



Effects of grazing and biogeographic regions on grassland biodiversity in Hungary – analysing assemblages of 1200 species

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ABSTRACT

Agricultural intensification is a major threat to biodiversity. Agri-environment schemes, the main tools to counteract negative impacts of agriculture on the environment, are having mixed effects on biodiversity. One reason for this may be the limited number of species (groups) covered by most studies. Here, we compared species richness and abundance of 10 different species groups on extensively (0.5 cattle/ha) and intensively (1.0–1.2 cattle/ha) grazed semi-natural pastures in 42 fields in three Hungarian regions. Plants, birds and arthropods (leafhoppers, true bugs, orthopterans, leaf-beetles, weevils, bees, carabids, spiders) were sampled. We recorded 347 plant species, 748 territories of 43 bird species, and 51,883 individuals of 808 arthropod species. Compared to West European farmlands, species richness was generally very high. Grazing intensity had minor effects on α and β diversity, abundance and composition of the species assemblages. Region had significant effects on species richness and abundance of four taxa, and had strong effects on β diversity and species composition of all taxa. Regional differences therefore contributed significantly to the high overall biodiversity. We conclude that both grazing regimes deliver significant biodiversity benefits. Agri-environmental policy at the EU level should promote the maintenance of large scale extensive farming systems. At the national level, the effectiveness of agri-environment schemes should be improved via promoting and using research evidence.

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1. Introduction

Biological diversity declines at an alarming rate, and one of the main causes is intensification of agriculture in response to the demand for food, fibre and fuel (Tilman et al., 2001). In particular, the increased use of inorganic fertilizers, pesticides and machinery, and changes in land use influence biodiversity directly (de Heer et al., 2005). These changes have led to cascading effects, like loss of food resources for insectivorous birds, or change in pollination networks (Biesmeijer et al., 2006; Vickery et al., 2009; Batáry et al., 2010).

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The increased attention for biodiversity and the ecosystem services provided by it (such as pollination, biological control, seed dispersion), promoted the development of more nature friendly, sustainable forms of agriculture. The potential loss of income associated with nature-friendly management is in many countries financially compensated by means of agri-environment schemes (AES). AES are important drivers of land use in Europe, as most countries have agri-environmental programs. In the EU alone, the annual budget amounts to roughly €5 billion/year (Farmer et al., 2008). The available evidence suggests that AES have mixed effects on biodiversity. Conservation management may have positive, negative or no effect, on both targeted and non-targeted species groups (Kleijn and Sutherland, 2003; Kleijn et al., 2006).

Most of the studies that have been carried out so far share a number of biases that may have an impact on their outcome. First, there is a significant bias towards studies on one or a few popular taxa, like birds, butterflies or plants (Schuldt and Assmann, 2010). This is consistent with most patterns in ecology (e.g. Báldi and McCollin, 2003), but it provides a biased knowledge, which

is probably insufficient to adequately support decision making. The influence of farmland management on several species-rich and/or important taxa remains largely unknown. Additionally we still know very little about the impact of one type of management on a wide range of taxa. With the recent interest in ecosystem services, many of which are related to the diversity of species-rich and/or difficult-to-identify groups, studies that examine simultaneously the response of a wide range of taxa are urgently needed to support effective conservation planning (Schuldt and Assmann, 2010).

Second, most studies have been carried out in intensively farmed areas of West Europe (UK, The Netherlands, France, Germany) (Stoate et al., 2009). Extrapolation of research results from one biogeographic region to another is hazardous at best (Whittingham et al., 2007), suggesting we have very little information of what conservation strategies may be effective in the low intensity, species rich farmlands in Central and Eastern Europe (CEE) (Kleijn and Báldi, 2005; Stoate et al., 2009). Insight in conservation management that is effective in CEE countries is particularly valuable, as they host large populations of species that are declining or have gone extinct in several West European countries (Donald et al., 2002; Gregory et al., 2005).

Third, species richness and abundance as descriptors of assemblage structure are the most widely used measures of success or failure of farmland habitat management under AES. However, these are often misleading indicators of habitat quality (Vanhorne, 1983; Mortelliti et al., 2010), at least if not complemented by compositional analysis, the third basic descriptor of assemblages (Worthen, 1996). The composition of species assemblages is rarely considered in studies examining biodiversity responses to conservation management on farmland, although this can reveal important impacts since two assemblages may have the same species richness but nevertheless consist of completely different species. It is of high conservation relevance, as the protection of only one assemblage is seemingly sufficient to maintain biodiversity if measured as species richness alone, while composition can reveal the differences among assemblages.

In this study we evaluated biodiversity responses to different grazing regimes in semi-natural grasslands in Hungary. These grasslands cover 12% of the country, and are the most important agricultural habitat for biodiversity (Ángyán et al., 2003). Grasslands are managed by grazing and mowing. Fertilisers and pesticides are applied on less than 5% of Hungarian grasslands (Nagy, 1998; Nagy, pers. commun.). Recently, a number of papers have been published on a large scale field study carried out in the framework of the EU-funded “EASY” project (Kleijn et al., 2006). In these papers we mainly focused on individual taxa, and used taxon specific approaches and analyses (Báldi et al., 2005; Batáry et al., 2007a,b,c, 2008; Sárospataki et al., 2009). Here, we use the complete dataset, consisting of 10 taxa and approximately 1200 species (plants, birds and various arthropod taxa belonging to different functional groups) to provide a summary analysis on the effects of grazing intensity and regional differences. This will contribute to a better general understanding of effects of AES, because of the multi-taxon approach and the study location in the less known Pannonian region of CEE (Sundseth, 2009).

First, we compared species richness of Hungarian grasslands with data from West European farmlands collected with the same sampling protocol to see if CEE farmlands are indeed more diverse across many taxa than West European ones. Second, we evaluated the effects of grazing intensity and region on 10 taxa, using all three basic descriptors of assemblage structure, i.e. species richness, abundance and composition (Worthen, 1996). For the latter we explored compositional differences using diversity partitioning and multivariate techniques. Third, we evaluated the potential of the studied taxa to indicate the effect of grazing

intensity. Finally, we formulated recommendations for AES design that effectively maintains high biological diversity on low input farmlands.

2. Methods

2.1. Study areas

Study fields were equally distributed among the three most widespread grassland types in the Great Plain of Hungary (Molnár et al., 2008a), referred to here as the Alkali, the Meadow and the Heves biogeographic regions, respectively. The three regions differed in their grassland type and landscape structure. Two were located between the Danube and the Tisza Rivers (Fig. A1 in the Supplementary Data). The first, Alkali region was situated on the former flood plain of the Danube River, which is flat and is characterised by large landscape units. As a consequence of river regulations, salinisation has accelerated, resulting in secondary Pannonic alkali steppe vegetation on solonchak-solonetz soil, with common grass species (blue grass *Poa pratensis*, false sheep's fescue *Festuca pseudovina*, bermudagrass *Cynodon dactylon*), and salt resistant species (sea wormwood *Artemisia santonicum*, sea lavender *Limonium gmelini*, chamomile *Matricaria chamomilla*). The Meadow region was located in the northern part of the Danube-Tisza interfluvies. The main characteristic of this region was the patchy habitat structure: a mosaic of swamp meadows, calcareous purple moorgrass (*Molinia caerulea*) meadows, salt steppes and Pannonic sand steppe grasslands, with scattered woodlots and farms. Dominant plant species were blue grass, false sheep's fescue and bermudagrass, while characteristic species were purple moorgrass, tufted hairgrass (*Deschampsia caespitosa*) and cinquefoil (*Potentilla*) species. The Heves region was situated near the River Tisza, 100 km to the east of the two former regions (Fig. A1 in the Supplementary Data). It consists of dry and wet alkali grasslands and marshes on solonetz soil. Dominant plant species were blue grass, false sheep's fescue, quackgrass (*Elymus repens*) and *Scorzonera cana*. Characteristic species were the sea plantain (*Plantago maritima*), sea wormwood, whitetop (*Cardaria draba*) and yarrow (*Achillea*) species.

The extensively managed study fields fit the grassland management prescriptions of the Hungarian agri-environmental program: low density of livestock (0.5–1.2 animal/ha depending on pasture productivity, but set to 0.5 animal/ha for the studied regions), no use of artificial fertilisers or pesticides, no burning, winter grazing, reseeded, harrowing or ploughing, and maintaining clean ditches, and roadsides, etc.

2.2. Sampling design

We selected field pairs with high and low grazing pressure in the vicinity of each other, so that the systematic differences of fields within pairs can be attributed to the intensity of grazing and other environmental factors have little effect (Kleijn et al., 2006). Each region had 7 field pairs (42 fields in total), consisting of an extensively and an intensively grazed field. For both types, the intensity of grazing was roughly constant over the last five years. The grazing regimes were typical of the “puszta” grasslands. The cattle density was about 0.5 cattle per hectare on extensive, and 1–1.2 cattle per hectare on intensive fields. Except for grazing intensity there were no other differences in management. Within regions, fields were in the same grassland type. At the time of study, in 2003, agri-environment schemes (AES) had only just begun operating. Therefore, we were not able to compare fields with and without AES (cf. Kleijn et al., 2006). However, there were extensive fields managed by the national parks for years in the same way, as AES

regulations were set from 2004. Thus, the extensive fields were chosen from pastures managed according to AES regulations, although these regulations were effective only from 2004. Intensive fields were selected in some cases in heavily grazed parts of the same large pasture where extensive fields were chosen (14 cases), or in nearby intensive pastures of farmers (7 cases). These intensive fields were considered non-scheme fields. None of the fields were fertilised. The size of individual pastures was as large as 100 ha, sometimes over 1000 ha, and the number of grazing cattle was 100–400.

Two transects of ten 5×1 m plots 5 m apart were established in all fields, one in the edge of the grassland (but not in ecotone habitat), the other 50 m inside the grassland (Fig. A2). The number of vascular plant species, and their percentage cover was estimated for the 840 plots (3 regions, 7 field pairs/region, edge and interior transect in each field) once in 2003. Subsequently, relative cover per species and the total number of plant species (i.e. species richness per 100 m²) were determined for each field. Relative plant cover (%) was calculated by including bare ground cover. Data of edge and interior transects were pooled for each field.

One pitfall trap was located in between the central two plots of each transect. Spiders (Araneae) and carabids (Coleoptera: Carabidae) were identified from the samples. We used funnel pitfall traps (Fig. A2), because they are three times more efficient than cup traps in term of number of individuals (Duelli et al., 1999). A roof was set above each trap to protect it from rain. Pitfall traps were opened two weeks after the full bloom of dandelion (*Taraxacum* sp.) in 2003. The traps were emptied on the 14th day, the 28th day and then after a two week break (until 42nd day), on the 56th day (Kleijn et al., 2006).

Orthopterans (Orthoptera), leafhoppers (Hemiptera: Auchenorrhyncha), true bugs (Hemiptera: Heteroptera), bees (Hymenoptera: Apoidea), weevils (Coleoptera: Curculionidae) and leaf-beetles (Coleoptera: Chrysomelidae) were sampled using sweep netting along each transect in 2003. Three times twenty sweeps along a transect gave one sample. Sampling was repeated in May, June and July, although not all the three samples were used in most cases: for Orthoptera only the July sample was included, since earlier samples were dominated by larvae, which are not identifiable in several species. In addition to the netting, orthopterans were also identified using acoustic counts along the transects. These results were combined with sweep-net samples to obtain species numbers, but not for the analysis on abundance, where only sweep-net samples were included (Batáry et al., 2007c). For leafhoppers the June sample was identified, for Heteroptera, Curculionidae and Chrysomelidae the May and June samples were identified. Large bees and bumblebees may escape sweep netting; therefore additional sampling was carried out catching individuals with a butterfly net. A sample comprised three 5 min catching periods along the transect. Experts identified all arthropods to species level. Only imagos were included in the statistical analyses. Paired extensive and intensive fields were sampled on the same day by the same observer. For each arthropod taxa the number of species and abundance per field was used in the analyses.

Birds were censused in 12.5 ha large areas, which included the sample field of the transects. The areas were visited four times in the breeding season (April and May) of 2003. Censuses were carried out under good weather conditions (no wind and rain), from sunrise to 9–10 a.m. The observer spent at least 30 min at a sample field, slowly walking across the area. In cases where many individual birds were present, the census of a single field may have taken more than an hour. All bird observations, heard or seen, were recorded. Birds just flying through were excluded from the analysis. Based on the four visits, territories were identified (Batáry et al., 2007a).

2.3. Diversity partitioning

We used an additive partitioning of biodiversity, which is a natural measure of similarity among multiple assemblages: the proportion of total diversity found within communities (Lande, 1996). The total observed diversity γ_{obs} , for each management type and location in field combination, can be partitioned as:

$$\gamma_{\text{obs}} = \alpha + \beta_w + \beta_b$$

where α is the mean species richness per field, β_w (β_{within}) is the mean diversity of fields according to treatment (e.g. β diversity between the total species number and each of the seven fields of alkali extensive category), and β_b (β_{between}) is the mean diversity between the six treatments (3 regions, two grazing intensities) in relation to γ_{obs} . Calculations followed Clough et al. (2007) and Dahms et al. (2010).

2.4. Analysis of species richness, β diversity and abundance

For analysing the effects of grazing intensity (extensive versus intensive), regions (Alkali, Meadow, Heves) and their interaction on species richness (α diversity), β_{within} diversity and abundance of the studied taxa, we applied general linear mixed models (GLMM) with the Restricted Maximum Likelihood method. Pair was included in all models as random factor. The normality of model residuals was assessed using normal quantile–quantile plots, and data were either log or square root transformed, when necessary. Plant cover data were arcsine transformed prior to analysis in order to obtain normal distribution of residuals. Calculations were made using the nlme package (version 3.1, Pinheiro et al., 2009) for R 2.10.1 software (R Development Core Team, 2010).

GLMMs were performed for the ten taxa separately. In addition, we applied a multi-taxon approach, where species richness and abundance of the ten taxa in the extensive versus intensive fields were analysed by the Wilcoxon Signed Ranks Test, thus we evaluated if grazing has a general effect for all studied taxa together.

2.5. Analysis of species composition of assemblages

To measure the influence of management and region on the species composition of the studied taxa, we applied partial redundancy analyses (RDA). The species matrices were constrained by either management or region. Each species matrix was transformed with the Hellinger transformation (Legendre and Gallagher, 2001). This transformation allows the use of ordination methods such as PCA and RDA, which are Euclidean-based, with community composition data containing many zeros, i.e. characterised by long gradients (Legendre and Gallagher, 2001). Calculations were performed using the vegan package (version 1.17, Oksanen et al., 2010) of R 2.10.1 software (R Development Core Team, 2010).

3. Results

We recorded 347 plant species, 43 bird species with 748 territories and 808 arthropod species represented by 51,883 individuals (Fig. A3). The total number of observed species of the five taxa that had been sampled concurrently in five West European countries using the same sampling protocol was highest in Hungary for four of the taxa (Fig. 1). The exception is the spiders, for which Hungary, Spain and Switzerland held roughly similar numbers of observations.

In general, the extensively and intensively grazed fields had similar species numbers (α diversity) in each taxon (Table 1). Significant differences in species numbers were restricted to the leaf-beetles with higher species richness on extensively grazed

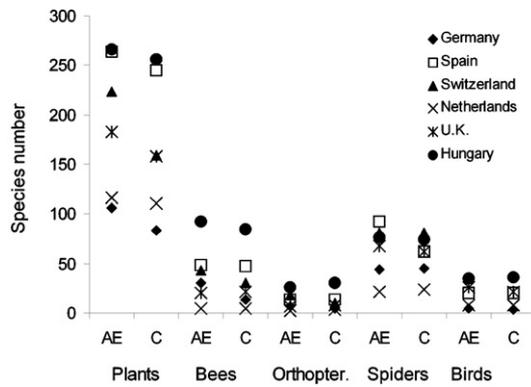


Fig. 1. Total species richness of plants, bees, orthopterans, spiders and birds on paired fields of extensively and intensively managed fields in Hungary, and in 5 west European countries (Germany, Netherlands, Spain, Switzerland, United Kingdom). AE means field with AES agreement (corresponds to the extensive grazing in Hungary), while C means conventionally managed fields (corresponds to intensive grazing in Hungary).

Data are from Kleijn et al. (2006) and this study.

fields (Table 2). When differences in species richness of all ten taxa were considered together, species richness on extensively grazed fields was significantly higher than that on intensively grazed fields (Wilcoxon test, $Z = 2.502$, $P = 0.012$).

β_{within} diversity was significantly different between extensively and intensively grazed fields for four out of ten taxa (Table 2). Three taxa had higher β_{within} diversity at intensively grazed fields (true bugs, carabids, birds), indicating larger differences among intensively grazed fields than among extensively grazed fields (Table 2).

Table 2

The effect of grazing intensity (management), grassland type (region) and their interaction on species richness (α diversity), β diversity and abundance of ten taxa in Hungary based on linear mixed models.

Taxon	Management		Region		Management \times region	
	F value	P	F value	P	F value	P
Species number						
Plant	1.056	0.318	22.346	<0.001	1.724	0.207
Leafhopper	3.208	0.090	16.694	<0.001	3.766	0.043
True bug	0.256	0.619	5.256	0.016	0.046	0.955
Orthopteran	0.302	0.589	6.747	0.007	1.172	0.332
Leaf-beetle	5.333	0.033 E > I	0.731	0.495	5.226	0.016
Weevil	0.001	0.973	2.403	0.119	0.189	0.830
Bee	0.200	0.660	0.920	0.417	0.278	0.760
Carabid	0.804	0.382	2.099	0.152	2.243	0.135
Spider	0.665	0.426	0.698	0.510	0.322	0.729
Bird	0.143	0.710	0.652	0.533	0.847	0.445
β_{within} diversity						
Plant	0.834	0.373	51.171	<0.001	4.111	0.034
Leafhopper	2.130	0.162	17.860	<0.001	3.843	0.041
True bug	11.810	0.003 E < I	11.529	<0.006	3.339	0.058
Orthopteran	0.257	0.618	27.530	<0.001	5.310	0.015
Leaf-beetle	21.821	<0.001 E > I	15.865	<0.001	41.209	<0.001
Weevil	0.867	0.364	47.971	<0.001	39.444	<0.001
Bee	0.574	0.459	10.917	0.001	18.657	<0.001
Carabid	9.589	0.006 E < I	3.827	0.041	4.614	0.024
Spider	0.665	0.426	13.120	<0.000	2.777	0.089
Bird	12.647	0.002 E < I	11.152	0.001	13.952	<0.001
Abundance						
Plant	6.576	0.020 E > I	5.398	0.015	2.301	0.129
Leafhopper	1.371	0.257	9.831	0.001	0.941	0.409
True bug	0.748	0.399	1.520	0.246	0.075	0.928
Orthopteran	7.589	0.013 E > I	24.133	<0.001	5.581	0.013
Leaf-beetle	1.210	0.286	0.495	0.618	0.926	0.414
Weevil	0.008	0.932	0.891	0.428	0.715	0.503
Bee	0.016	0.900	1.221	0.318	0.118	0.890
Carabid	5.831	0.027 E < I	0.753	0.485	1.921	0.175
Spider	0.107	0.747	4.706	0.023	0.575	0.573
Bird	29.635	<0.001 E > I	2.259	0.133	0.254	0.779

E: extensive fields, I: intensive fields. P values in bold are significant.

Table 1

Total species numbers and abundances of a wide range of taxa (plant, herbivore, pollinator, predator) from extensively and intensively grazed areas of three grassland types of Hungary.

Taxon	Species number		Abundance ^a	
	Extensive	Intensive	Extensive	Intensive
Plant	266	256	1709	1626
Leafhopper	79	69	11,968	15,299
True bug	116	104	4250	2425
Orthopteran	37	38	1868	1441
Leaf-beetle	76	64	1882	2321
Weevil	100	97	762	737
Bee	93	85	238	245
Carabid	77	75	1154	1636
Spider	79	73	2874	2783
Bird	35	36	463	285

^a Abundance is given as number of individuals, except for plants, where the % coverage of all species was summed, and birds, where number of territories is given.

In contrast, leaf-beetles had higher β_{within} diversity on extensively grazed fields than on intensively grazed fields.

Diversity partitioning revealed that local scale α diversity had the lowest contribution to total diversity, and $\beta_{between}$ had the highest (Fig. 2). This suggests that, assemblages at a given field can be relatively species-poor, but that differences between fields and especially regions are large thereby contributing to the general high species richness of Hungarian grasslands. The difference in abundance between extensively and intensively grazed fields varied according to taxa (Table 1). No difference was found with the Wilcoxon test, indicating a lack of consistent difference in abundance between extensive versus intensive fields within taxon

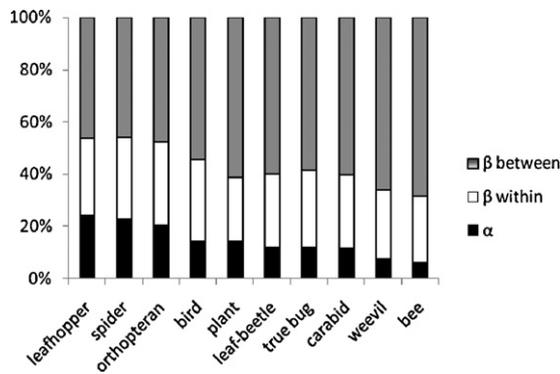


Fig. 2. Diversity partitioning of ten taxa from Hungarian semi-natural grasslands in order of decreasing α diversity. α is the mean species richness per field; β_{within} is the mean diversity within treatment (management and region); β_{between} is the between diversity among management and regions.

($Z=0.153$, $P=0.878$). This was supported by the GLMM results, with six taxa not showing any difference, three showing significantly higher abundance on extensively grazed fields than on intensively grazed fields, while one taxon showed the opposite pattern (Table 2).

Region significantly affected species number and abundance of four taxa each, while β_{within} differed for all taxa (Table 2), supporting the results of diversity partitioning (Fig. 2). It indicates that differences between regions are large compared to differences within regions or between fields with different grazing intensity. The response to grazing intensity of many taxa differed between regions, as indicated by significant management by region interactions. Effects of management on species richness were dependent on region for two taxa, while management effects on abundance and β_{within} diversity were dependent on region for one and eight taxa respectively (Table 2). For example, the mean difference between orthopteran species richness on extensively grazed and intensively grazed fields respectively was +5, -1 and -2, in three regions. For orthopteran abundance this difference was +85, +448, -109 in the three regions. This illustrates that species richness or abundance in extensively grazed fields can be higher than in intensively grazed fields in one region, but lower in another. The weak impacts of grazing management were supported by the analyses of the composition of the species assemblages. Although a significant part of the variation was explained by management in half of the taxa, it never amounted to more than a few percent of the total variation (Table 3). The composition of the species assemblages from the three regions, however, differed highly significantly for all taxa, explaining on average 18% of variations (Table 3). This indicates that regions with different vegetation and landscape types harbour largely distinct assemblages.

Table 3
Redundancy analysis (RDA) of community composition according to management (extensive versus intensive grazing) and region (three region/grassland types) of a wide range of taxa in Hungarian grasslands.

	Management			Region		
	Variation (%)	Pseudo-F	P	Variation (%)	Pseudo-F	P
Plant	2.37	1.196	0.246	22.45	5.672	<0.001
Leafhopper	3.39	1.775	0.018	23.95	6.264	<0.001
True bug	2.69	1.227	0.145	13.94	3.176	<0.001
Orthopteran	2.63	1.308	0.183	20.93	5.202	<0.001
Leaf-beetle	3.21	1.517	0.026	16.46	3.893	<0.001
Weevil	2.31	1.053	0.313	14.25	3.245	<0.001
Bee	2.72	1.166	0.158	8.55	1.831	<0.001
Carabid	3.12	1.458	0.048	15.57	3.638	<0.001
Spider	2.94	1.388	0.049	16.56	3.908	<0.001
Bird	3.52	1.729	0.036	19.02	4.666	<0.001

P values in bold are significant.

4. Discussion

We studied the biodiversity of semi-natural grasslands in Hungary, and recorded ca. 1200 species of 10 taxa collected in a large scale field sampling in 2003. A subsample of five taxa (plants, bees, orthopterans, spiders and birds) were compared with similarly obtained data from Dutch, English, German, Spanish and Swiss farmlands. For most taxa, Hungarian grasslands supported (considerably) larger species pools than the agricultural fields in other countries (Fig. 1, Batáry et al., 2010). Besides the generally less intensive farming in Hungary (Stoate et al., 2009; Báldi and Batáry, 2011), there are two mechanisms for this richness. First, it seems that species richness of one group may increase the richness of others, as Batáry et al. (2010) found that insect insect-pollinated plant richness was positively related to bee species richness. Second, our results suggest that the generally high species richness may be a result of the large dissimilarity of fields and regions. Possibly, in Hungary agricultural intensification has not yet homogenized the species assemblages across agricultural fields and regions. We have to note, however, that other factors, like biogeography or other large-scale processes also have effect on the distribution of biodiversity across European farmlands.

The effect of grazing pressure was relatively weak on all four measures at the taxon level that is on species richness (α diversity), β_{within} diversity, abundance and species composition. More exactly, species richness did not differ between fields with different grazing pressures for any taxa, while β_{within} diversity, abundance and species composition differed for few taxa only. It suggests that the studied difference in grazing pressure resulted for some taxa in a shift in species composition but not in different species numbers. Considering the high number of observed species on the Hungarian fields compared to West European farmland, we conclude that both levels of grazing pressure maintain high levels of biodiversity. Our results are in agreement with Hoste-Danyłow et al. (2010), who similarly found that four different management systems supported similar levels of biodiversity in extensive Polish grassland landscapes. However, both abandonment of grazing and intensification will probably have adverse effects on biodiversity. For example, Verhulst et al. (2004) demonstrated that in Central Hungary, bird species richness and abundance was significantly lower on fertilised than on extensively grazed grasslands. Abandoned grasslands had higher bird species richness and abundance than extensively managed grasslands, however, typical grassland species that are endangered in other parts of their range, like Skylark (*Alauda arvensis*) and Yellow Wagtail (*Motacilla flava*), were more abundant in extensively grazed fields.

The effect of biogeographic regions was strong on all measured assemblage parameters, including species composition. This indicates that extensively used areas in lowland Hungary are diverse and heterogeneous at large spatial scales, with different

regions supporting different sets of species. This is supported by the high β_{between} diversity observed in this study (46–69% of total diversity), which was almost twice as high as that observed in a study in Germany using the same sampling design and protocol in cereal fields (Clough et al., 2007). Dahms et al. (2010) measured β diversity in German grasslands (using a different sampling design so that comparisons have to be made bearing this in mind) and found that between grassland type β diversity never exceeded 28% (Fig. 1 in Dahms et al., 2010). Again this is considerably less than the β diversities observed in this study. These results suggest that in Hungary it is particularly important that the measures prescribed by agri-environment schemes maintain the differences between regions and prevent biodiversity homogenization across regions. In West Europe farmland communities have generally been homogenized as a result of the application of same high-input agricultural practices over large geographical areas. In Hungary, at least in grassland dominated regions, this is not (yet) the case, which may explain why biodiversity levels are still high compared to that in West European countries.

The dissimilarity of assemblages among regions calls for a management approach at that spatial level. Davey et al. (2010) described the same pattern while analysing the Entry Level Stewardship scheme of England. They found that farmland birds showed region-specific population trends and responses to the AES, supporting earlier findings by Whittingham et al. (2007), who showed that predictors from fields in one geographical region tended to have different effects on birds in other areas. These results are in line with ours as we found that the effect of grazing pressure vary in a wide range of taxa among the three studied regions in Hungary.

Species richness is the most widely used index of biodiversity, e.g. in assessing the success of AES. This is a simple index and easy to communicate to decision makers, but it is only one of several descriptors of assemblage structure (Worthen, 1996). Our results indicate that taxa that do not show any response when considering species numbers (species number in Table 2) may nevertheless be different when considering species composition as, for example, indicated by significant effects on β_{within} diversity (β diversity in Table 2). This was true for six out of ten taxa altogether. One possibility to avoid the problems of using species richness of a taxon is to analyse groups with similar traits. Earlier studies have suggested that farmland specialists are good indicators of the quality of extensively farmed habitats, while generalist species are often less clearly linked to habitat characteristics (e.g. Batáry et al., 2007b). Another way is to include compositional analysis in the studies, as in this paper, where contrary to species richness, compositional changes were considerable when we compared assemblages.

Many studies of farmland biodiversity use only one or a few taxa in their evaluation on the effects of management. However, management effects can be taxon-specific, which means that the same management may have different effects on different taxa. Not surprisingly, there are contradictory results in the literature, and it is not easy to figure out the reasons for differences, as studies were conducted in different fields and years. In this study we were able to demonstrate on 1200 species that the effect of management may vary across taxa (i.e. significant effect in some, but not all taxa, also depending on the used measure). In addition we showed that species richness had a consistent, but non-significant tendency to be larger in extensively grazed fields. However, if all the ten taxa were evaluated in one simple analysis, the difference was significant, showing that a multitaxon approach is an effective tool to detect even small differences in an ineffective measure.

Recently, a lot of effort has gone into finding efficient indicators of farmland biodiversity (de Heer et al., 2005). Our study involved ten taxa, providing the possibility to compare their sensitivity to grazing pressure using different measure of richness, abundance and composition of assemblages. No taxon showed

significant responses for all biodiversity measures. Birds, carabids and leaf-beetles showed significant effects for three measures, indicating that these species groups may be most sensitive to changes in grassland management. We propose to use more than one measure of biodiversity when evaluating management effects on biodiversity. Compositional analysis of assemblages, may offer the greatest insights. Important message is that both popular and the rarely used taxa were responsive to management differences. Therefore, it seems that there is no relationship between the popularity of a taxon and its sensitivity to grassland management, at least in Hungarian grasslands.

5. Implications for agri-environment schemes

Our study demonstrates that semi-natural grasslands in Hungary harbour a comparatively high farmland biodiversity compared to regions in Western Europe. This seems to be true both for fields with a grazing pressure according to agri-environment prescriptions, and for fields with higher grazing pressure (but without the use of agrochemicals). In countries with such extensive management the aim of schemes should be to prevent intensification. This probably can only be achieved by maintaining viable rural populations, small-scale farming, and nature-friendly, traditional agricultural management. The Hungarian Agri-environment program has schemes for all these measures, thus – in theory – providing potential solutions. Such schemes can be very effective both in terms of biodiversity and in terms of value-for-money, because they can maintain the already very species rich farmland habitats. If the maintenance of high levels of biodiversity is the objective of agri-environment schemes they should preferentially be implemented in traditionally managed, low-input farming systems because it is easier to conserve what is still there than to restore what has been lost in the intensively managed farmlands in West Europe (Marini et al., 2008; Kleijn et al., 2009, 2011). Therefore, an urgent task for the Hungarian agri-environment policy is to ensure the long-term operation of current grazing prescriptions, and to promote and use research evidence for other farmland types.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found in the online version, at doi:10.1016/j.agee.2012.03.005.

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Appendix A. Supplementary data (Fig. A1, A2 and A3) to the paper: Effects of grazing and biogeographic regions on grassland biodiversity in Hungary – analysing assemblages of 1200 species

Fig. A1. Location of the sample areas. Each dot represents a study field (red: Alkali region, blue: Meadow region, green: Heves region).



Fig. A2. Scheme of a site to sample plants and arthropods in Hungarian grasslands. The sampling included botanical plots, pitfall traps and sweep-netting. Sweep-netting was done on 95 m transects along the plots. We censused birds on 12.5 ha area (not shown), which included the sample site.

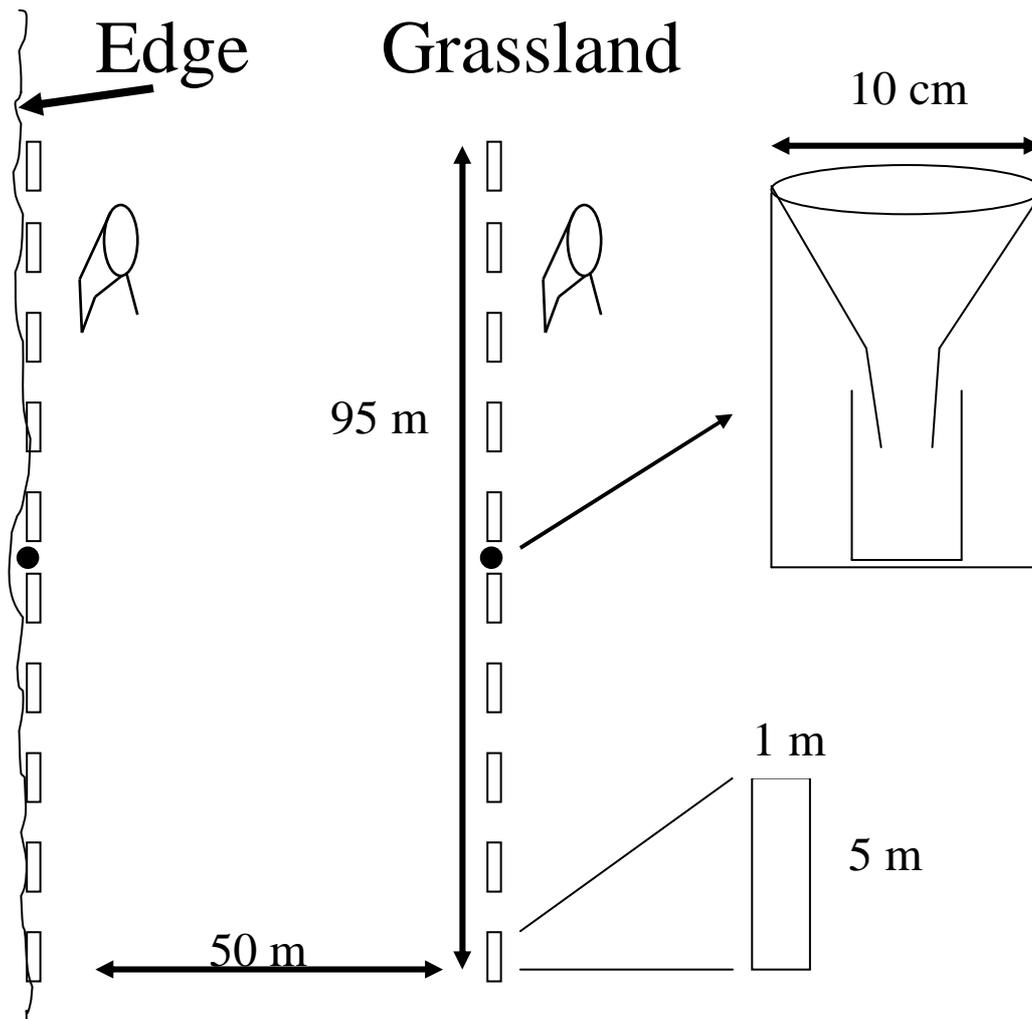


Fig. A3. Species richness (α diversity: white bars; and β_{within} diversity: grey bars) on the left figures, and abundance (percent coverage for plants, number of territories for birds, and number of individuals for all other taxa) on the right figures for ten taxa with SD. Data from Hungarian grasslands. AlkExt: extensively grazed grasslands in the Alkali region, MeaExt: extensively grazed grasslands in the Meadow region, HevExt: extensively grazed grasslands in the Heves region. AlkInt, MeaInt and HevInt are intensively grazed grasslands in the three regions.

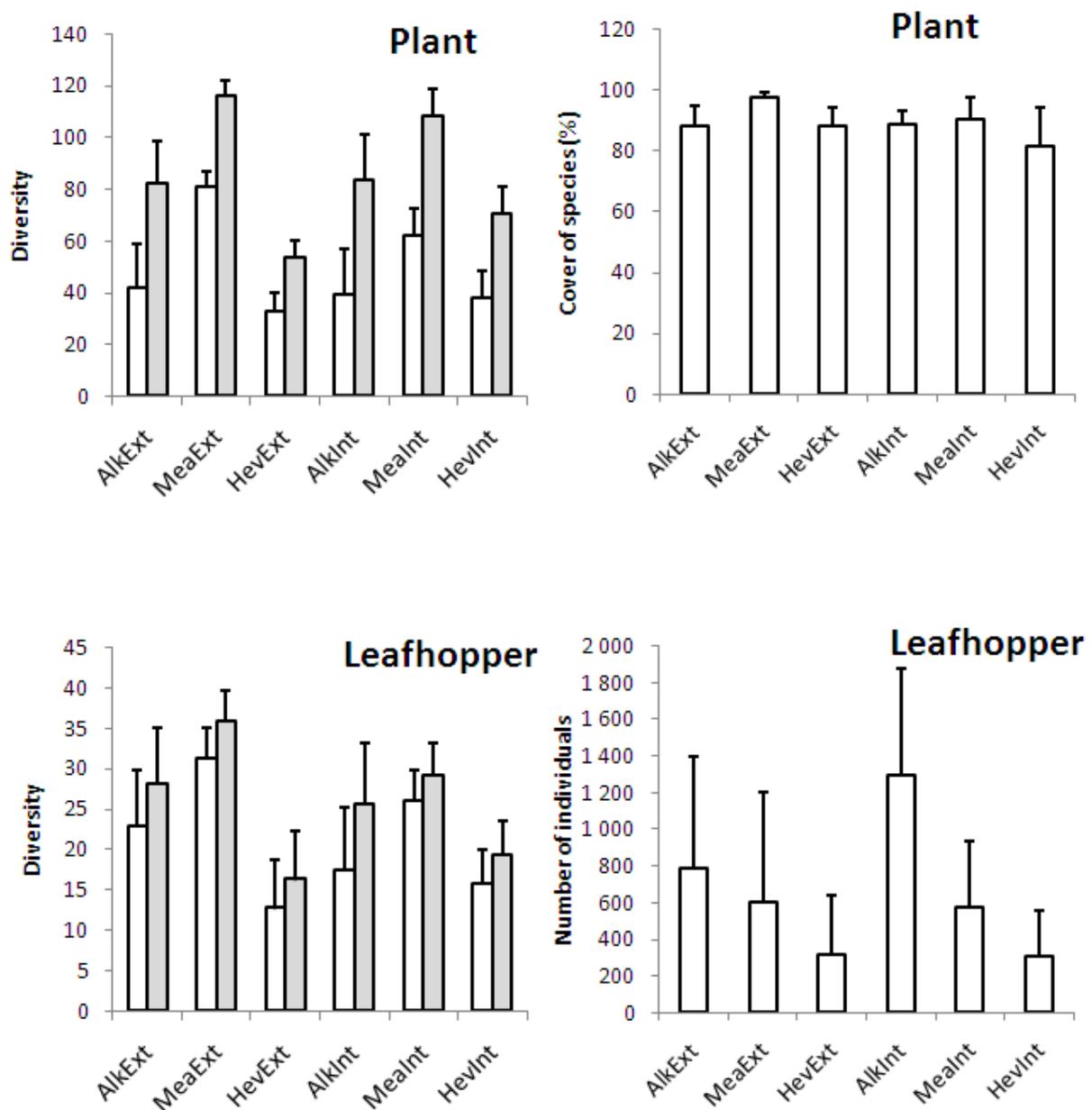


Fig. A3. Continued.

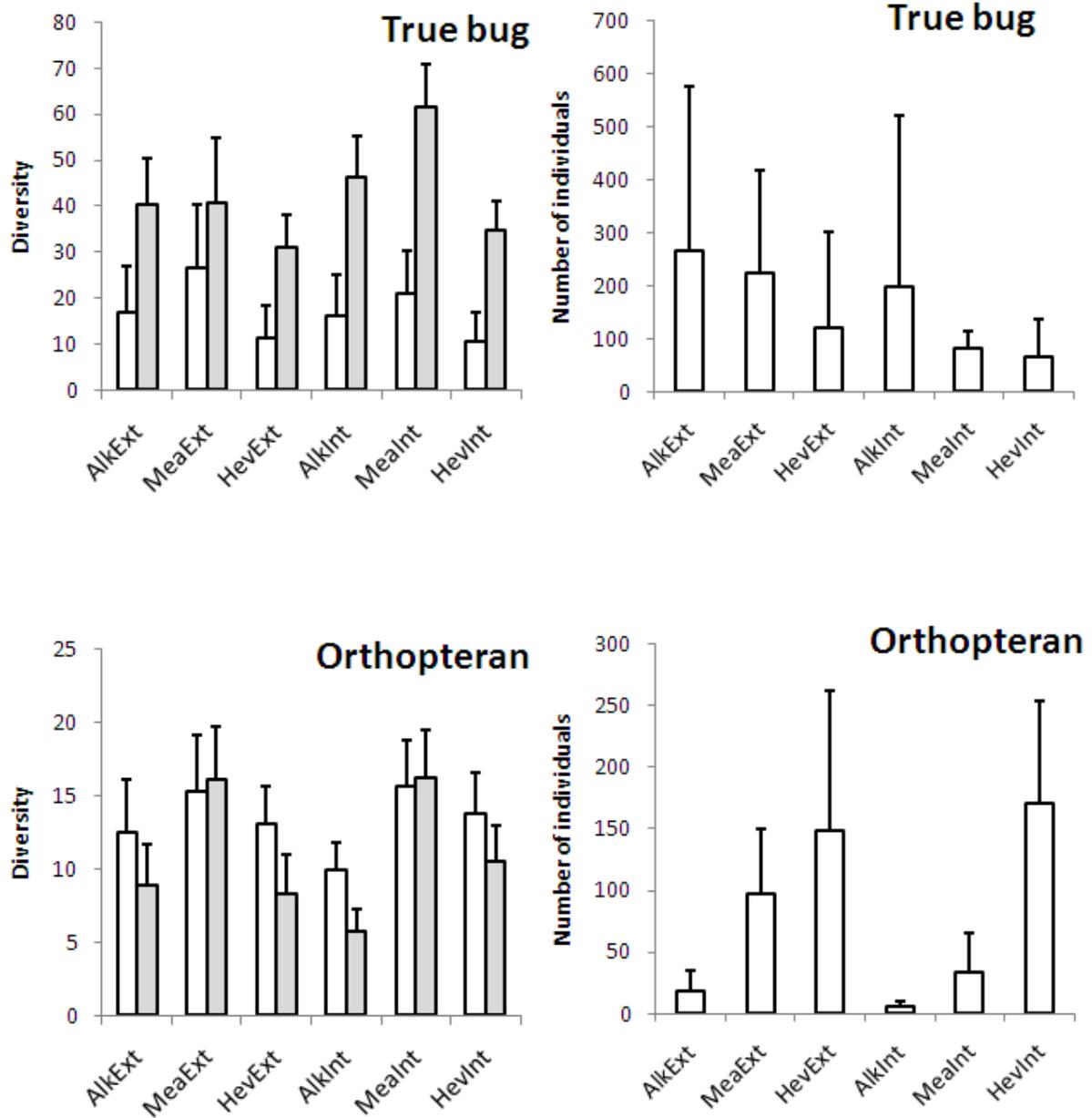


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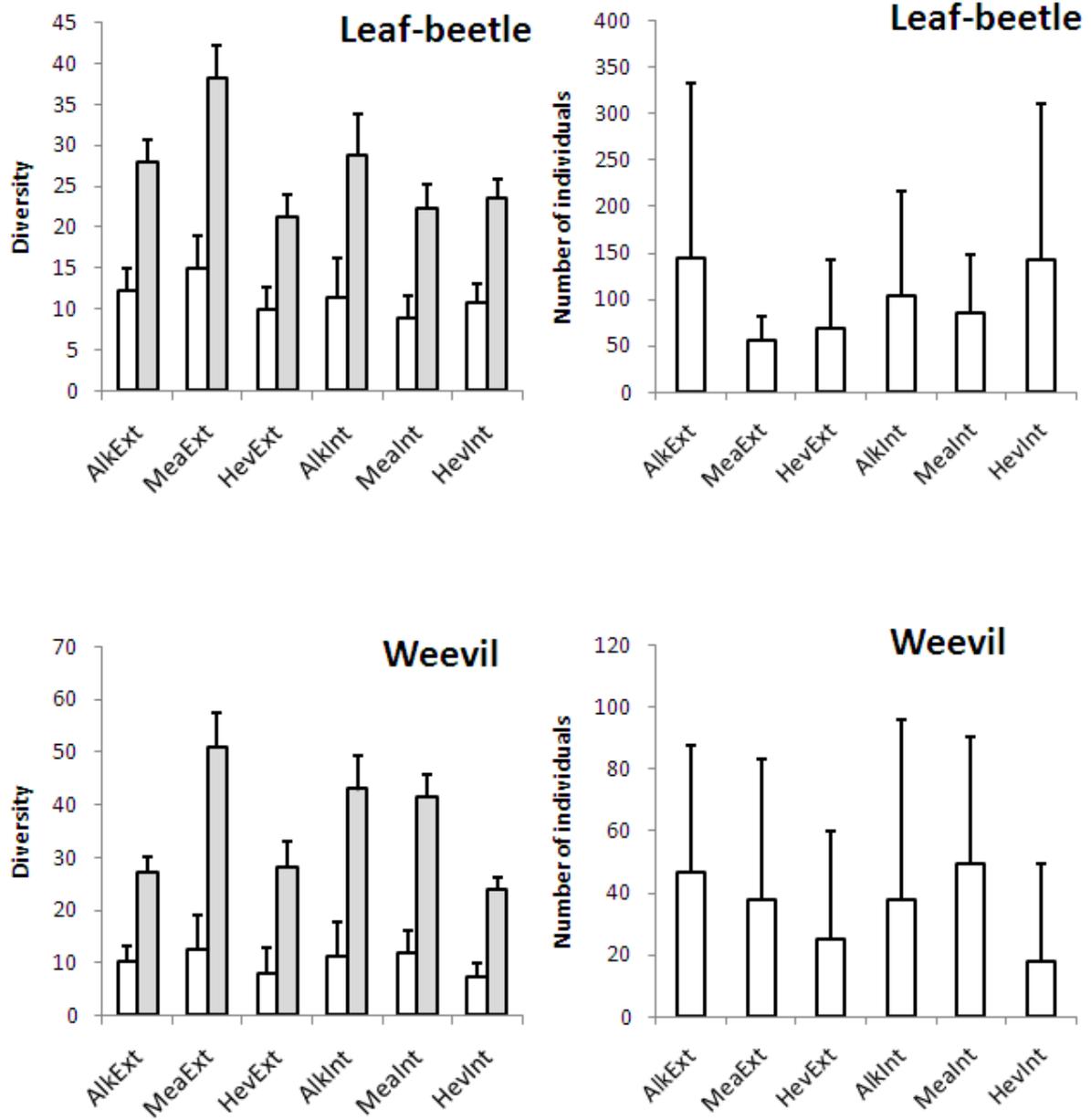


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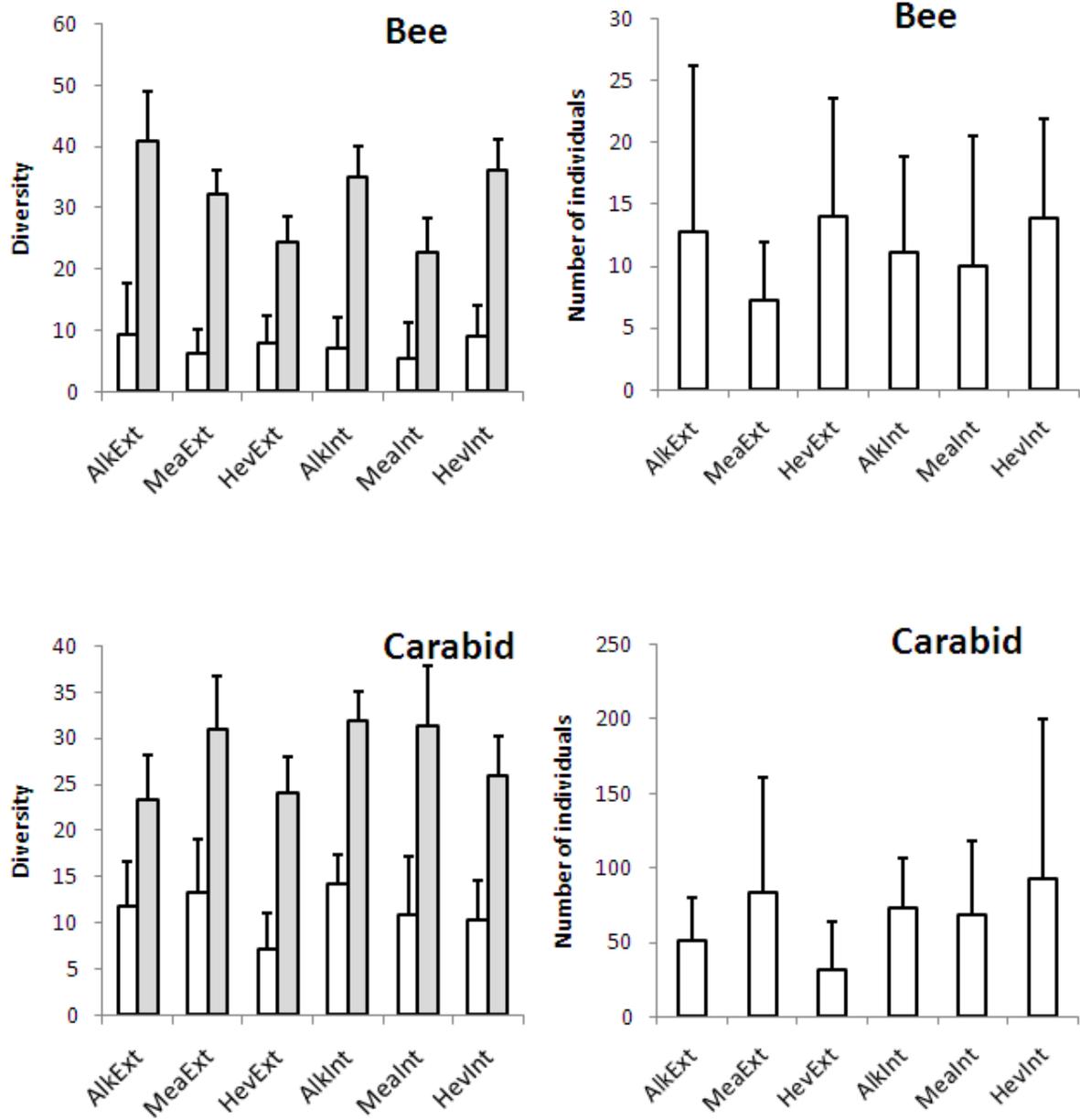


Fig. A3. Continued.

