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Effects of local climate, landscape structure and habitat quality on leafhopper assemblages of acidic grasslands

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A B S T R A C T

Grassland biodiversity is severely threatened by recent land-use change. Agricultural intensification on the one hand, and cessation of traditional land use on the other, have caused habitat loss, fragmentation and often a deterioration in habitat quality of the remaining habitat fragments. However, knowledge about the different environmental effects on species richness is still limited, in particular for under-sampled groups like leafhoppers (Auchenorrhyncha). Our study therefore aims to analyse the impact of local climate, landscape structure and habitat quality on leafhopper assemblages.

Several environmental factors were assessed and species richness of leafhoppers was sampled on 30 acidic grassland patches in Central Germany. We used generalised linear models (GLM) to determine the variables that influence species richness.

Both landscape structure and habitat quality had a strong influence on the number of leafhopper species. At the landscape scale, a high diversity of open land cover types positively affected species richness. Furthermore, species richness increased with decreasing cover of arable land in the surroundings of a habitat fragment. The best predictor at the habitat scale was the structural diversity, which had a positive impact on the numbers of leafhoppers. Local climatic conditions and patch area played a minor role and had an effect only on threatened species.

We recommend establishing a great variety of different structural types within a patch in order to promote species-rich leafhopper assemblages. In addition, conservationists should focus their efforts on the maintenance of different types of grasslands in the surroundings of habitat fragments.

1. Introduction

The recent decline of biodiversity is of global concern and primarily driven by land-use change (Sala et al., 2000; Stoate et al., 2009; Foley et al., 2005). Intensification of agricultural production has dramatically changed landscape composition, leading to homogenisation, loss and fragmentation of natural and semi-natural habitats (Benton et al., 2003; Fahrig, 2003; Marini et al., 2012). Additionally, habitat fragments have frequently suffered from degradation, i.e., from deterioration in habitat quality (Fischer and Lindenmayer, 2007). On the one hand, this can be caused by the intensification of land use, and on the other hand, by the abandonment of traditional land use (MacDonald et al., 2000; Duprè et al., 2010).

Particularly affected by land-use change are grasslands (Bakker and Berendse, 1999; Hodgson et al., 2005; Krause et al., 2011). Over the last few decades, the area and biodiversity of European grasslands have decreased considerably (Stoate et al., 2009). The majority of former nutrient-poor grasslands have been fertilised or transformed into arable land, forest or settlements (Hodgson et al., 2005; Walz, 2008; Krause et al., 2011). However, semi-natural grasslands are very rich in plant and animal species and, thus, are of vital importance for biodiversity conservation in Europe (Baur et al., 2006; Veen et al., 2009; Wilson et al., 2012).

In order to develop suitable conservation strategies, it is essential to gain detailed knowledge of the importance of landscape composition, habitat fragmentation and habitat quality on plant and animal communities. When analysing the effects of habitat fragmentation on species richness, the landscape is often divided into habitat fragments and the landscape matrix (Krämer et al., 2012). However, the composition of the matrix should also be considered because different elements may act either as a barrier or corridor for dispersal (Krämer et al., 2012; Poniatowski et al., 2016). Moreover, habitat quality is rarely taken into account in fragmentation studies or studies fail to detect an influence (Mortelliti et al., 2010; Krämer et al., 2012). According to Mortelliti

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Table 1
Comparison (mean values ± SE) of the six studied structural types. The measurement of structural parameters was conducted on every botanical plot before plants were recorded. The cover of different layers and vegetation density were estimated in steps of 5%. If values were above 95% or below 5%, steps of 2.5% were applied (cf. Helbing et al., 2014).

Sampled parameter	Structural type						*P*
	STONE (N = 17)	BARE (N = 27)	SHORT (N = 30)	MEDIUM (N = 24)	HIGH (N = 24)	SHRUB (N = 18)	
Cover (%)							
Total vegetation	56.2 ± 3.4^{a}	84.6 ± 2.2^{a}	97.5 ± 0.7^{b}	99.8 ± 0.1^{b}	100.0 ± 0.0^{b}	99.4 ± 0.6^{b}	***
Shrub layer	0.4 ± 0.3^{a}	0.5 ± 0.4^{a}	0.6 ± 0.3^{a}	1.1 ± 0.9^{a}	1.9 ± 1.2^{a}	83.1 ± 2.3^{b}	***
Field layer	46.5 ± 3.8^{a}	80.6 ± 2.4^{a}	96.5 ± 0.9^{b}	99.8 ± 0.1^{b}	99.6 ± 0.4^{b}	73.9 ± 4.1^{a}	***
Cryptogam layer	$14.0 \pm 2.5^{a,b}$	8.8 ± 1.2^{a}	44.4 ± 6.0^{b}	$33.3 \pm 6.3^{a,b}$	12.9 ± 3.1^{a}	$31.0 \pm 7.9^{a,b}$	***
Litter layer	4.0 ± 1.7^{a}	$8.6 \pm 0.9^{a,b}$	$17.1 \pm 3.0^{b,c}$	28.3 ± 4.6^{c}	66.9 ± 4.2^{d}	$38.6 \pm 4.9^{c,d}$	***
Bare ground	$7.2 \pm 1.5^{a,b}$	14.6 ± 2.0^{a}	$2.3 \pm 0.6^{b,c}$	0.2 ± 0.1^{c}	0.0 ± 0.0^{c}	0.6 ± 0.6^{c}	***
Stony surface	37.9 ± 3.6^{a}	0.3 ± 0.2^{b}	0.0 ± 0.0^{b}	0.0 ± 0.0^{b}	0.0 ± 0.0^{b}	0.0 ± 0.0^{b}	***
Vegetation height (cm)	7.1 ± 0.6^{a}	$13.1 \pm 0.5^{a,b}$	$22.0 \pm 0.7^{b,c}$	$38.0 \pm 1.6^{c,d}$	$50.9 \pm 2.8^{d,e}$	95.8 ± 7.0^{e}	***
Horizontal vegetation density (%)	9.8 ± 1.3^{a}	22.0 ± 1.4^{a}	48.4 ± 1.8^{b}	$60.9 \pm 2.2^{b,c}$	78.4 ± 3.0^{c}	74.5 ± 3.2^{c}	***
Plant species number	21.4 ± 1.4^{a}	22.2 ± 0.9^{a}	25.4 ± 0.9^{a}	24.6 ± 1.0^{a}	22.8 ± 1.1^{a}	15.1 ± 1.2^{b}	***

Differences between structural types were tested using a Kruskal–Wallis H test (ANOVA for 'Plant species number'). Different letters indicate significant differences (Holm–Sidak test for 'Plant species number', otherwise Dunn's test, $P < 0.05$).
*** $P < 0.001$.

et al. (2010), habitat quality should be described not only by structural parameters, such as vegetation height, but also by resource abundances in order to detect the most appropriate parameters for certain taxa. However, besides landscape and habitat quality parameters, the composition of animal communities within a certain area may also be influenced by the local climate (Nieto-Sánchez et al., 2015).

Leafhoppers are very rich in species and often have specific life strategies (Nickel, 2003). Moreover, they respond rapidly to environmental changes (Biedermann et al., 2005). The plant sucking insects constitute a major component of the phytophagous fauna as they are abundant consumers, prey for predators such as birds or invertebrates and hosts for parasitoids (Biedermann et al., 2005). In grassland habitats, densities may exceed 1000 individuals/m^2 (Biedermann et al., 2005). Thus, leafhoppers are well suited as indicators of habitat change or as model organisms for the study of effects at the landscape and habitat scale (Biedermann et al., 2005). In this study, the term leafhopper is used for all Auchenorrhyncha, including planthoppers, froghoppers and treehoppers.

Grasslands are important habitats for leafhoppers. A total of 120 species are known to prefer managed grasslands of Central Europe (Nickel, 2003; Nickel and Achtziger, 2005). Hitherto, there have been only a few studies, which have analysed the impact of climate (Masters et al., 1998), landscape complexity together with vegetation structure (Kőrösi et al., 2012; Zulka et al., 2014) or habitat connectivity together with fragment size (Rösch et al., 2013) on leafhopper assemblages. However, a study, which considers all of these parameters and tests for their effects on grassland leafhoppers, has not been previously conducted. In this study, we analysed the effects of local climate, landscape structure and habitat quality on leafhopper assemblages of 30 Central European acidic grassland patches. In particular, we have addressed the following hypotheses:

(i) Plant species richness and structural diversity on a habitat fragment positively influence leafhopper species richness.
(ii) Smaller and more isolated patches have a lower species richness.
(iii) The landscape composition in the surroundings of a habitat fragment mainly affects habitat generalists, because many species of this group are able to use a matrix dominated by open land cover types as stepping stone or even as permanent habitat.
(iv) Local climate affects thermophilous leafhopper species rather than eurythermous habitat generalists.

2. Materials and methods

2.1. Study area

The study area, Medebacher Bucht, is located in Central Germany along the border between the federal states of North Rhine-Westphalia and Hesse (51°11′ N/8°41′ E). It is a mountain basin 171 km^2 in size, with an elevation reaching from 300 m a.s.l. in the east to 680 m a.s.l. at its western edge. The Medebacher Bucht is situated on the leeward side of the Rothaar Mountains. Mean annual temperature decreases with increasing elevation from 7.6 to 6.4 °C, whilst the mean annual precipitation increases from 700 to 1100 mm (MURL NRW, 1989).

The Medebacher Bucht is an old cultural landscape, characterised by nutrient-poor and permeable acidic soils, small fields, low agricultural yields and a high proportion of species-rich semi-natural habitat fragments (e.g., mesic grasslands, heathlands and acidic grasslands) (Schmitt and Fartmann, 2006; LWL and LVR, 2008). Grassland and arable land are the dominant land cover types, accounting for 55–60% of the total area. Forests cover 30–35% of the landscape (Hölker, 2002). The heterogenous landscapes harbour a large number of endangered plant and animal species (e.g., plants: Schmitt and Fartmann, 2006; birds: Hölker, 2002; Orthoptera: Behrens and Fartmann, 2004). Three quarters of the study area are strictly protected sites designated under the EU Birds Directive or Habitats Directive (BfN, 2016).

2.2. Study patches

Semi-dry acidic grasslands are classified as vulnerable in Germany (Riecken et al., 2006) and host many stenotopic leafhopper species (Nickel et al., 2002). Therefore, they are a very suitable model systems for analysing the effects of environmental change on leafhopper assemblages. Sampling was conducted on all available patches of acidic grasslands (N = 30). Patch area varied between 0.1 and 6.4 ha. Patches were considered discrete when the distance to the nearest neighbouring patch exceeded 50 m (cf. Krämer et al., 2012). The majority of patches were grazed by cattle (43%) or sheep (30%). Only three patches (10%) were mown once a year, whereas five (17%) did not have any land use. Since type of land use had no effect on leafhopper species richness (Kruskal–Wallis H test, $P > 0.05$), it was not considered in further analyses.

Based on structural parameters (Table 1), we subdivided the patches into a maximum of six different types of vegetation structure according to Behrens and Fartmann (2004) and Poniatowski and Fartmann (2008). The structural types were characterised by increasing