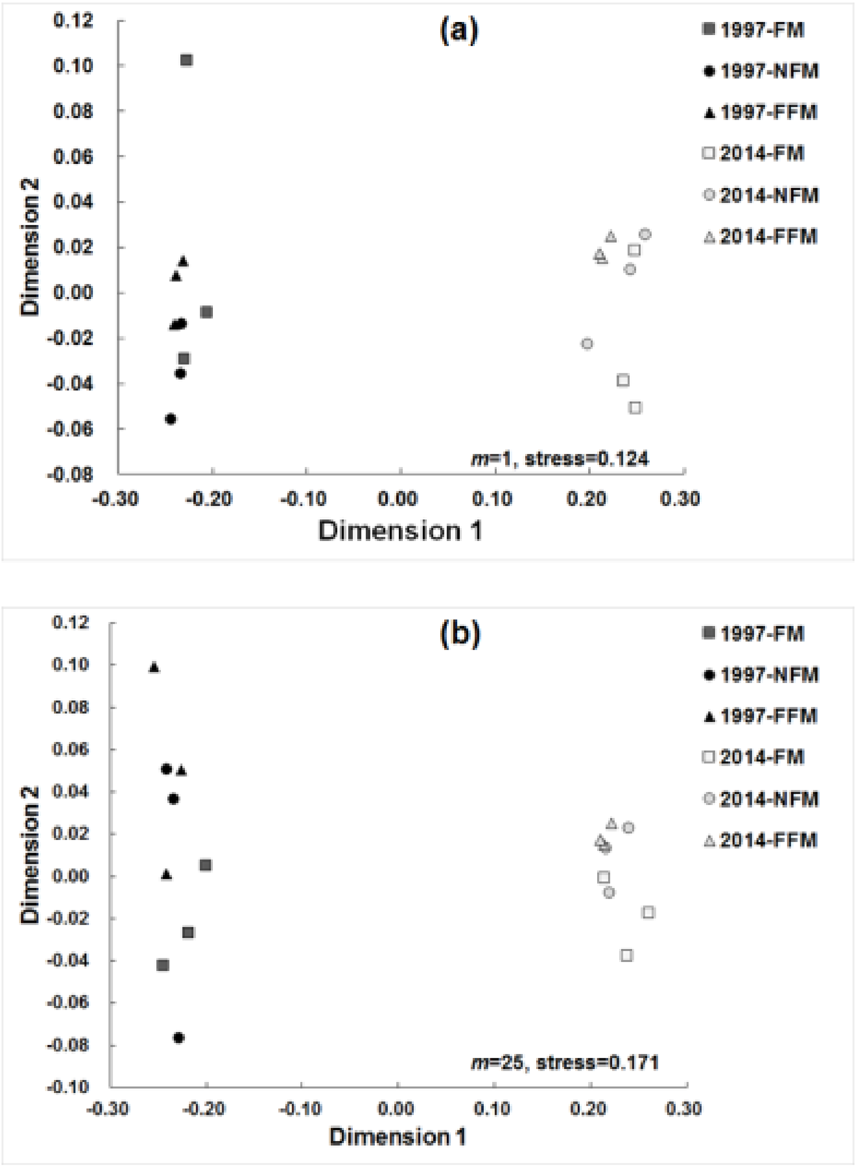

Fig. 2. Non-linear two-dimensional scaling of pooled carabid samples in the 3rd experimental district in 1997 and 2014 based on the chord-normalized expected species shared (CNESS)-index of dissimilarity. a) m=1, b) m=25 (FM: field margin; NFM: sites 10 m inside the field, FFM: sites 30 m (in 1997) / 20 m (in 2014) inside the field (3 samples of equal treatments randomly pooled).

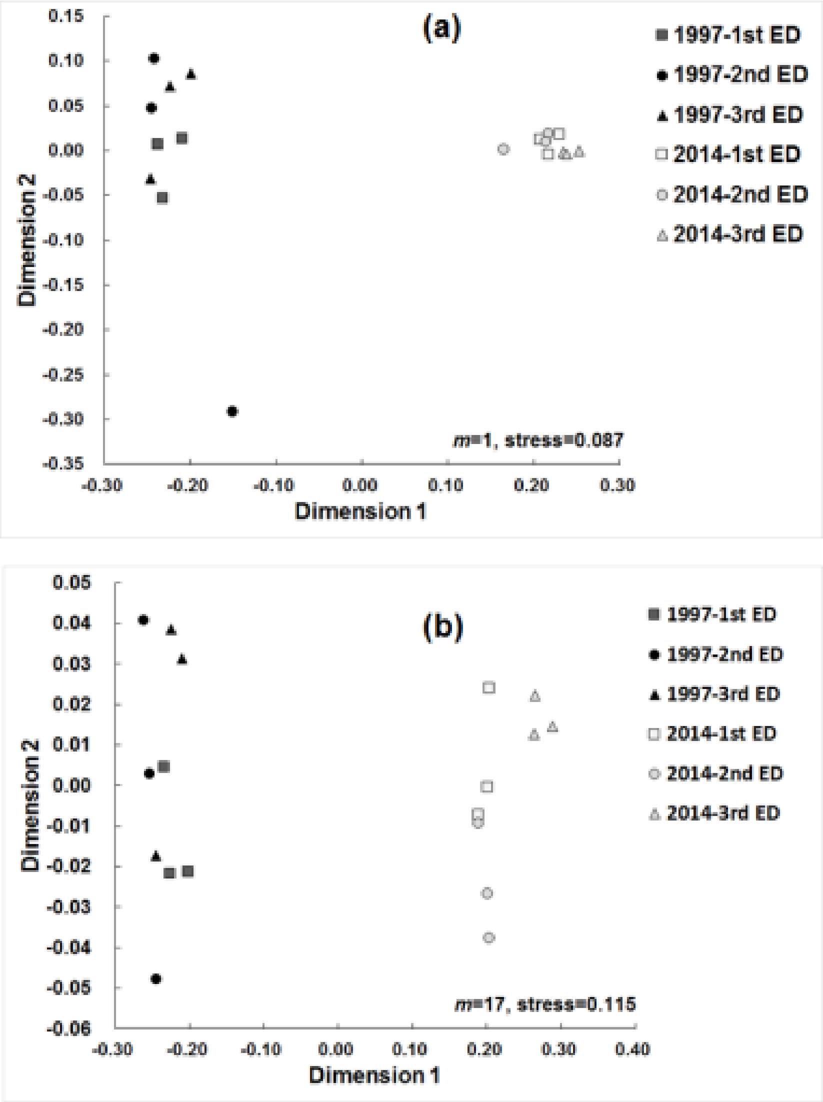

Fig. 3. Non-linear two-dimensional scaling of pooled carabid samples at field margins of different experimental districts in 1997 and 2014 based on the chord-normalized expected species shared (CNESS)-index of dissimilarity. a) m=1, b) m=17 (3 samples at the same experimental district randomly pooled).

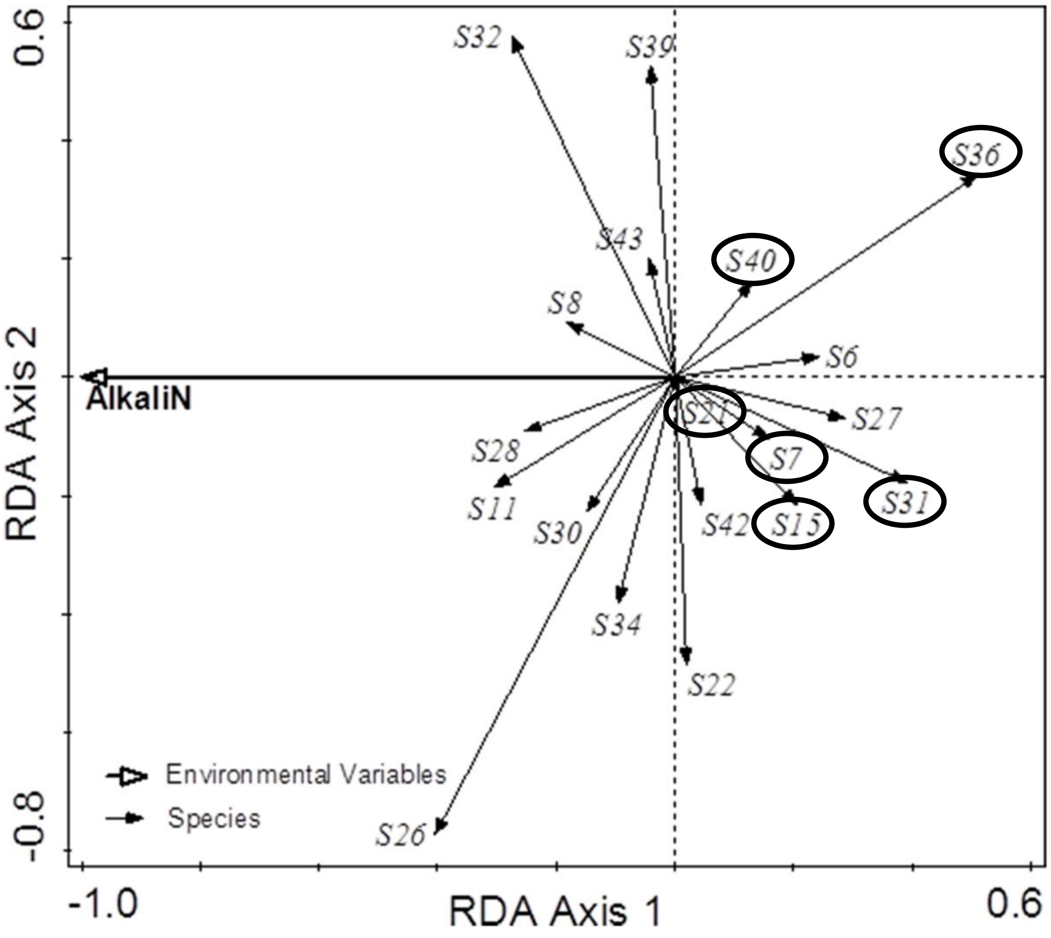

Fig. 4. Canonical correspondence analysis of carabid beetles and environmental parameters at field margins of different experimental districts in 1997 (Species with black were not present in 2014. For species names refer to Appendix A).

Table 1 Environmental parameters at field margins in different desalination districts (Mean ± S.D.)

Year	Desalination district	Plant species richness	Soil salt content (ms/cm)	Soil nitrogen content*
1997	1st experimental district	5.6±1.6	2.75±1.92	57.85±13.40
	2nd experimental district	6.5±1.2	3.82±1.27	50.82±16.79
	3rd experimental district	7.0±1.2	1.54±0.56	46.92±14.06
2014	1st experimental district	16.5±1.8	0.23±0.07	0.09±0.01
	2nd experimental district	20.3±6.1	0.36±0.13	0.09±0.02
	3rd experimental district	21.9±3.8	0.54±0.39	0.11±0.04

*measured as alkali-soluble soil N (ppm) in 1997 and as soil total nitrogen (g/kg) in 2014.