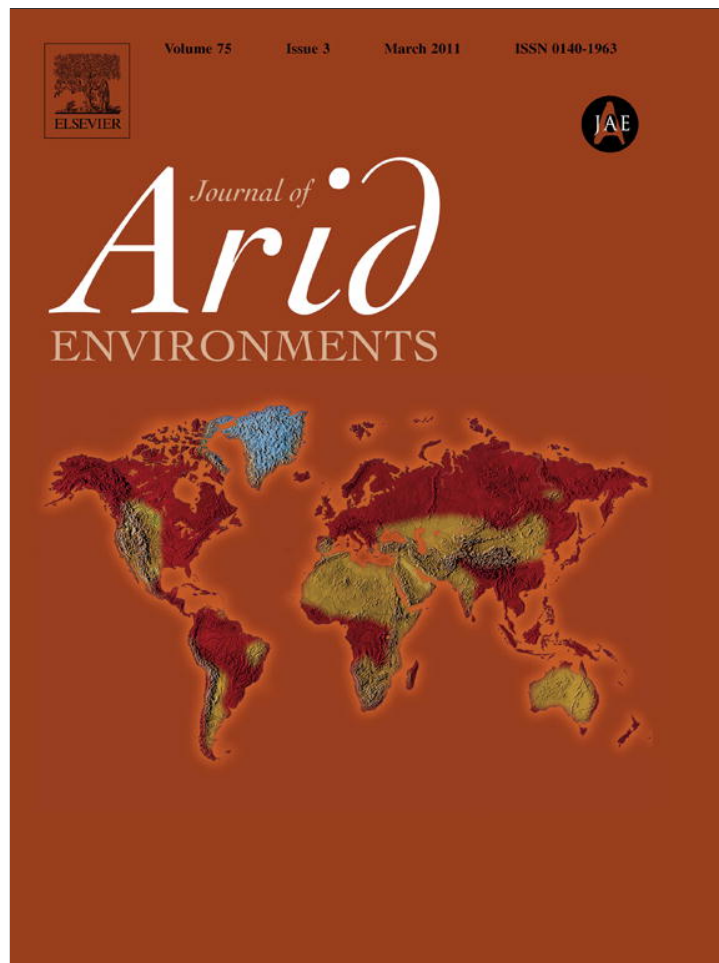


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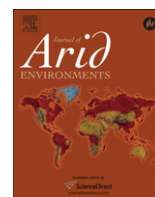
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Landscape and a political border determine desert arthropods distribution

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ABSTRACT

We studied the distribution of arthropods in the Arava desert on both sides of the Israeli-Jordanian border, to assess the impact of different anthropogenic pressures on the local fauna. We examined how different landscape units, proximity to agricultural fields, and human societies, might affect the diversity of ground dwelling beetles, and spiders, using ordination and diversity estimation methods. Our results suggest that although both countries contain similar habitats, each has its own unique characteristics, probably due to different cultural practices. The immediate repercussion is that loss of a habitat on one side of the border cannot be compensated with preservation of the same habitat across the border, due to fauna dissimilarity. For example, beetle species can be assembled according to landscape units, but within each landscape unit they show dissimilarities that are based on the geopolitical location. Spiders fail to assemble according to landscape units but cluster as a unique group within Israel. Both landscape unit type and the border, were found to be important for the overall species diversity of this ecosystem and therefore “redundancy” should be carefully applied, especially across geopolitical borders.

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1. Introduction

The reduction in biodiversity due to habitat loss and ecosystem degradation has made it vital to prioritize land usage for conservation (Wilson et al., 2006). Land-use prioritization has been suggested on a range of geographic scales: global [in which primarily worldwide hot spots of diversity or endemism are identified for preservation (Orme et al., 2005; Brooks et al., 2006; Lamoreux et al., 2006; Araujo and Rahbek, 2007)], continental (Williams et al., 2000; Moore et al., 2003; Larsen and Rahbek, 2005; Tognelli, 2005), regional (Gering et al., 2003), and national or state (Kati et al., 2004; Warman et al., 2004; Bonn and Gaston, 2005; Bani et al., 2006). However, to date, most of the land-use prioritization decisions are made on a national level rather than across international boundaries. Consequently, and unfortunately, the world's land is managed by multiple, independent bodies (i.e., countries),

each having different and often conflicting agendas concerning their own land use. The potential shortcomings of this situation are readily seen when countries share a single, continuous, biome, yet treat their natural resources in different ways (e.g., the Serengeti-Mara region shared by Kenya and Tanzania, Homewood et al., 2001). Therefore better collaboration and coordination are often sought between countries sharing a mutual priority region (Medail and Quezel, 1999).

In this study, we aimed to examine how a political border affects the diversity of two well established indicative taxonomic groups, beetles and spiders (Bromham et al., 2002; Pearce and Venier, 2006), and what consequences this has on the mutual management of biodiversity in an important desert system, the Arava valley. We chose to concentrate on epigeal arthropod communities by constructing pitfall traps, an unbiased method to relate species diversity to a specific sampling plot. In addition to the border effect, we examined the effects of proximity to agricultural fields and how different landscape units affect diversity levels. We also wished to compare the suitability of various surrogate species, from different taxonomic groups and at different levels of analysis, for prioritization decision-making.

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The Arava valley is a desert ecosystem, part of the great rift valley, shared by Israel and Jordan. It is one of the major migratory routes for old world birds, as approximately half a billion birds of more than 260 species pass through this valley twice a year on their spring and autumn migrations between Africa and Eurasia (Yosef, 1996). It is also an important biogeographical corridor, located at the junction of three continents. Nevertheless, the ecology of this region is under increasing threat that has escalated as a result of the peace treaty signed by Israel and Jordan in 1994. Immediate concerns include the proposed Red Sea – Dead Sea water canal (World Bank publications, 2007), the construction of a new international airport, and increased agricultural use of land. Therefore, land-use prioritization with special emphasis on biodiversity conservation and management should be sought for this region. For example, the main stakeholders on the Israeli side of this region, namely, the local regional council, the Israel Land Administration, and the Israel Nature and Park Authority, are in constant dispute on land use, often unsupported by empirical data.

The valley has been disproportionately developed on the Israeli side of the border, with vast areas settled and transformed into irrigated agricultural fields by ten collective communities (Kibbutzim: Efrat, 1993). The land on the Jordanian side has remained relatively intact and only sparsely populated mainly by traditional and pastoral societies, with few villages that started in recent years to develop irrigated farming (Khoury and Al-Shamli, 2006). This has resulted in different faunal representation across the political border (Shanas et al., 2006). Hence, in this study, we aimed to examine how the different patterns of land use affect biodiversity patterns across this political border. To date, this study is the most comprehensive biodiversity study along the Israel–Jordan border.

2. Methods

2.1. Study site

The southern Arava Valley, a part of the Great Rift Valley, located between Nahal Shita in Israel and Wadi Arandal in Jordan (030°07'10" N) in the north; the Red Sea (029°32'57" N) in the south; the mountains of the Israeli Negev in the west; and the Sharrah Mountains of Jordan in the east. Temperatures in this region vary from 23 to 45 °C during the summer and 0 to 23 °C during the winter. Average annual precipitation is 28.7 mm (1971–2000 mean, Israel Central Bureau of Statistics). During the two-year period of our study, precipitation levels were 12.0 mm and 22.5 mm for 2002–2003 and 2003–2004, respectively. The valley is comprised of different habitats, of which the major ones are alluvial fans, sand dunes, semi-stable sands, salt marshes, and wadi beds. Although the border politically divides Israel and Jordan, the physical division is only a loose wire mesh fence along the border.

2.2. Landscape units, proximity to agriculture and “border effect”

Based on literature and preliminary data, we selected four habitat types to represent landscape units in which we expected to find most of the region's biomass: **HD** – High Density, alluvial fans with a relatively high density of acacia trees (*Acacia tortilis* and *Acacia. raddiana*) and shrubs (10–20 acacia trees per hectare, *Sal-sola tetrandra* and *Lycium shawii* as major shrubs); **SM** – Salt Marsh, salt marsh edges typified by silty soil, where the most common shrub was *Nitraria retusa* (18–160 individuals per hectare), in some places joined by *Alhagi graecorum* and *Zygophyllum* spp. shrubs; **SD** – Sand Dunes, typified by shifting sands with approximately 30 *Haloxylon persicum* shrubs per hectare; and **MX** – Mix, semi-stable sands occasionally mixed with gravel, with approximately 25

Haloxylon persicum shrubs per hectare and sporadic occurrence of *Calligonum comosum* shrubs (see also Shanas et al., 2006). Within each landscape unit, we established three “close” plots close to (50–200 m) and three plots “far” (>2 km) from the agricultural fields (hereafter, close and far, respectively). Due to the imbalance in agricultural activity on the two sides of the border, we could not find enough “close” and “far” sites for all habitat types. Therefore, most of the “close” sites were on the Israeli side and most of the “far” sites were on the Jordanian side of the border. Because we also suspected that the border itself, dividing two societies with distinct and different impacts on the land, would affect biodiversity, we chose three additional plots in each country to enable an exclusive cross-comparison of the effect of agricultural fields as well as an exclusive cross-comparison of the “border effect” (for complete details of the study sites see Appendix A and Shanas et al., 2006). In this way, we were able to compare the four landscape units within each country; then use the SM plots close and far from agricultural fields within Israel, and the SD plots close and far from agricultural fields within Jordan to perform separate comparisons of the effect of agriculture proximity. The SM far plots and the SD close plots were used to compare the “border effect”, as these landscape unit types were monitored on both sides of the border. Each of the 30 plots (15 on each side of the border) was 150 × 150 m (2.25 ha). The agricultural fields were comprised mainly of irrigated date palms, seasonal onions, melons, and tomatoes.

2.3. Timeframe and replications

Each of the 30 plots was sampled four times a year; winter, spring, summer, and autumn. The precise timing of sampling sessions was chosen based on temperature, with mid-winter (January–February) and mid-summer (July–August) sampling sessions taking place during the extreme cold and hot seasons, respectively. Sampling dates within a given season were chosen according to lunar phase. All sites were sampled either immediately before or after the new moon. In each season, plots were sampled for three consecutive nights and days. Every night, four plots were sampled in parallel (two in Israel and two in Jordan) in a fixed order to ensure that the same landscape units were sampled at the same time on both sides of the border (i.e., to avoid temporal bias). Thus, sampling started in four plots six nights before the new moon and progressively moved to the next plots until sampling was completed in the last plots on the sixth night after the new moon.

2.4. Sampling method

We constructed 20 pitfalls (five sets of four pitfalls) in each plot. Each pitfall was an 18 L bucket, equipped with a movable double bottom. A set was comprised of four pitfalls with a central bucket from which three drift fences (18 m each) projected to three peripheral buckets (see Appendix B and Shanas et al., 2006). Between trapping sessions, we sealed the buckets with lids and lowered the drift fences. Drift fences were reset and pitfalls were opened before sunset of the first night in each trapping session. Buckets were emptied the next three mornings before sunrise and the two following afternoons. All invertebrates that fell into the pitfalls were lifted out of the bucket using the double bottom, and the entire sample was funneled into jars containing alcohol. These invertebrates were later separated from the sand with entomological tweezers under magnifying glasses and then sorted into three major taxonomic groups: Coleoptera, Araneae and others. We assembled Coleoptera species and Araneae family abundance data for each of the 30 sites. The data included all the specimens collected during the yearlong sampling and specimens collected during each season. Some specimens were identified only to the

genus or family level. In the case of Coleoptera over 95% of the individuals were identified to the species level, therefore we used the species level for analysis. Spiders were often not analyzed beyond the genus or family level [juveniles are notoriously hard to identify to the species levels, and thus family level is often used for analysis (Whitehouse et al., 2002; Gavish-Regev et al., 2008)]. Nevertheless, spider family diversity across the landscape units resembled the pattern found in spider species diversity (Spearman rank correlation coefficient: $\rho = 0.925$, $p = 0.0022$).

2.5. Data analysis

We compared species and family assemblages, and diversity indices among landscape units, across border, and between close and far to agricultural land. To compare species and family assemblages we used Primer 5 (Clarke and Gorley, 2001) to produce MDS graphs, after a square-root transformation of the data, and after constructing a Bray–Curtis similarity table. The species assemblages were analyzed separately for beetles and spiders across landscape units using ANOSIM (Primer 5). We used EstimateS (Colwell, 2004) for calculating diversity measures (Simpson and Fisher alpha). Statistical tests for comparing diversity indices were computed by using the Statview 5 for the Macintosh computer. To compare the landscape units, we computed ANOVA tests separately for the close sites within Israel and the far sites within Jordan. When data were not normally distributed and transformations failed to achieve normality, a non parametric test was employed (Spearman rank correlation test was employed to examine correlation between family and species richness across the landscape units, and Mann–Whitney test to compare diversity between close and far sites). We also attempted to look into the occurrence of beetle and spider families in close vs. far from agriculture fields. This was based on family groups that comprised at least 10 individuals in the sample.

In addition, we examined seasonality in abundances of both groups. To test the usefulness of collecting only representative species, we used the SIMPER function of Primer 5 to find the best representative Coleoptera species. To compare the landscape clustering based on these few Coleoptera species to the one based on all 195 species, we used the RELATE function of Primer 5. The RELATE function was further used to examine whether the Coleoptera distribution is congruent with other groups and thus it was compared to Araneae and to Reptilia distribution. Reptilia data was retrieved from a previous study (Shanas et al., 2006).

3. Results

We collected 11,499 beetles, belonging to 26 families, and 3546 spiders, belonging to 26 families. Since not all individuals were identified to the species level, the counts of 195 beetle species, and 45 spider species represent an underestimation of the actual resident species in the study area. However, we found a positive correlation between family and species richness across the landscape units (Spearman Rank Correlation, Coleoptera: $\rho = 0.726$, $p < 0.0001$; Araneae: $\rho = 0.962$, $p < 0.0014$).

3.1. Landscape units

Both cluster analysis and MDS analysis of Coleoptera species assemblages show that the landscape units cluster into three groups at a similarity level of 30%: SM, HD, and sandy habitats (SD, MX) (Fig. 1, full circles). Species assemblage was significantly different across landscape units (ANOSIM, $R = 0.693$, $p = 0.001$). However, post-hoc analysis revealed no difference between the two sandy habitats (ANOSIM, $R = 0.011$, $p > 0.05$). Geographic distance

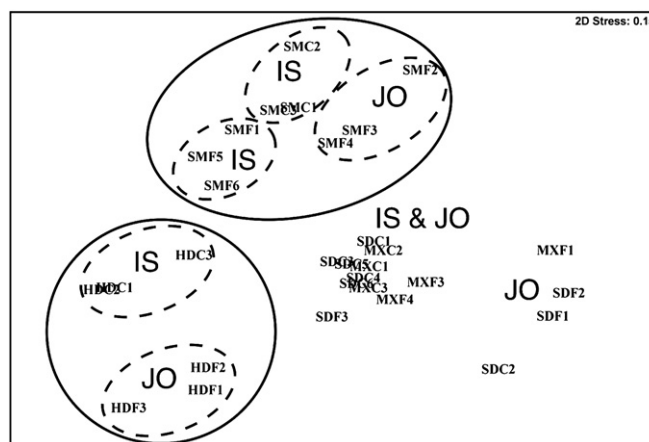


Fig. 1. MDS ordination of landscape units based on Coleoptera species. Full circles encompass high density Acacia (HD) and salt marsh (SM) landscape units, while dashed circles encompass same landscape units within a given country. Other landscape units are either sand dune (SD) or sandy mixed gravel (MX), C – close to farms, F – far from farm. All landscape units are in small capitals. In large capitals: IS – Israel, JO – Jordan.

between sites did not explain community similarity. For example, the geographic distance between the high density site far from agriculture, HDF1, and the high density site close to agriculture, HDC3, was 23 km, and only 5.2 km away from a salt marsh site close to agriculture, SMC1. Nevertheless the multivariate analysis shows that HDF1 clusters with HDC3 and not with the geographically closer site, SMC1 (Fig. 1).

For spider family assemblages, the similarity between plots failed to cluster according to landscape units (Fig. 2). However, spider family composition differed significantly among landscape units (ANOSIM, $R = 0.181$, $p = 0.005$).

Coleoptera Simpson diversity seemed to show a difference across the landscape units (Fig. 3), however it was not statistically different in either Israel or Jordan (ANOVA, $F_{3,8} = 3.803$, $p = 0.0581$; $F_{3,8} = 3.830$, $p = 0.0572$ respectively). Araneae diversity did not differ across the landscape units in Israel (ANOVA, $F_{3,8} = 1.818$, $p > 0.05$), or across the landscape units in Jordan (ANOVA, $F_{3,8} = 1.087$, $p > 0.05$) (Fig. 3).

3.2. Border effects and proximity to agriculture

Based on the Coleoptera data, landscape units tended to cluster according to the geopolitical location (Fig. 1; dashed circles), however no significant difference was found in Coleoptera diversity

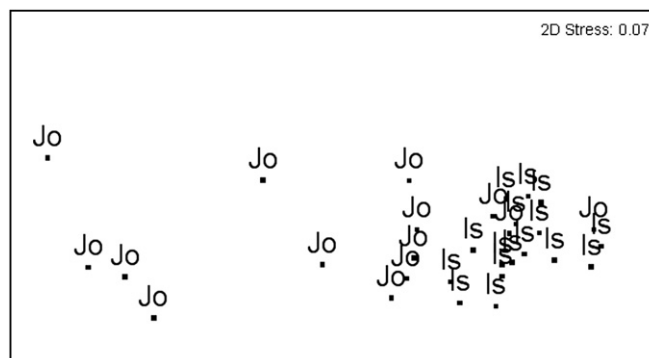


Fig. 2. MDS ordination of landscape units based on Araneae families located in Jordan (Jo) or in Israel (Is).

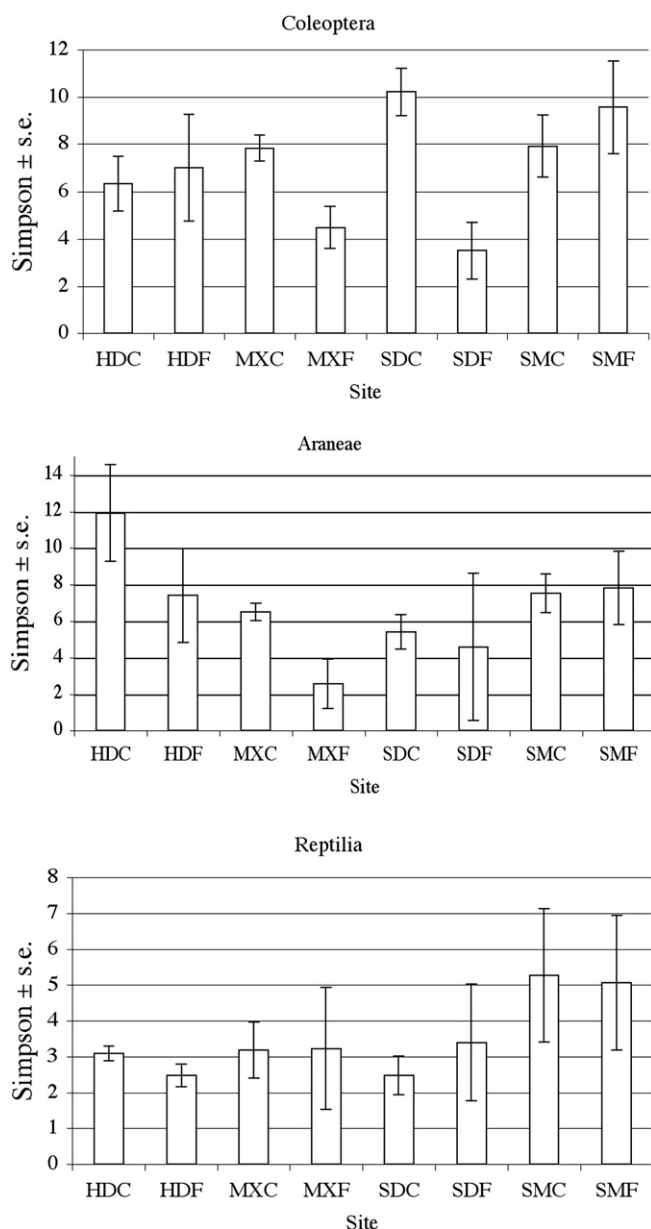


Fig. 3. Species diversity (Simpson) across landscape units. Araneae are represented by families (Reptile data from Shanas et al., 2006).

across the border (Paired sign test, $p > 0.05$). Araneae showed a clear trend of clustering by geopolitical location (ANOSIM, $R = 0.405$, $p = 0.001$, Fig. 2).

Coleopteran species composition showed a significant dissimilarity between the close and the far landscape units (ANOSIM, $R = 0.271$, $p < 0.05$). Coleoptera had higher diversity in close versus far SD in Jordan (Mann–Whitney, $Z = -1.964$, $p < 0.05$), however, no difference was found between close and far SM sites within Israel (Mann–Whitney, $Z = -0.655$, $p > 0.05$). Two Coleoptera families – Apionidae, phytophagous; and Oedemeridae, pollen feeders – occurred only at the far sites.

Spider families were dissimilar between close and far landscape units (ANOSIM, $R = 0.237$, $p < 0.05$), however the diversity (Simpson) of spiders did not differ between the close and far landscape units within either Jordan or Israel (Mann–Whitney, $Z = -7.07$, $p > 0.5$ for SD in Jordan, $Z = -0.655$, $p > 0.5$ for SM in Israel). One spider family, the nocturnal hunters Clubionidae, occurred only at close sites.

3.3. Seasonality

Beetles and spiders showed seasonality in the number of species observed. Fall had the lowest number of total species during the two years sampling (Beetles $S_{obs} = 40, 42$; Spiders $S_{obs} = 14, 11$), and spring had the highest (Beetles $S_{obs} = 75, 76$; Spiders $S_{obs} = 22, 21$).

3.4. Surrogacy and congruence

Using the SIMPER function we found that six species of Coleoptera, all belonging to the Tenebrionidae, could explain very well (RELATE, $\rho = 0.734$, $p = 0.001$, Fig. 4) the group clustering of landscape units based on all species within the Coleoptera (Fig. 1). These species were: *Adesmia Montana* (associated with HD), *Cheirodes asperulus* (SM), *Erodius reichei* (SD), *Mesostena angustata* (MX), *Pimelia arabica* (SD), and *Procoma audouni* (SM). We failed to find such surrogate species within the Araneae.

Using the RELATE function, we found that beetles and spiders showed relatively low congruence ($\rho = 0.575$, $p = 0.001$), with even lower congruence between beetles and reptiles ($\rho = 0.379$, $p = 0.001$), and spiders and reptiles ($\rho = 0.335$, $p = 0.001$) (reptile data taken from Shanas et al., 2006).

4. Discussion

Contemporary land management and conservation decisions are made within geopolitical or, even more local, jurisdictional boundaries (Hunter and Hutchinson, 1994; Rodrigues and Gaston, 2002). However, since ecological systems are usually not bound within these artificial human-created borders, it is questionable whether spending the efforts on cross border management or rather acting on a more parochial way will be more effective in conserving ecological systems within large regions. The Arava valley, politically shared by Israel and Jordan provides an excellent model to examine this question.

Our beetle results (Fig. 1) clearly suggest that although both countries contain similar habitats, each has its own unique characteristics, probably due to different cultural practices. The fact that the different landscape units on both sides of the border (HD, SM, and sandy habitat units) are clustered separately means that each unit in Israel is more similar in species composition to that same unit in Jordan than to a geographically closer but different

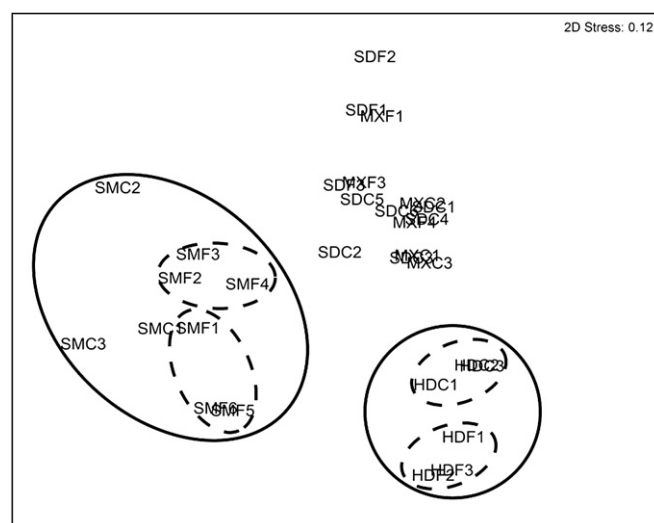


Fig. 4. MDS ordination based on six Coleoptera species. Symbols same as in Fig. 1.

landscape unit within Israel. However, our analysis further demonstrates that within each landscape unit, dissimilarities between sites exist based on the geopolitical location. The spider data (Fig. 2) show this trend most clearly. Though the landscape units are not distributed on the MDS graph according to their type, which might contradict the ordination graph based on beetle assemblages, they are clustered as almost a single group within Israel. These findings are consistent with our previous results from rodent and reptile assemblages that demonstrate fauna differences across the border (Shanas et al., 2006). The results could have arisen from differences across the border, due to a west-east environmental gradient. However, this is very unlikely, since the Arava valley is only 10 km across, and sites across the border were within the same altitude and shared the same vegetation type. We therefore suggest that land use prioritization of the Arava valley should be conducted on a national rather than trans-national level. Loss of a landscape unit on one side of the border cannot be compensated with preservation of the same landscape unit across the border, due to fauna dissimilarity. Therefore the landscape units across the border should be regarded as complements. This recommendation for updated management practice may apply also to other cross border ecosystems, especially where long term cross border cultural differences result in dissimilar effects on the ecosystem (Homewood et al., 2001).

Locally, our results support our *a priori* division of the valley habitats into the selected landscape units. However the MDS analysis further shows that the two sandy habitats (SD and MX) are indistinguishable. This is of crucial importance to the current management of the Arava sands. Sand dunes (SD), though abundant in Jordan, have been vastly exploited in Israel, mainly for agriculture. Currently, they are highly valued in the public view, and thereby receive relatively high level of protection. In contrast, many of the non-dune sandy habitats (MX) are considered to be of relatively low value and are very often used for agricultural development and expansion. Our study suggests that the MX habitat should not receive a lower prioritization value than the SD. Indeed, only when these habitats are carefully surveyed are conservation values revealed (e.g., finding a locally rare *Jaculus jaculus* on these lands; Shanas et al. unpublished). We therefore suggest that the current practice of expanding farms based on the low prioritization value given to non-dune sandy habitats should be revisited.

Although the use of faunal surrogates can presumably be an efficient way to characterize and to prioritize lands, it has been demonstrated that congruence of different taxonomic groups is not common (Caro and O'Doherty, 1999; Moore et al., 2003; Warman et al., 2004; Bonn and Gaston, 2005; Favreau et al., 2006). We compared the diversity of three groups, and found that each group had its own pattern of diversity across the different landscape units (Fig. 3; also according to Fisher alpha diversity index, see Appendix C). This finding not only undermines the use of surrogacy for prioritization, it questions the ability to actually prioritize the land based on diversity measures of indicator groups. We compared the average similarity of different landscape units (based on Morisita-Horn, see Appendix D), and found too, that different taxa hold different values. Furthermore, we found that a prioritization value based on similarity data could contradict one that is based on the diversity data. For example, spider diversity showed the highest value in HDC landscape unit. However this landscape unit showed the lowest value of similarity. Therefore, while HDC might be valued highly for prioritization based on diversity, it will do poorly to preserve fauna in other landscape units. Opposite results were found for the MXF that had high similarity value but very low diversity index. These results emphasize the limit of biological indicators as tools for land prioritization. We suggest that such

indicators should be taken into consideration but only under a wider spectrum of analysis, including socio-economic and political dimensions (Watzold et al., 2006).

Agricultural farming has a direct effect on species decline through habitat destruction (Green et al., 2005; Foley et al., 2008). In desert ecosystems, agricultural practices introduce additional water and nutrition resources to a relatively oligotrophic environment (Balba, 1995) and may result in the enhancement of species invasions (e.g. invasion of red fox to the southern Arava valley following the establishment of irrigated agricultural farms, Mendelsohn and Yom-Tov, 1999). Agricultural practices, even at low intensity, have been demonstrated to affect both spider and beetle diversity (Pffner and Luka, 2003). However, the change in diversity may take an unpredictable course, depending on the habitat and the existing fauna. For example, spiders may show a decrease (Shochat et al., 2004) or an increase (Siemann, 1998) in diversity in response to increased productivity. We found that the effect of agricultural land on the nearby landscape's species diversity depends on the taxonomic group under consideration and on the habitat (Fig. 3). Beetles showed higher diversity on SD but not on SM close to agricultural fields, while spiders' diversity was not affected by the proximity to agriculture. These results may be explained by the contrasting effects agricultural fields may have on nearby desert habitats. While the productivity next to agricultural fields is expected to be higher, especially in comparison to other extreme desert habitats, and thus to increase the species diversity, some agricultural practices such as pesticide use might put a pressure on both invertebrates and vertebrates species. In addition, many of the farms in the Arava are bounded with tree windbreakers, which might offer vantage points for foraging birds and thus increase the pressure on vertebrate (Hawlena and Bouskila, 2006) and probably invertebrate prey alike. We found two beetle families to be absent from sites that are close to agricultural fields. These two families Apionidae and Oedemeridae were found mainly in the SM far sites in Jordan. *A priori*, it would have been hard to recognize them as agriculturally-sensitive families based on their phytophagous and pollen feeder guilds, respectively. Nevertheless, it appears from our data that the main effect of intensive agriculture on spiders and beetles is within relatively close proximity to farms. We, therefore, suggest that the edge effect of farms on these two groups is relatively narrow in this extreme desert ecosystem.

Without a doubt, the Arava valley is under increasing pressure for development in the post-conflict era. Therefore, in addition to identifying the most appropriate land for preservation, it will also be necessary to monitor this region in a cost effective way. The uniqueness of the various landscape units contribute to the general diversity and the stability of the ecosystem, therefore, a monitoring program should be implemented to quantify the rate of changes to the land. We found that as few as six common species of beetles could be used to describe the current dissimilarity among landscape units; providing the same result found by using all 195 species of beetles (Fig. 4). Though these beetle species might be good landscape indicators, they should not be confused with indicator or surrogate taxonomic groups, as they do not necessarily represent other taxonomic groups as demonstrated above.

Based on this study, we are not able to recommend specific landscape units or sites for conservation or for land transformation. However, the study suggests caution when contemplating losing diversity at the level of landscape units. In this study we have shown that differences on two dimensions, landscape unit type and cross border, are important for the overall species diversity of this ecosystem and therefore "redundancy" should be carefully applied, especially across geopolitical borders.

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Appendix. Supplementary material

Supplementary data related to this article can be found online at doi:10.1016/j.jaridenv.2010.10.008.

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