

RESEARCH ARTICLE

Effects of Montane Heathland Restoration on Leafhopper Assemblages (Insecta: Auchenorrhyncha)

Fabian Borchard^{1,2} and Thomas Fartmann^{1,3}

Abstract

At the beginning of the 20th century, many montane heathlands were abandoned and became subject to natural succession or afforestation by humans. Thus, the formerly large montane heathlands slowly degraded into small and isolated patches. In this study, we evaluate the influence of restoration measures on leafhopper (Auchenorrhyncha) assemblages of montane heathland ecosystems in Central Europe. Our analyses comprised three different site types that were adjacent to each other: (1) montane heathlands, (2) restoration sites, and (3) control sites. Leafhoppers showed a clear response to montane heathland restoration. Thus, after 4–5 years since implementation of restoration measurements restoration sites were characterized by the highest species richness. However, detailed analyses of leafhopper diversity, species composition, and environmental parameters on the three site types revealed that

restoration sites were rather similar to control sites and significantly differing from montane heathlands. We conclude that leafhoppers are excellent bioindicators for restoration measurements because they reflected environmental differences between the three site types. Restoration measurements might only be a useful instrument to promote typical montane heathland leafhopper communities in the long run. Colonization by leafhoppers is, however, dependent on many different factors such as leafhopper mobility, vegetation structure, microclimate, and the establishment of ericaceous dwarf shrubs. Practitioners should establish a management regime (grazing and sod-cutting) that creates a mosaic of different habitat structures and increases typical heathland vegetation, thus, favoring the colonization of typical heathland leafhoppers.

Key words: biodiversity, conservation management, habitat specialist, microclimate, succession, vegetation structure.

Introduction

During recent decades, there has been a substantial reduction in biodiversity worldwide (Pimm et al. 1995). The major drivers of this biodiversity loss are land-use changes (Chapin et al. 2000; Sala et al. 2000; Koh et al. 2009), especially agricultural intensification on the one hand and abandonment of unproductive sites on the other. This economically driven transformation has led to a decline of traditional farming practices and a cessation of extensive land use (MacDonald et al. 2000). However, low-intensity farming is acknowledged to play a major role in the conservation of plant (McIntyre et al. 2003) and animal diversity (Kruess & Tscharntke 2002; Fartmann et al. 2012).

At the beginning of the 20th century, the cessation of traditional management practices started to have severe effects on heathland ecosystems (Symes & Day 2003). While lowland

heathlands were often converted into arable land, montane heathlands were abandoned and became subject to natural succession or to afforestation by humans (cf. Thompson et al. 1995). Thus, the formerly large montane heathlands in north-western and parts of Central Europe slowly degraded into small and isolated patches. However, some of the remaining montane heathlands are still biodiversity hotspots, particularly because of the occurrence of arctic-alpine and boreal-montane species (Thompson et al. 1995).

While numerous scientific studies have assessed the success of lowland heathland management and restoration (Gimingham 1992; Pywell et al. 1995; Keienburg & Prüter 2004), montane heathland ecosystems have been widely neglected (cf. Borchard et al. 2013). This is especially true for arthropods (Usher & Thompson 1993), even though the arthropod fauna of montane heathlands is very diverse (Usher 1992), particularly in comparison with the species-poor flora (Usher 1992; Littlewood et al. 2006).

The montane heathlands of our study area (Rothaargebirge, NW Germany) belong to one of the last regions within Central Europe retaining intact montane heathland ecosystems (Geringhoff & Daniëls 2003). In order to protect these unique semi-natural landscapes and their specialized wildlife, the

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remaining montane heathlands are nationally protected nature reserves (MUNLV 2012) and part of the European Natura 2000 network (European Community 1992). Furthermore, the European Union supported the restoration and enlargement of montane heathlands through the EU Life program. The aim of this study was to use leafhoppers as ecological indicators to evaluate the success of the conducted restoration measurements in our study area. Hereafter, the term leafhopper comprises all Auchenorrhyncha, including planthoppers.

Leafhoppers belong to a group of insects that are highly appropriate for reflecting environmental changes on different scales. Thus, they have been used as bioindicators for climate change (Masters et al. 1998; Whittaker & Tribe 1998), habitat fragmentation (Biedermann 2002), and management of different grassland ecosystems (Morris 1981; Nickel & Hildebrandt 2003; Hollier et al. 2005). Only recently have different studies successfully used leafhoppers as indicators for restoration and management of upland moorland in Great Britain (Littlewood et al. 2006, 2009, 2012). According to Nickel and Hildebrandt (2003) and Biedermann et al. (2005), the main reasons for their suitability as bioindicators are their (1) high species numbers and densities, (2) functional importance as consumers, prey, and hosts for parasitoids, (3) specific life strategies, comprising monophagous specialists, and polyphagous generalists, (4) immediate response to environmental changes, and (5) the possibility for a standardized sampling with high spatial resolution.

In our study, we compared three different site types that were adjacent to each other: (1) montane heathlands (MONHEATH), (2) restoration sites (RESSITE), and (3) control sites (CONTROL). We used leafhoppers as ecological indicators to answer the following questions with regard to restoration measurements on montane heathland ecosystems:

- i. Do leafhoppers of montane heathlands, restoration, and control sites differ in their diversity, density, and species composition?
- ii. What are the main environmental factors that determine the composition of leafhopper assemblages and how did the restoration measures affect leafhoppers?
- iii. Have typical montane heathland leafhoppers been able to colonize adjacent restoration sites and did they benefit from conducted measurements?

Methods

Study Area and Study Sites

The study was conducted in the “Rothaargebirge,” a low mountain range at the border of the German Federal States of North Rhine-Westphalia and Hesse (51°28'N, 7°33'E). The study area stretches 40 km from north to south and 30 km from east to west. It is characterized by a montane climate with an average annual temperature of 5°C, a mean annual precipitation of 1,450 mm and a prolonged snow cover of 100 d/a (Deutscher Wetterdienst, personal communication 2011). The montane heathlands in this region (Fig. 1a) are considered representative of Central Europe because they are valuable biodiversity hotspots for

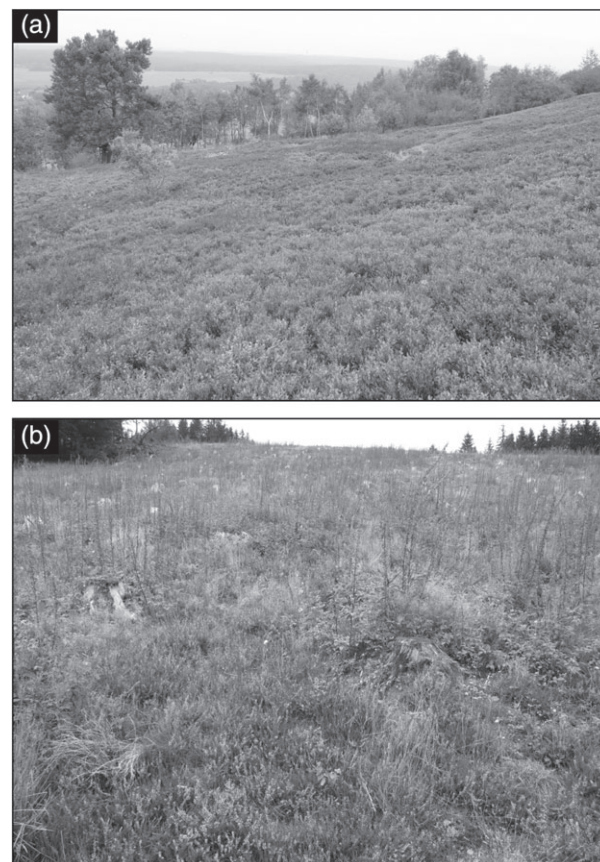


Figure 1. Typical aspect of old montane heathlands (a) and newly established restoration sites with a high cover of open soil and diverse vegetation (b).

many arctic-alpine and boreal-montane plant and animal species (Geringhoff & Daniëls 2003). All restoration sites were part of the EU LIFE project “Medebacher Bucht – A building block for Natura 2000,” which aimed to restore montane heathland habitats on former spruce (*Picea abies*) plantations (Fig. 1b). The spruce forests were cut and the sites cleared of remaining branches. The montane heathlands as well as the restoration sites were grazed by sheep or goats.

The study sites were located in the highest parts of the Rothaargebirge with altitudes ranging from 540 to 831 m a.s.l. (mean \pm SE = 705 \pm 39.8). In total, we established 19 permanent plots that were at least 450 m apart and each had an area size of 500 m² (20 \times 25 m). The three following site types were analyzed: (1) montane heathlands (MONHEATH) dominated by *Calluna vulgaris*, *Vaccinium myrtillus*, and *Vaccinium vitis-idaea* ($n = 7$). (2) Restoration sites (RESSITE) where chopper material or the hydroseeding procedure was applied ($n = 7$), and (3) clear-cuts of spruce forests as unprocessed and ungrazed control sites (CONTROL) ($n = 5$).

Restoration Methods

The restoration measures were carried out in 2008 and 2009 in the direct vicinity of existing montane heathlands. The seed

material was harvested on the largest heathland in our study area ("Neuer Hagen," 73.9 ha).

The hydroseeding procedure is particularly used for the revegetation of man-made steep slopes such as construction sites (Bochet & García-Fayos 2004; Matesanz et al. 2006). The hydroseeding material was composed of harvested seed material (threshed montane heathland species) from the donor site, water, and erosion control agents. In order to evenly spread the seed material, the agents were mixed to a homogenous suspension in an all-terrain hydroseeder. The mixture was evenly sprayed on the restoration sites.

The chopper material was harvested by a specifically designed machine that removes the complete organic layer down to the mineral soil (cf. Keienburg & Prüter 2004). The material was collected in a tractor-drawn trailer and transferred to the restoration sites. The material was spread over an area of land the same extent as that from which it was harvested (application rate of 1:1) using a manure wagon with a scatter roller.

Sampling Design

We recorded leafhoppers during two sampling periods in August 2011 and June 2012 on all permanent plots. In order to collect representative samples, we walked over the whole plot (500 m²) in loops, performing 100 strokes with a sweep net of 30 cm diameter. Additionally, we analyzed pitfall trap catches from mid-August until mid-October 2011 and from mid-May until the beginning of July 2012. We randomly installed three traps per plot (at least 10 m apart) and used a roof to prevent overflow (10 cm in diameter, 2.5 cm above each trap). The traps were 7.5 cm deep, 9 cm in diameter and half filled with Renner solution (40% ethanol, 30% water, 20% glycerine, 10% acetic acid). The traps were emptied every 3 weeks during the sampling period.

According to Nickel (2003), pitfall traps are a good complementation of sweep netting as pitfall traps primarily catch epigeic leafhoppers whereas sweep netting catches hypogeous leafhoppers of the middle to the upper vegetation layer (cf. Stewart 2002). Both methods combined usually sample the complete range of ground-dwelling and herb layer species (Nickel 2003). The collected sweep net catches were transferred to plastic bags and frozen. We determined all adult leafhoppers to species level (or genus level if species could not be separated, such as *Psammotettix* and *Muellerianella* females) using Biedermann and Niedringhaus (2004) and Kunz et al. (2011). All sweep-net sampling was conducted under warm and sunny weather conditions between 10:00 and 18:00 h.

We recorded the features of vegetation structure on three randomly established subplots (replicates) within each of the permanent plots. The relevés were carried out in July 2011 and September 2012. Each subplot had a size of 16 m² (4 × 4 m). We estimated vegetation cover for the herb, shrub, dwarf shrub, and tree layer in 5% steps. In cases where cover was above 95% or below 5%, 2.5% steps were used, according to Behrens and Fartmann (2004). Furthermore, we recorded the percentage of bare

soil, litter, and dead wood. For statistical evaluation we computed overall means of the subplot data per plot, incorporating both sampling periods (2011 and 2012).

From August 2011 to the end of September 2012, we recorded the microclimate on each of the permanent plots. We installed a Hygrochron Temperature/Humidity Logger (iButton, Maxim/Dallas, TX, U.S.A.) 10 cm above ground and measured air temperature and humidity every hour. In order to protect the Hygrochron sensor from direct sunlight and precipitation, it was placed in a self-constructed radiation shield. The radiation shield consisted of a plastic pipe with a matching lid. In order to avoid heat accumulation, we drilled holes in the pipe to allow for air circulation and isolated the lid with the help of Styrofoam (1 cm thick) to prevent overheating on sunny days. The iButton logger (Maxim/Dallas, DS9093F) was attached to the lid with a screw, a key ring, and a plastic snap-in fob. The whole radiation shield was mounted to an iron pole that was fixed to the ground. The construction proved to be reliable in terms of measurement accuracy and robustness with respect to damage from animals (sheep and goats). For statistical evaluation, we computed mean temperature values for each permanent plot, considering the whole sampling period.

Additionally, we measured soil moisture with the Theta Probe ML2. Soil samples were collected during dry weather in August 2011 and September 2012. In order to prevent the measurements from being biased by small-scale variations in soil properties, they were repeated at three different locations per subplot. The sample depth was 5 cm. For the final analysis, we calculated mean soil moisture values per plot, incorporating the results from the two sampling periods.

Data Analysis

Leafhopper species were classified according to their habitat preferences into two categories: (i) heathland species and (ii) non-heathland species. The assessment was based on Biedermann and Niedringhaus (2004). Thereby, all leafhoppers that are regularly found on heathland habitats were categorized as heathland species (Table 1).

A second classification was based on the diet niche of leafhoppers and follows data given in Nickel and Remane (2002). Many leafhoppers have very specific food requirements and only feed on one plant species or one plant genus (=monophagous species). These species were considered to be diet specialists. All other leafhopper species were considered to be diet generalists (Table 1).

The leafhopper data incorporates both sweepnet sampling periods (August 2011 and June 2012) and pitfall trap data (mid-August until mid-October 2011 and from mid-May until the beginning of July 2012), unless otherwise stated (e.g. Fig. 2b).

As both restoration procedures (hydroseeding, application of chopper material) did not differ in leafhopper species richness, densities, and assemblage composition nor in sampled environmental parameters (cover of vegetation, cover of bare soil, cover of dwarf shrubs, soil moisture, temperature) we analyzed data of both techniques together.

Table 1. Species list of leafhoppers, their habitat specificity (heathland species, non-heathland species), diet niche, frequency on study plots (%) and number of individuals. For further information about the classification of leafhoppers see section “data analysis.”

No	Scientific Name	Heathland Species	Diet Specialist	Frequency (%) on Study Plots (n = 19)	No. Individuals
1	<i>Acanthodelphax spinosa</i>		✓	15.8	3
2	<i>Anaceratagallia ribauti</i>		✓	5.3	1
3	<i>Anaceratagallia venosa</i>			10.5	2
4	<i>Anoscopus albifrons</i>	✓		21.1	6
5	<i>Anoscopus flavostriatus</i>			5.3	1
6	<i>Aphrodes bicincta</i>			15.8	3
7	<i>Aphrodes diminuta</i>			5.3	1
8	<i>Aphrodes makarovi</i>			21.1	10
9	<i>Aphrophora alni</i>			5.3	1
10	<i>Arocephalus longiceps</i>	✓		15.8	7
11	<i>Arthaldeus pascuellus</i>			10.5	3
12	<i>Balclutha punctata</i>			31.6	16
13	<i>Cicadula persimilis</i>		✓	5.3	1
14	<i>Conomelus anceps</i>		✓	5.3	1
15	<i>Deltocephalus pulicaris</i>			31.6	101
16	<i>Doratura stylata</i>			36.8	20
17	<i>Elymana sulphurella</i>			47.4	34
18	<i>Errhomenus brachypterus</i>			10.5	4
19	<i>Eupelix cuspidata</i>		✓	42.1	10
20	<i>Eupterix aurata</i>			5.3	1
21	<i>Eupterix notata</i>			5.3	2
22	<i>Eupterix urticae</i>		✓	5.3	1
23	<i>Euscelis incisus</i>			21.1	5
24	<i>Euscelis ohausi</i>			21.1	12
25	<i>Evacanthus interruptus</i>			5.3	1
26	<i>Forcipata forcipata</i>			5.3	1
27	<i>Graphocraerus ventralis</i>			10.5	2
28	<i>Hyledelphax elegantula</i>			15.8	3
29	<i>Idiodonus cruentatus</i>	✓		5.3	3
30	<i>Jassargus allobrogicus</i>	✓		36.8	61
31	<i>Javesella dubia</i>			10.5	2
32	<i>Macropsis fuscata</i>		✓	5.3	2
33	<i>Macropsis infuscata</i>		✓	5.3	1
34	<i>Macrostes sexnotatus</i>			5.3	1
35	<i>Macustus grisescens</i>			5.3	1
36	<i>Megophthalmus scaninus</i>			5.3	1
37	<i>Muellerianella brevipennis</i>		✓	15.8	5
38	<i>Muellerianella fairmai</i>		✓	5.3	2
39	<i>Neophilaenus campestris</i>			10.5	11
40	<i>Neophilaenus lineatus</i>			26.3	12
41	<i>Ophiola decumana</i>			21.1	6
42	<i>Ophiola russeola</i>	✓		5.3	3
43	<i>Philaenus spumarius</i>			21.1	5
44	<i>Planaphrodes bifasciata</i>			42.1	24
45	<i>Planaphrodes nigrita</i>			10.5	2
46	<i>Psammotettix alienus</i>			5.3	1
47	<i>Psammotettix confinis</i>			63.2	145
48	<i>Psammotettix helvolus</i>			78.9	217
49	<i>Psammotettix nodosus</i>			52.6	56
50	<i>Rhopalopyx adumbrata</i>	✓	✓	5.3	9
51	<i>Rhopalopyx preyssleri</i>		✓	10.5	3
52	<i>Streptanus marginatus</i>	✓		15.8	19
53	<i>Streptanus sordidus</i>			5.3	1
54	<i>Tachycixius pilosus</i>			5.3	1
55	<i>Ulopa reticulata</i>	✓	✓	10.5	5
56	<i>Verdanus abdominalis</i>			63.2	52
57	<i>Xanthodelphax straminea</i>		✓	10.5	7

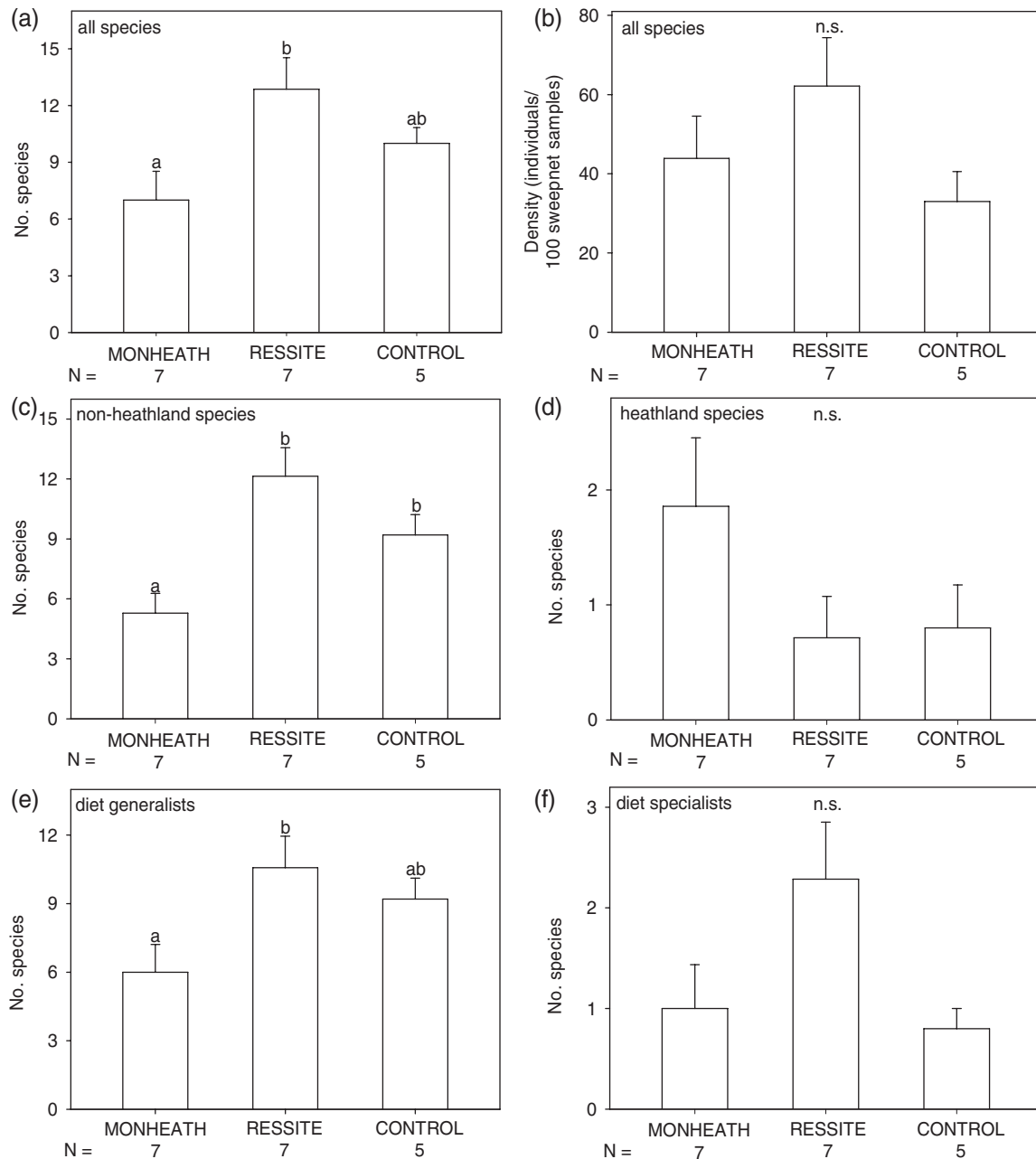


Figure 2. Mean values standard error (\pm SE) of species number (a) and density (b) (individuals/100 sweep net samples) for all leafhoppers as well as mean values (\pm SE) of habitat generalists (c) and specialists (d), diet generalists (e), and diet specialists (f) for montane heath (MONHEATH), restoration sites (RESSITE), and control sites (CONTROL). Statistics: (a) ANOVA, $F = 4.990$, $df = 2$, $p < 0.05$; (b) ANOVA, $F = 1.728$, $df = 2$, $p = 0.21$; (c) ANOVA, $F = 9.919$, $df = 2$, $p < 0.01$; (d) Kruskal–Wallis ANOVA on ranks, $H = 1.850$, $df = 2$, $p = 0.39$; (e) ANOVA, $F = 3.847$, $df = 2$, $p < 0.05$; (f) ANOVA, $F = 3.024$, $df = 2$, $p = 0.08$. Differences between groups were tested using the Holm–Sidak method (a–f) and Dunn’s test as a post hoc test (d), respectively. Different letters indicate significant differences between site types ($p < 0.05$); n.s. = not significant.

Prior to statistical evaluation, the data of all subplots per plot were pooled. If data were normally distributed with equal variances, differences among site types (MONHEATH, RESSITE, CONTROL) were analyzed using one-way analysis of variance (ANOVA) followed by Holm–Sidak tests for pair-wise comparisons. Otherwise, we performed a Kruskal–Wallis ANOVA on Ranks and Dunn’s test as a post hoc test.

Prior to Non-metric Multidimensional Scaling (NMDS) ordination and Generalized Linear Models (GLM) (see below), intercorrelations of all predictor variables were examined by applying a Spearman correlation matrix (correlation coefficient r_s) that included all metric predictor variables. In cases of high intercorrelation among variables ($|r_s| > 0.7$), one of them was excluded from the analyses. For two pairs of

intercorrelated variables (vegetation cover/cover of dwarf shrubs, bare soil/number of plant species), we summarized the factors to a new variable by conducting Principal Component Analysis (PCA). The new variable vegetation/dwarf shrub cover had an eigenvalue of 1.766 explaining 88% of the variance; it was positively correlated with vegetation cover ($r_s = 0.97$, $p < 0.001$) and cover of dwarf shrubs ($r_s = 0.90$, $p < 0.001$). The second new variable, bare soil/number of plant species, had an eigenvalue of 1.640 explaining 82% of the variance and was positively correlated with the cover of bare soil ($r_s = 0.89$, $p < 0.001$) and number of plant species ($r_s = 0.96$, $p < 0.001$). The variables entered in our NMDS and GLM analyses were soil moisture, temperature, and the two newly created variables, vegetation/dwarf shrub cover and bare soil/number of plant species.

Leafhopper assemblage structure and environmental parameters were analyzed using NMDS (VEGAN, Oksanen et al. 2008; MASS, Venables & Ripley 2008; software package R 2.15.3). In order to enhance accuracy of the NMDS, we omitted leafhopper species that occurred with less than two individuals in our data set. We used the Bray–Curtis distance as distance measure with a maximum number of 100 random starts in the search for a stable solution. The environmental variables were fitted onto the ordination afterwards and only significant variables ($p < 0.05$) are shown. Mantel test based on Spearman's rank correlation and 999 permutations were used to test for correlations between leafhopper species and environmental parameters.

GLM were used for the analysis of the relationship between number of leafhopper species (response variable) and environmental parameters (predictor variables). As our response variable showed overdispersion, we corrected the standard errors using quasi-Poisson GLMs. To assess the significance of the environmental parameters and to avoid an over-fitting, each variable was entered separately into a univariate regression model. The analyses were performed using SPSS statistics 20, R-2.15.3 (R Development Core Team, 2013) and SigmaPlot 11.0.

Results

Community Composition

A total of 908 adult leafhopper individuals, belonging to 57 species, were collected (Table 1). The five most frequent species were *Psammotettix helvolus* ($n = 217$, 24%), *Psammotettix confinis* ($n = 145$, 16%), *Deltocephalus pulicaris* ($n = 101$, 11%), *Jassargus allobrogicus* ($n = 61$, 7%), and *Psammotettix nodosus* ($n = 56$, 6%). Together, they accounted for 64% ($n = 580$) of all individuals.

Overall species richness, leafhopper density, number of non-heathland species, diet generalist, and diet specialist species was highest on RESSITE (Fig. 2a–c, e, f). However, for leafhopper density and number of diet specialist species the differences among the three site types were not significant. The number of all species, non-heathland species, and diet generalist species was lowest on MONHEATH and had an intermediate position on CONTROL.

The number of heathland species was the only parameter that peaked in MONHEATH; however, the values did not significantly differ from the two other site types (Fig. 2d).

Leafhopper Assemblages and Response to Environmental Parameters

NMDS ordination showed that the leafhopper species assemblages of MONHEATH on the one hand and RESSITE and CONTROL on the other, were distinctly different from each other (Fig. 3). Leafhopper species data were significantly correlated with environmental parameters (Mantel test, $r = 0.23$, $p < 0.05$). All environmental variables significantly contributed to the ordination model. In particular the cover of vegetation/dwarf shrubs (VDC) and cover of bare soil/number of plant species (BS/NoP) showed a highly significant contribution to the ordination ($p < 0.01$), whereas the temperature was of less importance ($p < 0.05$).

The variation in leafhopper species composition was determined by two environmental gradients. The first gradient represents a bare ground gradient and differentiated MONHEATH from RESSITE and CONTROL: The cover of vegetation and dwarf shrubs was negatively correlated with this axis, while the cover of bare soil and the number of plant species showed a highly positive correlation. All heathland specialists, except *Arocephalus longiceps* and *Anoscopus albifrons*, were more closely associated with MONHEATH. In contrast, most generalist species showed a clear preference for RESSITE and CONTROL. The second gradient represents a microclimate gradient that separated warmer from cooler site types. Particularly RESSITE showed higher temperatures and were thus separated from MONHEATH and CONTROL.

The results of the GLM analyses confirmed the high relevance of bare soil/plant species diversity as a driver of leafhopper species richness (Table 2). It was the only predictor of leafhopper diversity and had a high explanatory power (McFadden Pseudo $R^2 = 0.47$).

Discussion

Leafhoppers showed a clear response to montane heathland restoration. Thus, after 4–5 years since implementation of restoration measurements, RESSITE was characterized by the highest species richness. However, detailed analyses of leafhopper diversity, species composition, and environmental parameters on the three site types revealed that RESSITE was rather similar to CONTROL and significantly differing from MONHEATH. The main driver of leafhopper species richness and community composition on our study sites was the cover of bare soil/number of plant species.

Montane heathlands are known to have a high arthropod diversity (Usher 1992). However, concerning leafhoppers our findings show that old montane heathlands contain rather species-poor assemblages. Nevertheless, MONHEATH is relevant for leafhopper conservation as it harbors several specialist species that were restricted to this habitat in our study. Among

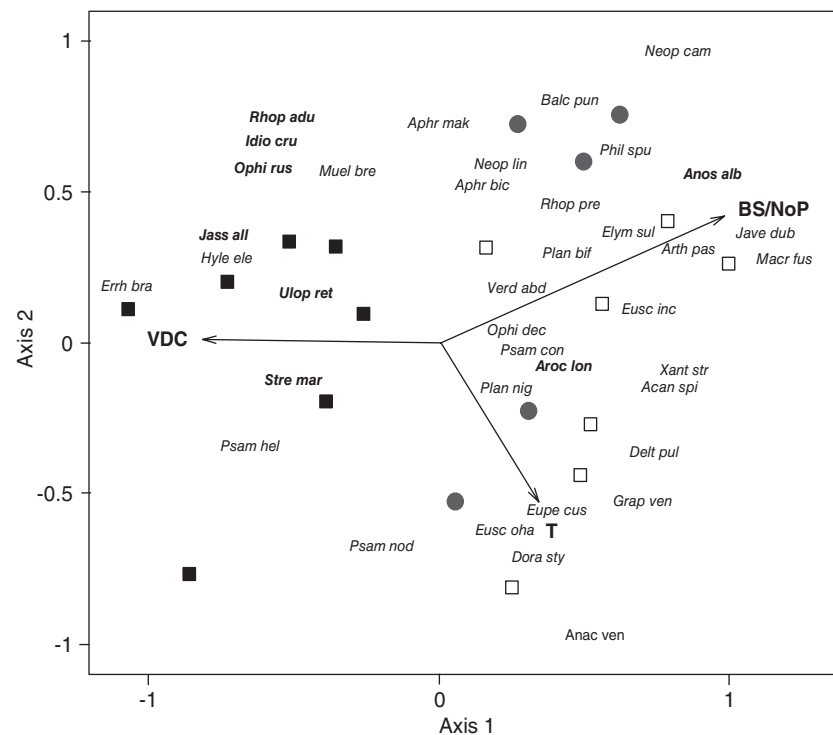


Figure 3. Non-metric Multidimensional Scaling (NMDS) (stress = 9.33, three dimensions) based on leafhopper species and environmental variables (for abbreviations of environmental variables see Table 2). Solid black quadrates = montane heathlands, open quadrates = restoration sites, and open circles = control sites. Leafhopper specialists are indicated in bold type. For classification of heathland species see section “data analysis”).

Table 2. Statistics of GLM: Relationship between number of leafhopper species (response variable) and four environmental variables (predictor variables).

Variable	Estimate	SE	Z	p	R ²
Soil moisture (SM)	−0.010	0.014	−0.723	n.s.	
Temperature (T)	0.014	0.129	−0.108	n.s.	
Vegetation/dwarf shrub cover (VDC)	−0.184	0.098	−1.888	n.s.	
Bare soil/number of plant species (BS/NoP)	0.305	0.076	3.996	< 0.001	0.47

To assess the significance of the environmental parameters, each variable was entered separately into a univariate regression model. n.s., not significant.

these specialists are two species (*Ophiola russeola* and *Ulopa reticulata*) that we exclusively found on montane heathlands generally having a high cover of the host plant *Calluna vulgaris* (cf. Borchard et al. 2013). We rarely found leafhopper habitat specialists on RESSITE or CONTROL where typical montane heathland plants such as *Calluna vulgaris*, *Vaccinium myrtillus*, and *Vaccinium vitis-idaea* had successfully established, however, their cover was clearly lower than on MONHEATH (Borchard et al. 2014). One possible explanation might be that the time period of 4–5 years since the restoration measurements were conducted was too short for the successful colonization of the adjacent RESSITE. Littlewood et al. (2009) showed that even distances of 50 m negatively affected colonization rates for *Ulopa reticulata*, while *Philaenus spumarius*, an eurytopic species (cf. Nickel et al. 2002) was far more mobile.

Another reason for the rarity of these heathland species on RESSITE and CONTROL might be structural and microclimatic differences compared to MONHEATH. Borchard et al.

(2013) showed that the structural differences between MONHEATH on the one hand and RESSITE/CONTROL on the other hand were considerably different. MONHEATH showed a significantly higher cover of total vegetation, dwarf shrubs, and mosses, but a significantly lower cover of herbs/grasses. Furthermore, the cover of bare soil was highest on RESSITE significantly differing from MONHEATH. These structural differences also had a clear effect on microclimatic conditions such as temperature and humidity on the different site types. MONHEATH were characterized by significantly lower temperatures than the other two treatments (RESSITE, CONTROL) and had a higher humidity. Accordingly, we found xero- and heliophilous species such as *Neophilaenus campestris* and *Rhopalopyx preysleri* (cf. Nickel et al. 2002) on RESSITE and CONTROL.

RESSITE and CONTROL had a significantly higher plant species richness (this study) and structural diversity (Borchard et al. 2013) compared to MONHEATH which attracted particularly many eurytopic leafhopper species. Typical pioneer plant

species such as *Cytisus scoparius*, *Epilobium angustifolium*, and *Rubus idaeus* established rapidly after clear felling (unpublished data) and provided an important food source for many generalist leafhoppers with a broad diet width, e.g. *Aphrodes makarovi*, *Elymana sulphurella*, *Philaenus spumarius*, and *Psammotettix confinis*. Furthermore, many species of Poaceae, which are by far the most-favored leafhopper food plant family (Nickel 2003) occurred on RESSITE and CONTROL (unpublished data). These results are in accordance with a study by Littlewood et al. (2006) in upland heath in northern England and Scotland. They detected a higher species richness of true bugs (Hemiptera) in grassland than in heathland samples and found a link between bug diversity and the number of grass species.

We conclude that only with a further establishment of target plant species (*Calluna vulgaris*, *Vaccinium myrtillus*, and *Vaccinium vitis-idaea*) and associated changes in vegetation structure and microclimate will RESSITE become a suitable habitat for typical heathland leafhopper species (cf. Morris 1990, 2000; Schaffers et al. 2008). This in turn indicates that the conducted restoration measurements were not able to establish environmental conditions typical for MONHEATH on RESSITE within the timescale studied here. Instead, RESSITE and CONTROL presented the earliest stages of plant succession and were dominated by many pioneer plant species. Hence, these site types did not differ much from each other concerning their leafhopper diversity, density, and species composition.

Implications for Practice

- The restoration of montane heathlands on former spruce forests, including the development of typical heathland vegetation and colonization by typical heathland leafhoppers, is a time demanding process. Therefore, conservation and proper management of long-existing montane heathlands should always be first choice.
- Leafhoppers were highly sensitive to environmental conditions that makes them to suitable indicator organisms reflecting current restoration status.
- Structural heterogeneity and plant species diversity is of high relevance for leafhopper species richness and abundance.
- Practitioners should establish a management regime (grazing and sod-cutting) that ideally creates a mosaic of different habitat structures. We advise not to manage the whole heathland at the same time, but to establish rotational management systems.

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LITERATURE CITED

- Behrens, M., and T. Fartmann. 2004. Die Heuschreckengemeinschaften isolierter Schieferkuppen der Medebacher Bucht (Südwestfalen/Nordhessen). *Tuexenia* **24**:303–327.
- Biedermann, R. 2002. Leafhoppers (Hemiptera, Auchenorrhyncha) in fragmented habitats. *Denisia* **4**:523–530.
- Biedermann, R., and R. Niedringhaus. 2004. Die Zikaden Deutschlands. Bestimmungstabellen für alle Arten, Scheeßel, WABV.
- Biedermann, R., R. Achtziger, H. Nickel, and A. J. A. Stewart. 2005. Conservation of grassland leafhoppers: a brief review. *Journal of Insect Conservation* **9**:229–243.
- Bochet, E., and P. García-Fayos. 2004. Factors controlling vegetation establishment and water erosion on motorway slopes in Valencia, Spain. *Restoration Ecology* **12**:166–174.
- Borchard, F., A. M. Schulte, and T. Fartmann. 2013. Rapid response of Orthoptera to restoration of montane heathland. *Biodiversity and Conservation* **22**:687–700.
- Borchard, F., A. M. Schulte, and T. Fartmann. 2014. Restitution montaner Heiden im Rothaargebirge – Evaluation der Restitutionsmaßnahmen montaner Heidebestände durch vegetations- und tierökologische Untersuchungen im Hochsauerland. *Natur in NRW* **1**:32–35.
- Chapin, F. S., E. S. Zavaleta, V. T. Eviner, R. L. Naylor, P. M. Vitousek, H. L. Reynolds, et al. 2000. Consequences of changing biodiversity. *Nature* **405**:234–242.
- European Community. 1992. The habitats directive 92/43/EEC. Brussels, European Community.
- Fartmann, T., B. Krämer, F. Stelzner, and D. Poniatowski. 2012. Orthoptera as ecological indicators for succession in steppe grassland. *Ecological Indicators* **20**:337–344.
- Geringhoff, H. J. T., and F. J. A. Daniëls. 2003. Zur Syntaxonomie des *Vaccinio-Callunetum* Büker 1942 unter besonderer Berücksichtigung der Bestände im Rothaargebirge. *Abhandlungen aus dem Westfälischen Museum für Naturkunde* **65**:1–80.
- Gimingham, G. H. 1992. The lowland heathland management handbook. English Nature Science 8, Peterborough.
- Hollier, J. A., N. Maczey, G. J. Masters, and S. R. Mortimer. 2005. Grassland leafhoppers (Hemiptera: Auchenorrhyncha) as indicators of habitat condition – a comparison of between-site and between-year differences in assemblage composition. *Journal of Insect Conservation* **9**:299–307.
- Keienburg, T., and J. Prüter. 2004. Conservation and management of Central European lowland heathlands. Case study: Lüneburger Heide nature reserve, North-West Germany. *Mitteilungen der Alfred Toepfer Akademie für Naturschutz* **15**:1–64.
- Koh, L. P., P. Levang, and J. Ghazoul. 2009. Designer landscapes for sustainable biofuels. *Trends in Ecology & Evolution* **24**:431–438.
- Kruess, A., and T. Tschardt. 2002. Contrasting responses of plant and insect diversity to variation in grazing intensity. *Biological Conservation* **106**:293–302.
- Kunz, G., H. Nickel, and R. Niedringhaus. 2011. Fotoatlas der Zikaden Deutschlands. Fründ, WABV.
- Littlewood, N. A., R. J. Pakeman, and S. J. Woodin. 2006. The response of plant and insect assemblages to the loss of *Calluna vulgaris* from upland vegetation. *Biological Conservation* **128**:335–345.
- Littlewood, N. A., R. J. Pakeman, and S. J. Woodin. 2009. Isolation of habitat patches limits colonisation by moorland Hemiptera. *Journal of Insect Conservation* **13**:29–36.

- Littlewood, N. A., R. J. Pakeman, and G. Pozsgai. 2012. Grazing impacts on Auchenorrhyncha diversity and abundance on a Scottish upland estate. *Insect Conservation and Diversity* **5**:67–74.
- MacDonald, D., J. R. Crabtree, G. Wiesinger, T. Dax, N. Stamou, P. Fleury, J. G. Lazpita, and A. Gibon. 2000. Agricultural abandonment in mountain areas of Europe: environmental consequences and policy response. *Journal of Environmental Management* **59**:47–69.
- Masters, G. J., V. K. Brown, I. P. Clarke, J. P. Whittaker, and J. A. Hollier. 1998. Direct and indirect effects of climate change on insect herbivores: Auchenorrhyncha (Homoptera). *Ecological Entomology* **23**:45–52.
- Matesanz, S., F. Valladares, D. Tena, M. Costa-Tenorio, and D. Bote. 2006. Early dynamics of plant communities on revegetated motorway slopes from southern Spain: is hydroseeding always needed? *Restoration Ecology* **14**:297–307.
- McIntyre, S., K. M. Heard, and T. G. Martin. 2003. The relative importance of cattle grazing in subtropical grasslands: does it reduce or enhance plant biodiversity? *Journal of Applied Ecology* **40**:445–457.
- Morris, M. G. 1981. Responses of grassland invertebrates to management by cutting. 4. Positive responses of Auchenorrhyncha. *Journal of Applied Ecology* **18**:763–771.
- Morris, M. G. 1990. The Hemiptera of two sown calcareous grasslands II. Differences between treatments. *Journal of Applied Ecology* **27**:379–393.
- Morris, M. G. 2000. The effects of structure and its dynamics on the ecology and conservation of arthropods in British grasslands. *Biological Conservation* **95**:129–142.
- MUNLV, Ministerium für Umwelt und Naturschutz, Landwirtschaft und Verbraucherschutz. 2012. Natura 2000 Gebiete in Nordrhein-Westfalen (available from <http://www.naturschutzinformationen-nrw.de/natura2000-meldedok/de/karten>).
- Nickel, H. 2003. The leafhoppers and planthoppers of Germany (Hemiptera, Auchenorrhyncha): patterns and strategies in a highly diverse group of phytophagous insects. Pensoft, Sofia-Moscow.
- Nickel, H., and J. Hildebrandt. 2003. Auchenorrhyncha communities as indicators of disturbance in grasslands (Insecta, Hemiptera) – a case study from the Elbe flood plains (northern Germany). *Agriculture, Ecosystems & Environment* **98**:183–199.
- Nickel, H., W. E. Holzinger, and E. Wachmann. 2002. Mitteleuropäische Lebensräume und ihre Zikadenfauna (Hemiptera: Auchenorrhyncha). *Denisia* **4**:279–328.
- Nickel, H., and R. Remane. 2002. Artenliste der Zikaden Deutschlands, mit Angaben zu Nährpflanzen, Nahrungsbreite, Lebenszyklen, Areal und Gefährdung (Hemiptera, Fulgoromorpha et Cicadomorpha). *Beiträge zur Zikadenkunde* **5**:27–64.
- Oksanen, J., R. Kindt, P. Legendre, B. O'Hara, G. L. Simpson, P. Solymos, M. H. Stevens, and H. Wagner. 2008. The vegan package version 2.0-6 (available from <http://cran.r-project.org/>, <http://vegan.r-forge.r-project.org/>).
- Pimm, S. L., G. J. Russel, J. L. Gittleman, and T. M. Brooks. 1995. The future of biodiversity. *Science* **269**:347–350.
- Pywell, R. F., N. R. Webb, and P. D. Putwain. 1995. A comparison of techniques for restoring heathland on abandoned farmland. *Journal of Applied Ecology* **32**:400–411.
- R Development Core Team. 2009. R: a language and environment for statistical computing (available from <http://www.R-project.org>).
- Sala, O. E., F. S. Chapin, J. J. Armesto, E. Berlow, J. Bloomfield, R. Dirzo, et al. 2000. Biodiversity – global biodiversity scenarios for the year 2100. *Science* **287**:1770–1774.
- Schaffers, A. P., I. P. Raemakers, K. V. Sykora, and C. J. F. terBtaak. 2008. Arthropod assemblages are best predicted by plant species composition. *Ecology* **89**:782–794.
- Stewart, A. J. A. 2002. Techniques for sampling Auchenorrhyncha in grasslands. *Denisia* **17**:491–512.
- Symes, N., and J. Day. 2003. A practical guide to the restoration and management of lowland heathland. RSPB, Sandy.
- Thompson, D. B. A., A. J. MacDonald, J. H. Marsden, and C. A. Galbraith. 1995. Upland heather moorland in Great Britain – a review of international importance, vegetation change and some objectives for nature conservation. *Biological Conservation* **71**:163–178.
- Usher, M. B. 1992. Management and diversity in *Calluna* heathland. *Biodiversity and Conservation* **1**:63–79.
- Usher, M. B., and D. B. A. Thompson. 1993. Variation in the upland heathlands of Great Britain: conservation importance. *Biological Conservation* **66**:69–81.
- Venables, W. N., and B. Ripley. 2008. The VR Package Version 7. 2-45 (available from <http://www.stats.ox.ac.uk/pub/MASS4/>).
- Whittaker, J. B., and N. P. Tribe. 1998. Predicting numbers of an insect (*Neophilaenus lineatus*: Homoptera) in a changing climate. *Journal of Animal Ecology* **67**:987–991.