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# Orthoptera assemblages as indicators for the restoration of sand grassland networks

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**Abstract.** Orthoptera are prime indicators of changes in habitats and habitat networks. Sand grasslands, one of the most important orthopteran habitats, have become refuges for specialized, endangered species in much of Europe. Grassland patches of different sizes and levels of fragmentation within the Günyűi-homokvidék Natura 2000 site (HUFH20009) were restored in 2012. Here, we investigated the effects of changes to extended, opened, and less isolated sand grassland patches of the habitat network on rehabilitation of Orthoptera assemblages. Our main results showed: (1) species richness and relative frequency of sand habitat specialists were mainly driven by the proportion of bare soil surface (low vegetation cover), local extent of sand grassland patches, and fine fraction of the soil structure; (2) restoration of sand grasslands promotes the protection of several orthopteran species considered Vulnerable or Endangered in Europe; and (3) minimizing disturbance of the soil (no stump removal or soil tillage) during restoration is crucial for the enhancement of orthopteran populations related to sand grasslands.

**Key words**: density, habitat specialist, landscape, soil, species richness, vegetation-structure

## Introduction

Insect diversity is declining drastically (Samways 2018) caused by habitat fragmentation, land-use intensification and forestation (Travis 2003, Tscharntke et al. 2005). Based on this, a considerable attention is devoted to the maintenance and restoration of natural and semi-natural habitats (Borchard et al. 2017, Tonietto & Larkin 2017). It can be achieved by active habitat restoration (Maron et al. 2012), which aims to improve habitat connectivity (Saunders et al. 1991, Fartmann 2006, Roesch et al. 2013, Poniatowski et al. 2016) between fragmented patches with corridors covered by adequate habitat structure (Valtonen & Saarinen 2005, Baur et al. 2017, Zalewski et al. 2012). It is important to note, that restoration success (Schirmel et al. 2010, 2015) is depend on inherent spontaneous colonization possibilities of surrounding areas (Prach et al. 2015).

In its westernmost regions, Eurasian steppe vegetation takes the form of sand grasslands of Central and Eastern Europe (Molnár et al. 2012), usually defined as xeric sand calcareous grasslands (code: 6120, 2002/83/EC Habitat Directive) (Király et al. 2015). Most of the naturally treeless habitats of the sandy steppes have been fragmented and significantly affected by forestation (Krištín et al. 2004). Mitigating these detrimental effects, however, are patches of sand grassland that have been excluded from forestation, and which are suitable for preserving rare habitat specialists (Erdős et al. 2015). Group of habitat specialist insects of sand grasslands includes several species being Endangered or Vulnerable in Europe (Warren & Büttner 2008, Gardiner & Benton 2009, Holuša 2012, Hochkirch et al. 2016).

Since the rate and pace of their response to restoration differ significantly, it is always necessary to investigate several taxa in parallel (Alignan et al. 2014). In the most cases the latter is difficult or expensive (Pakkala et al. 2014). Choosing one indicator taxon is based on that, its characteristics indicate status of the environmental conditions of interest precisely (Dallinger et al. 1992, Hilty & Merenlender 2000, Pakkala et al. 2014). Due to their sensitivity to changes in habitat structure (Samways 1997), orthopterans are prime indicators of the effectiveness of habitat network restoration (Bazelet & Samways 2011, Alignan et al. 2018). The character of the landuse strongly influences microclimate, vertical and horizontal structure of vegetation (Guido & Chemini 2000, Kruess & Tscharntke 2002, Knop et al. 2006, Kenyeres & Cservenka 2014), and orthopteran assemblages, with shifts in their community composition, respond quickly to environmental changes (Torma & Bozsó 2016, Zografou et al. 2017), disturbance (Heneberg et al. 2016, Řehounková et al. 2016) and habitat restoration (Borchard et al. 2013, Alignan et al. 2014).

In this study, we focused on patches of restored sand grassland where the soil had not been disturbed. Restoration was carried out by removal of black locust (*Robinia pseudoacacia*), black pine (*Pinus nigra*) trees, shrub cutting, restitution of grazing, or post-treatment by mowing. Restoration of the landscape was carried out in 2012, post-treatment activities have been started in 2013.

To compare the restoration success of orthopteran assemblages in sand grasslands, we evaluated the following hypotheses: (1) We expected species richness and density of the habitat specialists to increase on all sites. (2) Changes in assemblages are affected by natural state before restoration and size of the restored sand grassland patches. (3) Enlarging of habitat specialists' density is not independent from vegetation and soil structure, also from the degrees of isolation.

## Material and Methods

### Study area

The study area (Kisalföld, Hungary, centre of the study area: N47.71482 E17.80208) consisted of 700 ha, and was found at an altitude of 112–122 metres a.s.l. It is characterized by relatively flat, low, broad-backed sand hill plateaus and wet flatlands. The total duration of annual insolation reaches 2,000 hours in the region, the mean annual temperature is around 10.0 °C, the annual mean temperature during the growing season is 16.0 °C. The annual mean values of absolute maximum and minimum temperatures are 33.5–34.0 °C and

-16.0 – -16.5 °C. The annual precipitation is 580–620 mm, and the number of snow-covered days is 35–38. The study area is generally covered by sand, chernozem and, in some places, by meadow soils (Dövényi 2010).

The study area was almost entirely covered by grasslands until the end of the 18th century, dominated by pastures and open sand surfaces with a minor presence of arable lands. The forest cover during that time was 21 hectares (http://mapire.eu/hu/map/ firstsurvey). By the mid-19th century, the expansion of arable lands showed some growth in sub-regions characterized by humic soils. However, the expansion of forest areas grew fivefold (120.5 hectares, http://mapire.eu/hu/map/secondsurvey). Forestation took place artificially, by planting black locust and pine trees in order to bind quicksand and improve barren soils. As a result, the extent of forests in the region amounted to 4.407 ha by the beginning of the 20th century. The conversion of grasslands reached its peak in the 1950s, and the landscape structure had not changed significantly until 2012 when the present study began. In the second half of the 20th century, individual trees of the exotic cultivated species and invasive herbaceous species began to spontaneously and frequently appear in the vast majority of the grasslands. Grasslands which remain in a high-quality natural state have been maintained all along, but they are isolated to varying degrees from each other (Bozsóki & Takács 2015).

In 2012 (from October to December) restorations took place at the study area which significantly improved the naturalness of the local habitats (Király et al. 2015). As a result, in the western part of the study area the extent of open and dry grasslands grew 2.3–7.3 fold (mean: 4.2) per sampling site due to deforestation, bush clearing, and grazing, while the extent of areas covered by woody or shrubby vegetation shrank (Fig. 1). In the region of the highly isolated (IS) sampling sites found in the eastern half of the study area, the proportion of grassland habitats increased 4.1–31.3 fold (mean: 13.4) due to removal of pine and black locust trees, followed by mowing as post-treatment (Fig. 1). Changes in habitat structure were calculated using QGIS 2.16 (QGIS Development Team 2016).

Experimental design

We established 10 sampling sites of 50×50 m sized quadrats. We classified the sampling sites based on botanical examinations of Király et al. (2015): habitat structure of grasslands with high botanical naturalness were handled natural (NAT); more or less weedy grasslands having pine trees in that at the start of the study were handled semi-natural (SEMINAT); sampling site was covered by closed black locust forest prior to 2012 was handled as restoration site (RES). Of the sampling sites without robust isolation by forests, four were considered to be natural sand grassland (referred to as NAT1, NAT2, NAT3, NAT4, see also Fig. 1.) and one weedy site was considered to be semi-natural (referred to as SEMINAT1). Of the sampling sites isolated by forests, two had natural sand grassland vegetation (referred to as ISNAT1, ISNAT2), two semi-natural sites were covered by pine trees (referred to as ISSEMINAT1, ISSEMINAT2) prior restoration, and one semi-natural site contained a black locust forest prior to 2012 (referred to as ISRES1). For determination the habitat structure of the sampling sites being potentially accessible for the local orthopterans (sand grasslands without forest or shrub patches and trees), the area of sand grasslands was measured within 100 m circular buffers around the centre point of each sampling site using QGIS 2.16 (Fig. 1).

Soil

Sand soil samples were collected from 3 plots selected pseudo-randomly (same plots on which the vegetation parameters were recorded) per sampling site in 2015. The following soil parameters were measured using laboratory analyses, as these are considered to be most important: $CaCO_3$ content (%), humus content (%), pH-$H_2O$, pH-KCl, and proportions (%) of the following soil component granular sizes: gravel (2–4 mm); sand / very coarse (1–2 mm), coarse (0.63–1 mm), intermediate coarseness (0.2–0.63 mm), slightly coarse (0.1–0.2mm), fine (0.05–0.1 mm), powder (0.02–0.05mm), silt (0.002–0.02 mm), and loam (<0.002 mm). Means of all soil parameters were calculated and the following classes were pooled: very coarse and coarse fractions (=coarse fraction); slightly coarse, fine and powder fractions (=fine fraction).

Vegetation and landscape structure

Vegetation parameters were measured in 3 pseudo-randomly selected plots per sampling site in June and July of 2012 and 2016. The following parameters were recorded: average height of the vegeta-

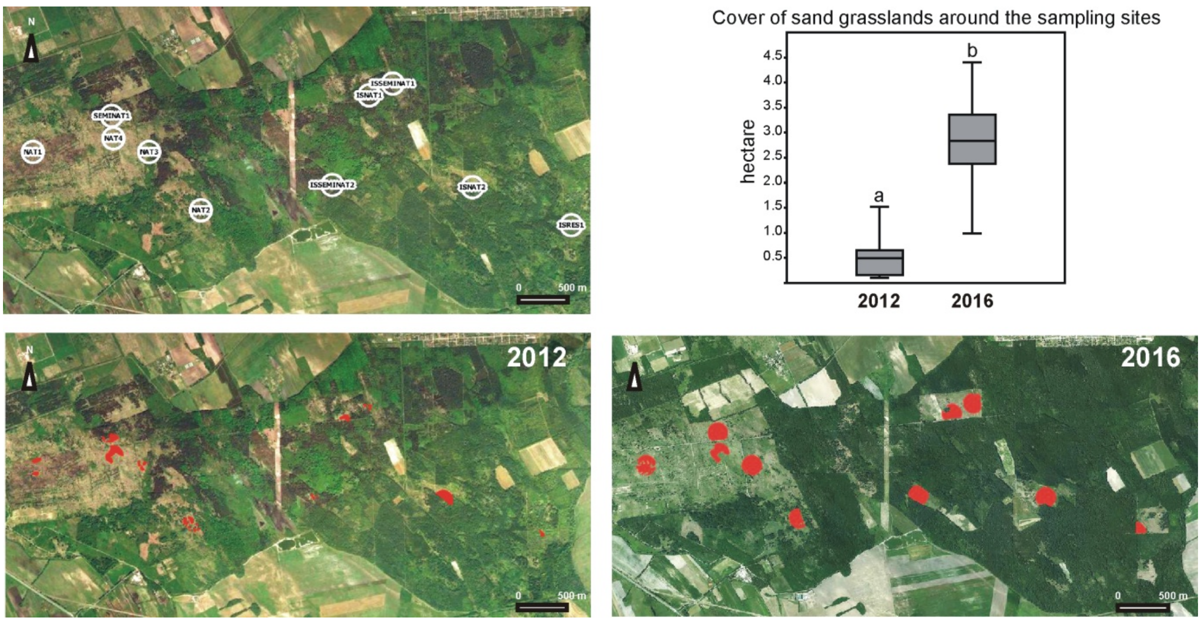


Figure 1 The sampling areas. Cover of sand grasslands within 100-m circular buffers around the sampling sites in 2012 and 2016 is shown with red patches (on the basis of their vegetation structure in 2012 sand grasslands NAT1, NAT2, NAT3, NAT4 are non-isolated natural, SEMINAT1 is semi-natural, ISNAT1 and ISNAT2 are isolated natural, ISSEMINAT1 and ISSEMINAT2 are isolated semi-natural covered by pine trees, ISRES1 is reconstructed from black locust forest)(digitised by the authors with using of QGIS 2.16 and aerial photographs of Google Earth. 13 August, 2012. 29 July, 2017). Increasing in cover of sand grasslands showed significant ($p<0.001$) differences by Wilcoxon test.

tion (cm), total vegetation cover (%), bare soil (%).Total cover of the vegetation was measured in a square metre quadrat. Percentage cover of each plant species was also estimated. Each parameter was averaged year and per sampling site.

Orthoptera

Orthoptera were sampled in June, July, and August of 2012 and 2016, twice per month (with intermissions of 15–20 days). Sweep-netting was conducted on randomly-selected patches within each sampling site (altogether 120 samples: 2012: 10 sites / 2 surveys per sites in June, July and August; 2016: 10 sites / 2 surveys per sites in June, July and August). Orthoptera densities were recorded by 300 sweeps. The collection was standardized as much as possible. Orthopterans were collected in 10 collateral transects, by covering 10 m for every 30-sweep transect. Orthoptera species were identified using works of Harz (1969, 1975). Based on Rácz (1998), Ingrisch & Köhler (1998) and Krištín et al. (2009), *Aiolopus thalassinus, Calliptamus barbarus, Dociostaurus brevicollis, Euchorthippus pulvinatus, Myrmeleotettix antennatus, Myrmeleotettix maculatus, Oedaleus decorus, Montana montana,* and *Stenobothrus fischeri* were classified as habitat specialists of local sand grasslands (psammophilic and pseudopsammophilic species). Scientific nomenclature follows Cigliano et al. (2017).

Statistical analysis

Orthoptera samples collected in the same sampling sites in the same year were pooled (number of pooled samples was 20). We used Wilcoxon's test to compare assemblages occurring on sites in 2012 and in 2016 (relative frequencies of species were transformed using the Isometric Burnaby method (Elliott et al. 1995)). Orthoptera species richness, density (individual/10 m²) of assemblages, relative frequencies of habitat generalist species of dry grasslands, furthermore species richness, relative frequencies and densities of habitat specialist species were used as response variables of restoration. To eliminate multicollinearity, we analysed groups of recorded environmental variables using Spearman's rank correlation. In case of high intercorrelation, one of the variables was excluded from further analysis. Finally, we used four environmental variables to describe changes in habitat conditions: two of these reflected the habitat structure (total vegetation cover and average height of the grass); probability of colonisation was represented by cover (ha) of sand grasslands in 100 m circular buffers around the sampling sites; and soil conditions were represented as the percentage of the fine fraction. After normality test, Wilcoxon test and paired t-test was used to evaluate statistical differences among the derived variables in the two sampling years.

Orthoptera data and environmental variables were modelled using Canonical Correspondence Analysis (CCA) (presence-absence data of Orthoptera species present in at least two sampling sites; environmental data were log-transformed,). Only significant variables (Monte Carlo permutation test with 999 permutations at $p<0.05$) were regarded in results. All statistical analyses were performed in Past 3.14. (Hammer et al. 2001).

## Results

A total of 2,832 individuals of Orthoptera were recorded belonging to 35 species (see Appendix I). The most abundant species were *Euchorthippus declivus* with 580 individuals (20.4 %), followed by *Calliptamus italicus* with 526 individuals (18.5 %), *Calliptamus barbarus* with 329 individuals (11.6 %), *Euthystira brachyptera* with 187 individuals (6.6 %), *Euchorthippus pulvinatus* with 182 individuals (6.4 %), *Stenobothrus lineatus* with 169 individuals (5.9 %), and *Bicolorana bicolor* with 160 individuals (5.8 %). From the collected species, *Calliptamus barbarus* is protected in Hungary, *Myrmeleotettix antennatus* and *Montana montana* are Endangered, *Euchorthippus pulvinatus* is Vulnerable in EU28 (Hochkirch et al. 2016).

Wilcoxon tests showed significant differences (Table 1) among non-isolated orthopteran assemblages sampled before and after restoration, except locality of NAT4 affected by the lowest change in habitat-structure (increase in habitat area was <30%). Latter locality can be handled as control area testing dependency of changes on habitat restoration or just weather fluctuation. Of the isolated orthopteran assemblages two showed significant differences before and after restoration.

Table 1. Results of Wilcoxon tests on assemblages occurring on sites in 2012 and 2016 (n.s. not significant; * $p<0.05$; ** $p<0.01$; *** $p<0.001$)

| | Z | p |
|---|---|---|
| ISNAT1 | 1.409 | n.s. |
| ISNAT2 | 1.282 | n.s. |
| ISSEMINAT1 | 1.841 | ** |
| ISSEMINAT2 | 0.368 | n.s. |
| ISRES1 | 2.450 | * |
| NAT1 | 2.400 | * |
| NAT2 | 2.730 | ** |
| NAT3 | 2.501 | * |
| NAT4 | 0.698 | n.s. |
| SEMINAT1 | 2.095 | * |

In 2016, the assemblages were significantly richer in habitat specialist species, further density and relative frequency of those species was significantly higher in 2016 than before the habitat restoration (Fig. 2).

Total vegetation cover, percentage of bare soil and cover of sand grasslands within 100 m circular buffers around the sampling sites differed significantly between 2012 and 2016 (Table 2). From the examined soil parameters, just proportion of fine fraction of the soil differed significantly between sites having orthopteran assemblages being rich in and poor in habitat-specialist species (Table 3).

CCA showed separation of orthopteran assemblages of sampling areas before and after restoration (Fig. 3). Two predictor variables contributed significantly to the ordination model: the structure of the assemblages was mainly determined by gradients of total vegetation cover and cover of sand grasslands around the sampling sites. Presence of *Pseudochorthippus parallelus, Euthystira brachyptera, Chorthippus biguttulus, Pholidoptera fallax,* and *Phaneroptera falcata* were positively correlated with the total vegetation cover and negatively correlated with cover of sand grasslands. Furthermore, *Euchorthippus pulvinatus, Dociostaurus brevicollis,* and *Myrmeleotettix maculatus* were negatively correlated to high total cover of vegetation and positively correlated to high extent of sand grasslands (Fig. 3).

## Discussion

These results show that habitat restoration resulted in significant differences in the structure of orthopteran assemblages. Species richness, density and relative frequency of sand habitat specialist species significantly increased after

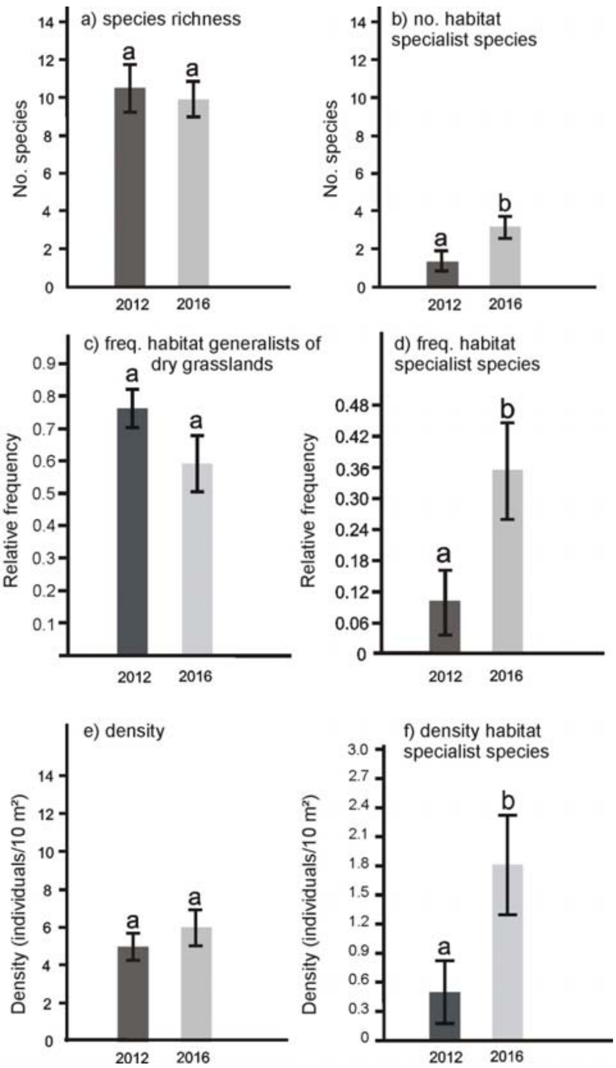


Figure 2 Mean values (±SE) of tested Orthoptera assemblage variables in 2012 and 2016 (significant (p<0.05) differences detected by paired t test are indicated by different letters)

Table 2. Mean values ± SE of recorded vegetation parameters. Differences among environmental parameters were analysed by paired t tests (n.s. not significant; * p<0.05; ** p<0.01; *** p<0.001)

| Parameter | 2012 | 2016 | Pair-wise comparison |
|---|---|---|---|
| Vegetation | | | |
| total cover (%) | 82.5±3.5 | 76.5±2.4 | * |
| height (cm) | 26.5±4.4 | 18.5±1.8 | n.s. |
| bare soil (%) | 11.9±3.2 | 18.5±3.3 | * |
| Landscape structure | | | |
| cover of sand grasslands (ha) | 0,6±0,17 | 2,8±0,29 | *** |

restoration occurred, and were mainly driven by total cover of vegetation, the fine fraction of the soil structure, and local extent of sand habitats. Our results confirmed that orthopterans quickly respond to changes of their habitat in ecological restoration context (Borchard et al. 2013, Alignan et al. 2018). Furthermore, species richness and relative frequency of sand habitat specialists seem to be much more suitable for detection of changes correlated with habitat restoration of sand grasslands than species richness, abundance and diversity of common or ubiquitous species.

Table 3. Mean values ± SE of recorded soil parameters. Differences among environmental parameters were analysed by paired t tests (n.s. not significant; * p<0.05; ** p<0.01; *** p<0.001)

| Soil parameters | Rich in Hab-spec | Poor in Hab-spec | Pair-wise comparison |
|---|---|---|---|
| $CaCO_3$ (%) | 10,1±1,4 | 10,0±1,4 | n.s. |
| humus content (%) | 1,9±0,1 | 3,4±0,8 | n.s. |
| pH-H2O | 7,8±0,0 | 7,5±0,1 | n.s. |
| pH-KCl | 7,6±0,0 | 7,3±0,2 | n.s. |
| rough fraction (%) | 0,2±0,1 | 0,6±0,2 | n.s. |
| middle-class fraction (%) | 22,6±0,7 | 26,4±3,3 | n.s. |
| fine fraction (%) | 68,7±0,5 | 62,7±1,3 | ** |
| silt fraction (%) | 6,3±0,8 | 7,9±1,9 | n.s. |
| loam fraction (%) | 2,1±0,1 | 2,4±0,3 | n.s. |

There was a clear positive impact of the total vegetation cover on sand orthopteran assemblages, which may be attributed to the energy relations of geophilic, psammophilic and pseudopsammophilic species with the soil surface (Stebaev & Nikitina 1976, Willott 1997). Further, it is also not surprising that fine fraction of the soil considerably influenced species richness and relative frequency, particularly for the habitat specialist species that lay their eggs in the soil (Ingrisch & Köhler 1998, Willott & Hassall 1998, Stauffer & Whitman 2007, Wünsch et al. 2011, Crous et al. 2014).

During opening of the habitats caused by restoration the total cover and the height of the vegetation decreased, the extension of bare soil increased. Accordingly, in the studied grasslands, similar to the former results (Sharma 1984, Ingrisch 1988, Johnson 1989, Quinn et al. 1991, Willott 1997, Schell & Lockwood 1997, El Shazly & Shahpa 2004), the high proportions of bare soil surfaces and low vegetation, supplementing by the caused warm-dry macro- and microclimatic conditions, were seen as decisive factors for the presence and density of the habitat specialist species.

Several sand grassland grasshopper species have restricted distributions or have been extirpated from regions of Central Europe due to the cultivation of non-native trees (mainly black locust, black pine, and tree of heaven (*Ailanthus altissima*); Holuša & Kocarek 2006). Based on the results of this study, restoration of sand grassland habitat networks may be beneficial for the populations of species considered Vulnerable (*Euchorthippus pulvinatus*), or Endangered (*Myrmeleotettix antennatus, Montana montana*) in Europe (Hochkirch et al. 2016). In addition, restoration might be beneficial for several further habitat specialist species, such as *Oedaleus decorus,* being already rare or narrowly distributed in most of Central Europe (Krištín et al. 2004, Kuřavová 2015), and *Calliptamus barbarus*. It should be noted, that *Calliptamus barbarus* did not colonize in the isolated patches despite the fact that mobile species can often travel between habitat patches regardless of the presence of corridors (Collinge 2000). Recolonization of *Montana montana* was also unsuccessful in the isolated patches. It was presumably caused by the following: for species with low mobility, a patch size of <0.5 ha and a distance between patches of >100 m represented the critical limit for recolonization (Kindvall & Ahlen 1992).

Our results showed that habitat specialist orthopterans respond rapidly to habitat restoration in sand grassland networks that have patches of high-quality, natural condition.

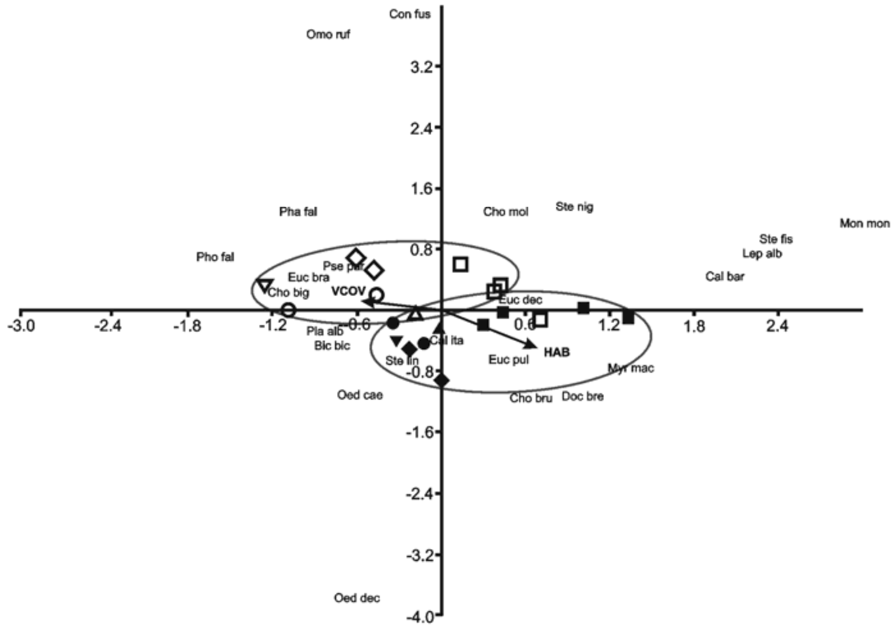


Figure 3 CCA ordination (sum of all eigenvalues: 0.732) based on Orthoptera data and environmental parameters. Only significant environmental parameters (VCOV: total vegetation cover; HAB: cover of sand grasslands in 100-m circular buffers around the sampling sites) are shown (Monte Carlo permutation test with 999 permutations at $p<0.05$). Environmental data were log transformed and presence-absence data of Orthoptera species presenting in minimum two assemblages were used. Patches of large sand grassland area: empty square=natural sand grassland in 2012; filled square=natural sand grassland in 2016; empty triangle=semi-natural sand grassland in 2012; filled triangle=semi-natural sand grassland in 2016; isolated habitat patches: empty circle=isolated natural sand grassland in 2012; filled circle=isolated natural sand grassland in 2016; empty diamond=isolated semi-natural sand grassland in 2012; filled diamond=isolated semi-natural sand grassland in 2016; empty inverse triangle=sand grassland reconstruction area in 2012; filled inverse triangle=sand grassland reconstruction area in 2016. Abbreviations of species names: Bic bic: *Bicolorana bicolor*; Cal bar: *Calliptamus barbarus*; Cal ita: *Calliptamus italicus*; Cho big: *Chorthippus biguttulus*; Cho bru: *Chorthippus brunneus*; Cho mol: *Chorthippus mollis*; Con fus: *Conocephalus fuscus*; Doc bre: *Dociostaurus brevicollis*; Euc dec: *Euchorthippus declivus*; Euc pul: *Euchorthippus pulvinatus*; Eut bra: *Euthystira brachyptera*; Lep alb: *Leptophyes albovittata*; Mon mon: *Montana montana*; Myr mac: *Myrmeleotettix maculatus*; Oec pel: *Oecanthus pellucens*; Oed cae: *Oedipoda caerulescens*; Oed dec: *Oedaleus decorus*; Omo ruf: *Omocestus rufipes*; Pha fal: *Phaneroptera falcata*; Pho fal: *Pholidoptera fallax*; Pla alb: *Platycleis albopunctata*; Pse par: *Pseudochorthippus parallelus*; Ste fis: *Stenobothrus fischeri*; Ste lin: *Stenobothrus lineatus*; Ste nig: *Stenobothrus nigromaculatus*; Tet vir: *Tettigonia viridissima*

This means that minimizing the disturbance of the soil (strict cessation of stump removal and soil tillage) during restoration is crucial for the promotion of orthopteran assemblages related to sand grasslands. Stump removal and soil tillage are a benefit for undesirable species of the soil seed bank (Mortimer et al. 1998). Additionally, high N availability of deforested areas favours fast growing weeds and invasive plants against native grassland species (Huenneke et al. 1990, Mclendon & Redente 1992). Disturbance of the soil during restoration, followed by weeds appearing, has different importance at different insect taxa. Colonization success of pollinators (Tonietto & Larkin 2017) or ground beetles (Kotze et al. 2011) is less affected by spreading of weeds or invasive plant species. Based on our study, Orthoptera, similar to butterflies (Sands & New 2013), belongs to taxa being really sensitive to general condition and weed invasion of restored habitats.

Implications for conservation

Based on our results, in restored grasslands characterized by open surfaces, suitable soil conditions (fine fraction (0.02–0.2 mm) of the soil of approximately 70%; humus content of approximately 1–2 %) ensure the conservation of habitat specialist orthopteran assemblages without any active post-restoration management. In restored grasslands that have been isolated due to habitat circumstances, and/or weedy due to landuse actions, active and responsive post-treatment management is recommended (Maron et al. 2012) in order to conserve Orthoptera. These post-treatment practices should be low intensity (Fartmann et al. 2012). In our opinion, in sand grasslands characterized by low productivity, this usually means extensive grazing, in which grazing pressure should be between 0.1–0.2 Livestock Units/ha. In case the spontaneous recolonization of restored patches fails to occur, even translocations of sand specialist species from populations of neighbouring areas should be considered (Gardiner 2010).

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+ Appendix I - 1 page Table, " Mean abundance and frequency of the collected orthopteran species in 2012 and 2016 per habitat-types "

Appendix I. Mean abundance and frequency of the collected orthopteran species in 2012 and 2016 per habitat-types

| Taxa/Code | NAT 2012 | | NAT 2016 | | ISNAT 2012 | | ISNAT 2016 | | SEMINAT 2012 | | SEMINAT 2016 | | ISSEMINAT 2012 | | ISSEMINAT 2016 | | ISRES 2012 | | ISRES 2016 | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | ab. | freq. | ab. | freq. | ab. | freq. | ab. | freq. | ab. | freq. | ab. | freq. | ab. | freq. | ab. | freq. | ab. | freq. | ab. | freq. |
| *Aiolopus thalassinus* (Fabricius, 1781) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.5 | 0.008 | 0 | 0 | 0 | 0 |
| *Bicolorana bicolor* (Philippi, 1830) | 4.5 | 0.026 | 9.5 | 0.035 | 17.5 | 0.104 | 11 | 0.116 | 2 | 0.015 | 2 | 0.022 | 11 | 0.135 | 8 | 0.051 | 4 | 0.118 | 6 | 0.063 |
| *Calliptamus barbarus* (Costa, 1836) | 21.75 | 0.146 | 60 | 0.388 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.006 | 0 | 0 | 0 | 0 |
| *Calliptamus italicus* (Linnaeus, 1758) | 20 | 0.131 | 28.75 | 0.072 | 29 | 0.124 | 23 | 0.232 | 41 | 0.299 | 32 | 0.348 | 9.5 | 0.098 | 48 | 0.362 | 0 | 0 | 39 | 0.406 |
| *Chorthippus biguttulus* (Linnaeus, 1758) | 6.75 | 0.054 | 0.75 | 0.002 | 9 | 0.038 | 2 | 0.021 | 10 | 0.073 | 0 | 0 | 8 | 0.104 | 0 | 0 | 3 | 0.088 | 9 | 0.094 |
| *Chorthippus brunneus* (Thunberg, 1815) | 5 | 0.048 | 3.75 | 0.019 | 0 | 0 | 2.5 | 0.027 | 3 | 0.022 | 11 | 0.120 | 0 | 0 | 12.5 | 0.095 | 0 | 0 | 9 | 0.094 |
| *Chorthippus dichrous* (Eversmann, 1859) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.007 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Chorthippus mollis* (Charpentier, 1825) | 4.75 | 0.028 | 3 | 0.021 | 0 | 0 | 0 | 0 | 12 | 0.088 | 6 | 0.065 | 6.5 | 0.076 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Conocephalus fuscus* (Fabricius, 1793) | 0.75 | 0.004 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 5 | 0.052 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Dociostaurus brevicollis* (Eversmann, 1848) | 5 | 0.032 | 5.75 | 0.047 | 0 | 0 | 1 | 0.010 | 0 | 0 | 1 | 0.011 | 0 | 0 | 8.5 | 0.066 | 0 | 0 | 0 | 0 |
| *Ephippiger ephippiger* (Fiebig, 1784) | 1 | 0.006 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Euchorthippus declivus* (Brisout de Barneville, 1848) | 46.75 | 0.297 | 44.75 | 0.174 | 45.5 | 0.194 | 9 | 0.092 | 59 | 0.431 | 20 | 0.217 | 6 | 0.063 | 4.5 | 0.044 | 0 | 0 | 5 | 0.052 |
| *Euchorthippus pulvinatus* (Fischer de Waldheim, 1846) | 3 | 0.019 | 28.75 | 0.118 | 0 | 0 | 17 | 0.176 | 0 | 0 | 3 | 0.033 | 2.5 | 0.026 | 6.5 | 0.045 | 0 | 0 | 0 | 0 |
| *Euthystira brachyptera* (Ocskay, 1826) | 1.25 | 0.007 | 1.5 | 0.006 | 52.5 | 0.342 | 14 | 0.149 | 0 | 0 | 2 | 0.022 | 8 | 0.085 | 1.5 | 0.008 | 22 | 0.647 | 0 | 0 |
| *Gomphocerippus rufus* (Linnaeus, 1758) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2.5 | 0.026 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Leptophyes albovittata* (Kollar, 1833) | 0.75 | 0.004 | 1.5 | 0.008 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Montana montana* (Kollar, 1833) | 0.75 | 0.008 | 0.5 | 0.004 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Myrmeleotettix antennatus* (Fieber, 1853) | 0.75 | 0.005 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Myrmeleotettix maculatus* (Thunberg, 1815) | 2.5 | 0.016 | 5.5 | 0.048 | 0 | 0 | 1 | 0.010 | 0 | 0 | 3 | 0.033 | 0 | 0 | 2.5 | 0.024 | 0 | 0 | 0 | 0 |
| *Oecanthus pellucens* (Scopoli, 1763) | 0.5 | 0.003 | 2.25 | 0.006 | 2 | 0.009 | 0 | 0 | 1 | 0.007 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Oedaleus decorus* (Germar, 1826) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 17 | 0.102 | 0 | 0 | 0 | 0 |
| *Oedipoda caerulescens* (Linnaeus, 1758) | 0.75 | 0.004 | 3.25 | 0.008 | 0 | 0 | 0 | 0 | 3 | 0.022 | 6 | 0.065 | 0.5 | 0.005 | 8 | 0.057 | 0 | 0 | 22 | 0.229 |
| *Omocestus rufipes* (Zetterstedt, 1821) | 1.75 | 0.010 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7 | 0.088 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Pezotettix giornae* (Rossi, 1794) | 0.75 | 0.004 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Phaneroptera falcata* (Poda, 1761) | 0.5 | 0.006 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.011 | 1 | 0.013 | 0 | 0 | 2 | 0.059 | 0 | 0 |
| *Pholidoptera fallax* (Fischer, 1853) | 0 | 0 | 0 | 0 | 2.5 | 0.011 | 0 | 0 | 0 | 0 | 0 | 0 | 1.5 | 0.015 | 2.5 | 0.014 | 0 | 0 | 0 | 0 |
| *Pholidoptera griseoaptera* (De Geer, 1773) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.5 | 0.006 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Platycleis grisea* (Goeze, 1778) | 0.5 | 0.004 | 1.75 | 0.004 | 6 | 0.026 | 1.5 | 0.016 | 0 | 0 | 0 | 0 | 6 | 0.070 | 6 | 0.040 | 0 | 0 | 5 | 0.052 |
| *Pseudochorthippus parallelus* (Zetterstedt, 1821) | 4 | 0.031 | 0 | 0 | 8 | 0.040 | 2.5 | 0.027 | 5 | 0.036 | 0 | 0 | 2.5 | 0.028 | 3 | 0.017 | 0 | 0 | 0 | 0 |
| *Ruspolia nitidula* (Scopoli, 1786) | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 2 | 0.059 | 0 | 0 |
| *Stenobothrus fischeri* (Eversmann, 1848) | 1.5 | 0.014 | 4.5 | 0.023 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Stenobothrus lineatus* (Panzer, 1796) | 11.25 | 0.078 | 5.25 | 0.017 | 19 | 0.102 | 11 | 0.114 | 0 | 0 | 5 | 0.054 | 9 | 0.109 | 9.5 | 0.062 | 0 | 0 | 1 | 0.010 |
| *Stenobothrus nigromaculatus* (Herrich-Schäffer, 1840) | 1.75 | 0.011 | 0 | 0 | 1 | 0.004 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Stenobothrus stigmaticus* (Rambur, 1838) | 0.25 | 0.001 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| *Tettigonia viridissima* Linnaeus, 1758 | 0 | 0 | 0 | 0 | 1 | 0.007 | 1 | 0.010 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0.029 | 0 | 0 |