

2
3 **Effects of grazing and biogeographic regions on grassland biodiversity in Hungary –**
4 **analysing assemblages of 1200 species**

5
6
7 Bál^a, A.^{a, b, *}, Batáry, P.^{c, d} & Kleijn, D.^e

8
9 ^aAnimal Ecology Research Group, Hungarian Academy of Sciences and Hungarian Natural
10 History Museum, Baross u. 13, H-1088 Budapest, Hungary

11 ^bMTA ÖK Centre for Ecological Research, Institute of Ecology and Botany, Alkotmány u. 2-
12 4, H-2163 Vác^rátót, Hungary, baldi.andras@okologia.mta.hu

13 ^cAgroecology, Georg-August University, Grisebachstr. 6, D-37077 Göttingen, Germany

14 ^dMTA-ELTE-MTM Ecology Research Group, Pázmány P. s. 1C, H-1117 Budapest, Hungary

15 ^eAlterra, Centre for Ecosystem Studies, Droevendaalsesteeg 3, P.O. Box 47, 6700 AA,
16 Wageningen, The Netherlands

17
18 *: Corresponding author is András Bál^a

19 MTA ÖK Centre for Ecological Research, Institute of Ecology and Botany, Alkotmány u. 2-
20 4, H-2163 Vác^rátót, Hungary, E-mail: baldi.andras@okologia.mta.hu

21 (Secondary e-mail: andrasbaldi@hotmail.com)

22 Phone (cell): +36-30 5485656

23 Skype name: abaldi

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

18

19

20

21 Keywords: agri-environment scheme; arthropod; assemblage composition; bird; Central
22 Europe; plant

23

24

1 **1. Introduction**

2

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

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

19 Most of the studies that have been carried out so far share a number of biases that may
20 have an impact on their outcome. First, there is a significant bias towards studies on one or a
21 few popular taxa, like birds, butterflies or plants (Schuldt & Assmann, 2010). This is
22 consistent with most patterns in ecology (e.g. Báldi & McCollin, 2003), but it provides a
23 biased knowledge, which is probably insufficient to adequately support decision making. The
24 influence of farmland management on several species-rich and/or important taxa remains
25 largely unknown. Additionally we still know very little about the impact of one type of
26 management on a wide range of taxa. With the recent interest in ecosystem services, many of
27 which are related to the diversity of species-rich and/or difficult-to-identify groups, studies
28 that examine simultaneously the response of a wide range of taxa are urgently needed to
29 support effective conservation planning (Schuldt & Assmann, 2010).

30 Second, most studies have been carried out in intensively farmed areas of West Europe
31 (UK, the Netherlands, France, Germany) (Stoate et al., 2009). Extrapolation of research
32 results from one biogeographic region to another is hazardous at best (Whittingham et al.,
33 2007), suggesting we have very little information of what conservation strategies may be
34 effective in the low intensity, species rich farmlands in Central and Eastern Europe (CEE)

1 (Kleijn & Báldi, 2005; Stoate et al., 2009). Insight in conservation management that is
2 effective in CEE countries is particularly valuable, as they host large populations of species
3 that are declining or have gone extinct in several West European countries (Donald et al.,
4 2002; Gregory et al., 2005).

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

16 In this study we evaluated biodiversity responses to different grazing regimes in semi-
17 natural grasslands in Hungary. These grasslands cover 12% of the country, and are the most
18 important agricultural habitat for biodiversity (Ángyán et al., 2003). Grasslands are managed
19 by grazing and mowing. Fertilisers and pesticides are applied on less than 5% of Hungarian
20 grasslands (Nagy 1998; G. Nagy pers. communication). Recently, a number of papers have
21 been published on a large scale field study carried out in the framework of the EU-funded
22 “EASY” project (Kleijn et al., 2006). In these papers we mainly focused on individual taxa,
23 and used taxon specific approaches and analyses (Báldi et al., 2005; Batáry et al., 2007a, b, c;
24 Batáry et al., 2008; Sárospataki et al., 2009). Here, we use the complete dataset, consisting of
25 10 taxa and approximately 1200 species (plants, birds and various arthropod taxa belonging to
26 different functional groups) to provide a summary analysis on the effects of grazing intensity
27 and regional differences. This will contribute to a better general understanding of effects of
28 AES, because of the multi-taxon approach and the study location in the less known Pannonian
29 region of CEE (Sundseth 2009).

30 First, we compared species richness of Hungarian grasslands with data from West
31 European farmlands collected with the same sampling protocol to see if CEE farmlands are
32 indeed more diverse across many taxa than West European ones. Second, we evaluated the
33 effects of grazing intensity and region on 10 taxa, using all three basic descriptors of
34 assemblage structure, i.e. species richness, abundance and composition (Worthen, 1996). For

1 the latter we explored compositional differences using diversity partitioning and multivariate
2 techniques. Third, we evaluated the potential of the studied taxa to indicate the effect of
3 grazing intensity. Finally, we formulated recommendations for AES design that effectively
4 maintains high biological diversity on low input farmlands.

5 6 7 **2. Methods**

8 9 **2.1. Study areas**

10
11 Study fields were equally distributed among the three most widespread grassland types in the
12 Great Plain of Hungary (Molnár et al., 2008a), referred to here as the Alkali, the Meadow and
13 the Heves biogeographic regions, respectively. The three regions differed in their grassland
14 type and landscape structure. Two were located between the Danube and the Tisza Rivers
15 (Fig. A1 in the Supplementary Data). The first, Alkali region was situated on the former flood
16 plain of the Danube River, which is flat and is characterised by large landscape units. As a
17 consequence of river regulations, salinisation has accelerated, resulting in secondary Pannonic
18 alkali steppe vegetation on solonchak-solonetz soil, with common grass species (blue grass
19 *Poa pratensis*, false sheep's fescue *Festuca pseudovina*, bermudagrass *Cynodon dactylon*),
20 and salt resistant species (sea wormwood *Artemisia santonicum*, sea lavender *Limonium*
21 *gmelini*, chamomile *Matricaria chamomilla*). The Meadow region was located in the
22 northern part of the Danube-Tisza interfluves. The main characteristic of this region was the
23 patchy habitat structure: a mosaic of swamp meadows, calcareous purple moorgrass (*Molinia*
24 *caerulea*) meadows, salt steppes and Pannonic sand steppe grasslands, with scattered
25 woodlots and farms. Dominant plant species were blue grass, false sheep's fescue and
26 bermudagrass, while characteristic species were purple moorgrass, tufted hairgrass
27 (*Deschampsia caespitosa*) and cinquefoil (*Potentilla*) species. The Heves region was situated
28 near the River Tisza, 100 km to the east of the two former regions (Fig. A1 in the
29 Supplementary Data). It consists of dry and wet alkali grasslands and marshes on solonetz
30 soil. Dominant plant species were blue grass, false sheep's fescue, quackgrass (*Elymus*
31 *repens*) and *Scorzonera cana*. Characteristic species were the sea plantain (*Plantago*
32 *maritima*), sea wormwood, whitetop (*Cardaria draba*) and yarrow (*Achillea*) species.

33 The extensively managed study fields fit the grassland management prescriptions of
34 the Hungarian agri-environmental program: low density of livestock (0.5-1.2 animal/ha

1 depending on pasture productivity, but set to 0.5 animal/ha for the studied regions), no use of
2 artificial fertilisers or pesticides, no burning, winter grazing, reseeding, harrowing or
3 ploughing, and maintaining clean ditches, and roadsides, etc.

6 **2.2. Sampling design**

7
8 We selected field pairs with high and low grazing pressure in the vicinity of each other, so
9 that the systematic differences of fields within pairs can be attributed to the intensity of
10 grazing and other environmental factors have little effect (Kleijn et al., 2006). Each region
11 had 7 field pairs (42 fields in total), consisting of an extensively and an intensively grazed
12 field. For both types, the intensity of grazing was roughly constant over the last five years.
13 The grazing regimes were typical of the "puszta" grasslands. The cattle density was about 0.5
14 cattle per hectare on extensive, and 1-1.2 cattle per hectare on intensive fields. Except for
15 grazing intensity there were no other differences in management. Within regions, fields were
16 in the same grassland type. At the time of study, in 2003, agri-environment schemes (AES)
17 had only just begun operating. Therefore, we were not able to compare fields with and
18 without AES (cf. Kleijn et al., 2006). However, there were extensive fields managed by the
19 national parks for years in the same way, as AES regulations were set from 2004. Thus, the
20 extensive fields were chosen from pastures managed according to AES regulations, although
21 these regulations were effective only from 2004. Intensive fields were selected in some cases
22 in heavily grazed parts of the same large pasture where extensive fields were chosen (14
23 cases), or in nearby intensive pastures of farmers (7 cases). These intensive fields were
24 considered non-scheme fields. None of the fields were fertilised. The size of individual
25 pastures was as large as 100 ha, sometimes over 1000 ha, and the number of grazing cattle
26 was 100-400.

27 Two transects of ten 5x1 m plots 5 m apart were established in all fields, one in the
28 edge of the grassland (but not in ecotone habitat), the other 50 m inside the grassland (Fig.
29 A2). The number of vascular plant species, and their percentage cover was estimated for the
30 840 plots (3 regions, 7 field pairs/region, edge and interior transect in each field) once in
31 2003. Subsequently, relative cover per species and the total number of plant species (i.e.
32 species richness per 100 m²) were determined for each field. Relative plant cover (%) was
33 calculated by including bare ground cover. Data of edge and interior transects were pooled for
34 each field.

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

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

25 Birds were censused in 12.5 ha large areas, which included the sample field of the
26 transects. The areas were visited four times in the breeding season (April and May) of 2003.
27 Censuses were carried out under good weather conditions (no wind and rain), from sunrise to
28 9-10 a.m. The observer spent at least 30 minutes at a sample field, slowly walking across the
29 area. In cases where many individual birds were present, the census of a single field may have
30 taken more than an hour. All bird observations, heard or seen, were recorded. Birds just flying
31 through were excluded from the analysis. Based on the four visits, territories were identified
32 (Batáry et al., 2007a).

33

34

1 **2.3. Diversity partitioning**

2

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

7

$$8 \gamma_{\text{obs}} = \alpha + \beta_w + \beta_b$$

9

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

15

16

17 **2.4. Analysis of species richness, beta diversity and abundance**

18

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

28

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

32

33

34 **2.5. Analysis of species composition of assemblages**

1
2 To measure the influence of management and region on the species composition of the
3 studied taxa, we applied partial redundancy analyses (RDA). The species matrices were
4 constrained by either management or region. Each species matrix was transformed with the
5 Hellinger transformation (Legendre & Gallagher, 2001). This transformation allows the use of
6 ordination methods such as PCA and RDA, which are Euclidean-based, with community
7 composition data containing many zeros, i.e. characterised by long gradients (Legendre &
8 Gallagher, 2001). Calculations were performed using the vegan package (version 1.17,
9 Oksanen et al., 2010) of R 2.10.1 software (R Development Core Team, 2010).

12 3. Results

13
14 We recorded 347 plant species, 43 bird species with 748 territories and 808 arthropod species
15 represented by 51,883 individuals (Fig. A3). The total number of observed species of the five
16 taxa that had been sampled concurrently in five West European countries using the same
17 sampling protocol was highest in Hungary for four of the taxa (Fig. 1). The exception is the
18 spiders, for which Hungary, Spain and Switzerland held roughly similar numbers of
19 observations.

20 In general, the extensively and intensively grazed fields had similar species numbers
21 (α diversity) in each taxon (Table 1). Significant differences in species numbers were
22 restricted to the leaf-beetles with higher species richness on extensively grazed fields (Table
23 2). When differences in species richness of all ten taxa were considered together, species
24 richness on extensively grazed fields was significantly higher than that on intensively grazed
25 fields (Wilcoxon test, $Z=2.502$, $P=0.012$).

26 β_{within} diversity was significantly different between extensively and intensively grazed
27 fields for four out of ten taxa (Table 2). Three taxa had higher β_{within} diversity at intensively
28 grazed fields (true bugs, carabids, birds), indicating larger differences among intensively
29 grazed fields than among extensively grazed fields (Table 2). In contrast, leaf-beetles had
30 higher β_{within} diversity on extensively grazed fields than on intensively grazed fields.

31 Diversity partitioning revealed that local scale α diversity had the lowest contribution
32 to total diversity, and β_{between} had the highest (Fig. 2). This suggests that, assemblages at a
33 given field can be relatively species-poor, but that differences between fields and especially
34 regions are large thereby contributing to the general high species richness of Hungarian

1 grasslands. The difference in abundance between extensively and intensively grazed fields
2 varied according to taxa (Table 1). No difference was found with the Wilcoxon test,
3 indicating a lack of consistent difference in abundance between extensive versus intensive
4 fields within taxon ($Z=0.153$, $P=0.878$). This was supported by the GLMM results, with six
5 taxa not showing any difference, three showing significantly higher abundance on extensively
6 grazed fields than on intensively grazed fields, while one taxon showed the opposite pattern
7 (Table 2).

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

27

28

29 **4. Discussion**

30

31 We studied the biodiversity of semi-natural grasslands in Hungary, and recorded ca. 1200
32 species of 10 taxa collected in a large scale field sampling in 2003. A subsample of five taxa
33 (plants, bees, orthopterans, spiders and birds) were compared with similarly obtained data
34 from Dutch, English, German, Spanish and Swiss farmlands. For most taxa, Hungarian

1 grasslands supported (considerably) larger species pools than the agricultural fields in other
2 countries (Fig. 1, Batáry et al., 2010). Besides the generally less intensive farming in Hungary
3 (Stoate et al., 2009, Báldi & Batáry, in press), there are two mechanisms for this richness.
4 First, it seems that species richness of one group may increase the richness of others, as
5 Batáry et al. (2010) found that insect insect-pollinated plant richness was positively related to
6 bee species richness. Second, our results suggest that the generally high species richness may
7 be a result of the large dissimilarity of fields and regions. Possibly, in Hungary agricultural
8 intensification has not yet homogenized the species assemblages across agricultural fields and
9 regions. We have to note, however, that other factors, like biogeography or other large-scale
10 processes also have effect on the distribution of biodiversity across European farmlands.

11 The effect of grazing pressure was relatively weak on all four measures at the taxon
12 level that is on species richness (α diversity), β_{within} diversity, abundance and species
13 composition. More exactly, species richness did not differ between fields with different
14 grazing pressures for any taxa, while β_{within} diversity, abundance and species composition
15 differed for few taxa only. It suggests that the studied difference in grazing pressure resulted
16 for some taxa in a shift in species composition but not in different species numbers.
17 Considering the high number of observed species on the Hungarian fields compared to West
18 European farmland, we conclude that both levels of grazing pressure maintain high levels of
19 biodiversity. Our results are in agreement with Hoste-Danyłow et al. (2010), who similarly
20 found that four different management systems supported similar levels of biodiversity in
21 extensive Polish grassland landscapes. However, both abandonment of grazing and
22 intensification will probably have adverse effects on biodiversity. For example, Verhulst et al.
23 (2004) demonstrated that in Central Hungary, bird species richness and abundance was
24 significantly lower on fertilised than on extensively grazed grasslands. Abandoned grasslands
25 had higher bird species richness and abundance than extensively managed grasslands,
26 however, typical grassland species that are endangered in other parts of their range, like
27 Skylark (*Alauda arvensis*) and Yellow Wagtail (*Motacilla flava*), were more abundant in
28 extensively grazed fields.

29 The effect of biogeographic regions was strong on all measured assemblage
30 parameters, including species composition. This indicates that extensively used areas in
31 lowland Hungary are diverse and heterogeneous at large spatial scales, with different regions
32 supporting different sets of species. This is supported by the high β_{between} diversity observed in
33 this study (46%-69% of total diversity), which was almost twice as high as that observed in a
34 study in Germany using the same sampling design and protocol in cereal fields (Clough et al.,

1 2007). Dahms et al. (2010) measured β diversity in German grasslands (using a different
2 sampling design so that comparisons have to be made bearing this in mind) and found that
3 between grassland type β diversity never exceeded 28% (Fig. 1 in Dahms et al., 2010). Again
4 this is considerably less than the β diversities observed in this study. These results suggest that
5 in Hungary it is particularly important that the measures prescribed by agri-environment
6 schemes maintain the differences between regions and prevent biodiversity homogenization
7 across regions. In West Europe farmland communities have generally been homogenized as a
8 result of the application of same high-input agricultural practices over large geographical
9 areas. In Hungary, at least in grassland dominated regions, this is not (yet) the case, which
10 may explain why biodiversity levels are still high compared to that in West European
11 countries.

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

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

33 Many studies of farmland biodiversity use only one or a few taxa in their evaluation
34 on the effects of management. However, management effects can be taxon-specific, which

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

10 Recently, a lot of effort has gone into finding efficient indicators of farmland
11 biodiversity (de Heer et al., 2005). Our study involved ten taxa, providing the possibility to
12 compare their sensitivity to grazing pressure using different measure of richness, abundance
13 and composition of assemblages. No taxon showed significant responses for all biodiversity
14 measures. Birds, carabids and leaf-beetles showed significant effects for three measures,
15 indicating that these species groups may be most sensitive to changes in grassland
16 management. We propose to use more than one measure of biodiversity when evaluating
17 management effects on biodiversity. Compositional analysis of assemblages, may offer the
18 greatest insights. Important message is that both popular and the rarely used taxa were
19 responsive to management differences. Therefore, it seems that there is no relationship
20 between the popularity of a taxon and its sensitivity to grassland management, at least in
21 Hungarian grasslands.

22
23

24 **5. Implications for agri-environment schemes**

25

26 Our study demonstrates that semi-natural grasslands in Hungary harbour a comparatively high
27 farmland biodiversity compared to regions in Western Europe. This seems to be true both for
28 fields with a grazing pressure according to agri-environment prescriptions, and for fields with
29 higher grazing pressure (but without the use of agrochemicals). In countries with such
30 extensive management the aim of schemes should be to prevent intensification. This probably
31 can only be achieved by maintaining viable rural populations, small-scale farming, and
32 nature-friendly, traditional agricultural management. The Hungarian Agri-environment
33 program has schemes for all these measures, thus – in theory – providing potential solutions.
34 Such schemes can be very effective both in terms of biodiversity and in terms of value-for-

1 money, because they can maintain the already very species rich farmland habitats. If the
2 maintenance of high levels of biodiversity is the objective of agri-environment schemes they
3 should preferentially be implemented in traditionally managed, low-input farming systems
4 because it is easier to conserve what is still there than to restore what has been lost in the
5 intensively managed farmlands in West Europe (Marini et al., 2008; Kleijn et al. 2009, 2011).
6 Therefore, an urgent task for the Hungarian agri-environment policy is to ensure the long-
7 term operation of current grazing prescriptions, and to promote and use research evidence for
8 other farmland types.

11 **Acknowledgements**

13 We are grateful for the comments of the referees and the editors which greatly improved the
14 manuscript. We thank the following persons for help in field work: András Bankovics, Anikó
15 Csecserits, Sarolta Erdős, Zsolt Erős, László Gálhidy, Júlia Honti, Béla Kancsal, Katalin
16 Kenderes, Eszter Kovács, Ildikó László, Barbara Lhotsky, Barbara Mihók, László Molnár,
17 Tamás Rédei and Rebeka Szabó. We are extremely grateful for the following individuals for
18 identification of various taxa and advices: Zsolt Józán and Miklós Sárospataki (bees), Tibor
19 Kisbenedek and Kirill Orci (orthopterans), András Orosz (Homoptera), Dávid Rédei
20 (Heteroptera), Győző Szél (carabids), Attila Podlussány (weevils), István Rozner (leaf-
21 beetles), and Tamás Szűts (spiders). Staff of the Kiskunság and Bükk National Parks gave
22 valuable help, among others we thank Jenő Farkas, András Máté, László Molnár, István
23 Nagy, László Tóth and Tibor Utassy. Dénes Vonah helped with the logistics in the Heves
24 area. We thank the Kiskunság and Bükk National Park Directorates for permissions, and
25 landowners János Csaplár, Pál Oláh and Ferenc Pongrácz for allowing us to work on their
26 fields. The study was supported by the EU-funded project 'EASY' (QLK5-CT-2002-01495),
27 and the paper writing partly by the Faunagenezis project (NKFP 3B023-04). A.B. was
28 supported by the Hungarian Academy of Sciences (Lendület program) and P.B. by the Bolyai
29 Research Fellowship of the Hungarian Academy of Sciences during the preparation of the
30 paper.

33 **Appendix A. Supplementary data**

1 Supplementary data associated with this article can be found, in the online version, at
2 [doi:xxxxxxx](https://doi.org/xxxxxxx).

5 **Literature**

- 7 Ángyán, J., Tardy, J., Vajnáné-Madarassy, A., (eds) 2003. Védett és érzékeny természeti
8 területek mezőgazdálkodásának alapjai. Mezőgazda, Budapest (in Hungarian).
- 9 Báldi, A., Batáry, P., 2011. The past and future of farmland birds in Hungary. *Bird Study*. 58,
10 365–377.
- 11 Báldi, A., McCollin, D., 2003. Island ecology and contingent theory: the role of spatial scale
12 and taxonomic bias. *Global Ecol. Biogeogr.* 12, 1–3.
- 13 Báldi, A., Batáry, P., Erdős, S., 2005. Effects of grazing intensity on bird assemblages and
14 populations of Hungarian grasslands. *Agric. Ecosyst. Environ.* 108, 251–263.
- 15 Batáry, P., Báldi, A., Erdős, S. 2007a. Grassland versus non-grassland bird abundance and
16 diversity in managed grasslands: local, landscape and regional scale effects. *Biodiv.*
17 *Conserv.* 16, 871–881.
- 18 Batáry, P., Báldi, A., Szél, G., Podlussany, A., Rozner, I., Erdős, S., 2007b. Responses of
19 grassland specialist and generalist beetles to management and landscape complexity.
20 *Divers. Distrib.* 13, 196–202.
- 21 Batáry, P., Orci, K.M., Báldi, A., Kleijn, D., Kisbenedek, T., Erdős, S., 2007c. Effects of local
22 and landscape scale and cattle grazing intensity on Orthoptera assemblages of
23 Hungarian Great Plain. *Basic Appl. Ecol.* 8, 280–290.
- 24 Batáry, P., Báldi, A., Samu, F., Szűts, T., Erdős, S., 2008. Are spiders reacting to local or
25 landscape scale effects in Hungarian pastures? *Biol. Conserv.* 141, 2062–2070.
- 26 Batáry, P., Báldi, A., Sárospataki, M., Kohler, F., Verhulst, J., Knop, E., Herzog, F., Kleijn,
27 D., 2010. Bees and insect-pollinated grassland plant communities in three European
28 countries. *Agric. Ecosyst. Environ.* 136, 35–39.
- 29 Biesmeijer, J.C., Roberts S.P.M., Reemer, M., Ohlemüller, R., Edwards, M., Peeters, T.,
30 Schaffers, A.P., Potts, S.G., Kleukers, R., Thomas, C.D., Settele, J., Kunin, W.E.,
31 2006. Parallel declines in pollinators and insect-pollinated plants in Britain and the
32 Netherlands. *Science* 313, 351–354.

- 1 Clough, Y., Kruess, A., Tschardtke, T., 2007. Local and landscape factors in differently
2 managed arable fields affect the insect herbivore community of a non-crop plant
3 species. *J. Appl. Ecol.* 44, 22–28.
- 4 Dahms, H., Mayr, S., Birkhofer, K., Chauvat, M., Melnichnova, E., Wolters, V., Dauber, J.,
5 2010. Contrasting diversity patterns of epigeic arthropods between grasslands of high
6 and low agronomic potential. *Basic Appl. Ecol.* 11, 6–14.
- 7 Davey, C., Vickery, J., Boatman, N., Chamberlain, D., Parry, H., Siriwardena, G., 2010.
8 Regional variation in the efficacy of Entry Level Stewardship in England. *Agric.*
9 *Ecosyst. Environ.* 139, 121–128.
- 10 de Heer, M. Kapos, V., ten Brink, B.J.E., 2005. Biodiversity trends in Europe: development
11 and testing of a species trend indicator for evaluating progress towards the 2010 target.
12 *Phil. Trans. R. Soc. B.* 360, 297–308.
- 13 Donald, P.F., Pisano, G., Rayment, M.D., Pain, D.J., 2002. The Common Agricultural Policy,
14 EU enlargements and the conservation of Europe's farmland birds. *Agric. Ecosyst.*
15 *Environ.* 89, 167–182.
- 16 Duelli, P., Obrist, M.K., Schmatz, D.R., 1999. Biodiversity evaluation in agricultural
17 landscapes: above-ground insects. *Agric. Ecosyst. Environ.* 74, 33–64.
- 18 Farmer, M., Cooper, T., Swales, V., Silcock, P., 2008. Funding for Farmland Biodiversity in
19 the EU: Gaining Evidence for the EU Budget Review. Institute of European
20 Environmental Policy, London.
- 21 Gregory, R.D., van Strien, A., Vorisek, P., Meyling, A.W.G., Noble, D.G., Foppen, P.B.,
22 Gibbons, D.W., 2005. Developing indicators for European birds. *Phil. Trans. R. Soc.*
23 *B.* 360, 269–288.
- 24 Hoste-Danyłow, A., Romanowski, J., Zmihorski, M., 2010. Effects of management on
25 invertebrates and birds in extensively used grassland of Poland. *Agric. Ecosyst.*
26 *Environ.* 139, 129–133.
- 27 Kleijn, D., Báldi, A., 2005. Effects of set-aside land on farmland biodiversity: comments on
28 Van Buskirk and Willi. *Conserv. Biol.* 19, 963–966.
- 29 Kleijn, D., Sutherland, W.J., 2003. How effective are European agri-environment schemes in
30 conserving and promoting biodiversity? *J. Appl. Ecol.* 40, 947–969.
- 31 Kleijn, D., Baquero, R.A., Clough, Y., Díaz, M., De Esteban, J., Fernández, F., Gabriel, D.,
32 Herzog, F., Holzschuh, A., Jöhl, R., Knop, E., Kruess, A., Marshall, E.J.P., Steffan-
33 Dewenter, I., Tschardtke, T., Verhulst, J., West, T.M., Yela, J.L., 2006. Mixed

- 1 biodiversity benefits of agri-environment schemes in five European countries. *Ecol.*
2 *Lett.* 9, 243–254.
- 3 Kleijn, D., Kohler, F., Báldi, A., Batáry, P., Concepción, E.D., Clough, Y., Díaz, M., Gabriel,
4 D., Holzschuh, A., Knop, E., Kovács, A., Marshall, E.J.P., Tschardtke, T., Verhulst, J.,
5 2009. On the relationship between farmland biodiversity and land-use intensity in
6 Europe. *Proc. R. Soc. B.* 276, 903–909.
- 7 Kleijn, D., Rundlöf, M., Scheper, J., Smith, H.G. & Tschardtke, T., 2011. Does conservation
8 on farmland contribute to halting the biodiversity decline? *Trends Ecol. Evol.* 26, 474–
9 481.
- 10 Lande, R., 1996. Statistics and partitioning of species diversity, and similarity among multiple
11 communities. *Oikos* 76, 5–13.
- 12 Legendre, P., Gallagher, E.D., 2001. Ecologically meaningful transformation for ordination of
13 species data. *Oecologia* 129, 271–280.
- 14 Marini, L., Fontana, P., Scotton, M., Klimek, S., 2008. Vascular plant and Orthoptera
15 diversity in relation to grassland management and landscape composition in the
16 European Alps. *J. Appl. Ecol.* 45, 361–370.
- 17 Molnár, Z., Bíró, M., Bölöni, J., Horváth, F., 2008a. Distribution of the (semi-)natural habitats
18 in Hungary I. Marshes and grasslands. *Acta Bot. Hung.* 50(Suppl), 59–105.
- 19 Mortelliti, A., Amori, G., Boitani, L., 2010. The role of habitat quality in fragmented
20 landscapes: a conceptual overview and prospectus for future research. *Oecologia* 163,
21 535–547.
- 22 Nagy, G., 1998. Ecological conditions, yield potential and grassland management in Hungary,
23 in: Nagy, G., Pető, K., (Eds.), *Ecological aspects of grassland management. Grassland*
24 *Science in Europe*, volume 3. European Grassland Federation, Zürich, pp. 1–13.
- 25 Oksanen, J., Blanchet, F.G., Kindt, R., Legendre, P., O'Hara, B., Simpson, G.L., Sólymos, P.,
26 Stevens, M.H.H., Wagner, H., 2010. The vegan package: Community Ecology
27 Package. R package version 1.17-2. URL: <http://vegan.r-forge.r-project.org>
- 28 Pinheiro, J., Bates, D., DebRoy, S., Sarkar, D., the R Core team, 2009. The nlme package:
29 Linear and nonlinear mixed effects models. R package version 3.1-96. URL:
30 <http://cran.r-project.org/src/contrib/Descriptions/nlme.html>
- 31 R Development Core Team, 2010. R: A language and environment for statistical computing.
32 R Foundation for Statistical Computing, Vienna. URL: <http://www.R-project.org>

- 1 Sárospataki, M., Báldi, A., Batáry, P., Józán, Z., Erdős, S., Rédei, T., 2009. Factors affecting
2 the structure of bee assemblages in extensively and intensively grazed grasslands in
3 Hungary. *Comm. Ecol.* 10, 182–188.
- 4 Schuldt, A., Assmann, T., 2010. Invertebrate diversity and national responsibility for species
5 conservation across Europe – A multi-taxon approach. *Biol. Conserv.* 143, 2747–
6 2756.
- 7 Stoate, C., Báldi, A., Beja, P., Boatman, N.D., Herzon, I., van Doorn, A., De Snoog, G.,
8 Rakosy, L., 2009. Ecological impacts of early 21st century agricultural change in
9 Europe. *J. Environ. Manage.* 91, 22–46.
- 10 Sundseth, K., 2009. Natura 2000 in the Pannonian Region. European Commission,
11 Environment Directorate General, Brussels
12 (<http://ec.europa.eu/environment/nature/info/pubs/docs/biogeos/pannonian.pdf>)
- 13 Tilman, D., Fargione, J., Wolff, B., D'Antonio, C., Dobson, A., Howarth, R., Schindler, D.,
14 Schlesinger, W., Simberloff, D., Swackhamer, D., 2001. Forecasting Agriculturally
15 Driven Global Environmental Change. *Science* 292, 281–284.
- 16 Vanhorne, B. 1983. Density as a misleading indicator of habitat quality. *J. Wildlife Manage.*
17 47, 893–901.
- 18 Verhulst, J., Báldi, A., Kleijn, D., 2004. Relationship between land-use intensity and species
19 richness and abundance of birds in Hungary. *Agric. Ecosyst. Environ.* 104, 465–473.
- 20 Vickery, J.A., Feber, R.E., Fuller, R.J., 2009. Arable field margins managed for biodiversity
21 conservation: A review of food resource provision for farmland birds. *Agric. Ecosyst.*
22 *Environ.* 133, 1–13.
- 23 Whittingham, M.J., Krebs, J.R., Swetnam, R.D., Vickery, J.A., Wilson, J.D., Freckleton, R.P.,
24 2007. Should conservation strategies consider spatial generality? Farmland birds show
25 regional not national patterns of habitat association. *Ecol. Lett.* 10, 25–35.
- 26 Worthen, W.B., 1996. Community composition and nested- subset analyses: basic descriptors
27 for community ecology. *Oikos* 76, 417–426.

28
29

1 Table 1. Total species numbers and abundances of a wide range of taxa (plant, herbivore,
 2 pollinator, predator) from extensively and intensively grazed areas of three grassland types of
 3 Hungary.

4

Taxon	Species number		Abundance ¹	
	Extensive	Intensive	Extensive	Intensive
Plant	266	256	1709	1626
Leafhopper	79	69	11968	15299
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

5 ¹ abundance is given as number of individuals, except for plants, where the % coverage of all
 6 species was summed, and birds, where number of territories is given

7

Table 2. The effect of grazing intensity (Management), grassland type (Region) and their interaction on species richness (alpha diversity), beta diversity and abundance of ten taxa in Hungary based on linear mixed models.

Taxon	Management		Region		Management x Region	
	F value	P	F value	P	F value	P
Plant	1.056	0.318	22.346	<.001	1.724	0.207
Leafhopper	3.208	0.090	16.694	<.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	<.001	4.111	0.034
Leafhopper	2.130	0.162	17.860	<.001	3.843	0.041
True bug	11.810	0.003 E<I	11.529	<.006	3.339	0.058
Orthopteran	0.257	0.618	27.530	<.001	5.310	0.015
Leaf-beetle	21.821	<.001 E>I	15.865	<.001	41.209	<.001
Weevil	0.867	0.364	47.971	<.001	39.444	<.001
Bee	0.574	0.459	10.917	0.001	18.657	<.001
Carabid	9.589	0.006 E<I	3.827	0.041	4.614	0.024
Spider	0.665	0.426	13.120	<.000	2.777	0.089
Bird	12.647	0.002 E<I	11.152	0.001	13.952	<.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	<.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	<.001 E>I	2.259	0.133	0.254	0.779

*: P<0.05; **: P<0.01; ***: P<0.001; E: extensive fields, I: intensive fields. P values in bold are significant.

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- <i>F</i>	<i>P</i>	Variation	pseudo- <i>F</i>	<i>P</i>
	(%)			(%)		
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.

Legend to figures:

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.

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.

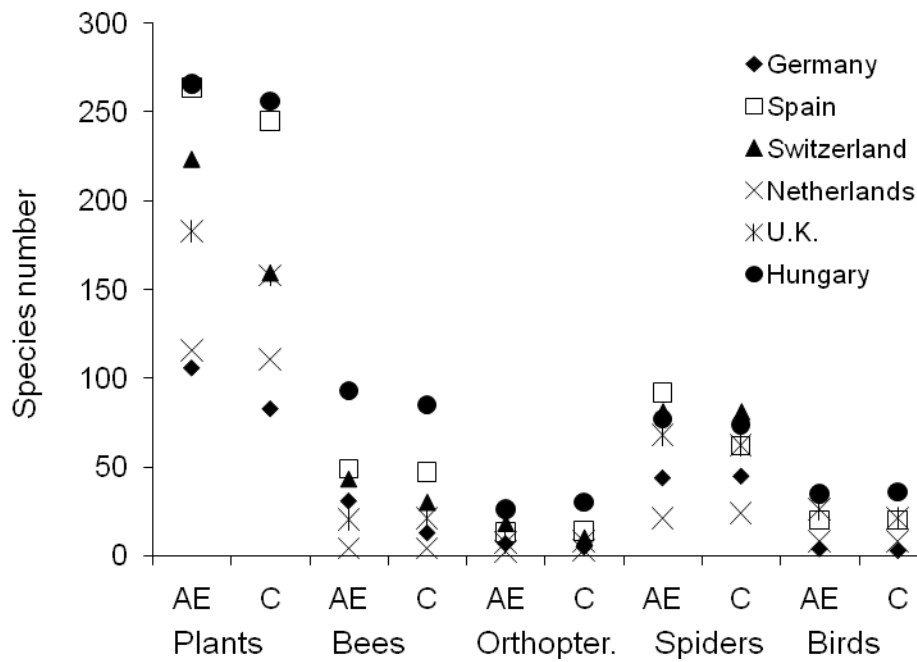


Fig. 1. Total species richness of plants, bees, orthopterans, spiders and birds on paired fields of extensively and intensively managed fields in Hungary, and in 5 west European countries (Germany, Netherlands, Spain, Switzerland, United Kingdom). AE means field with AES agreement (corresponds to the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

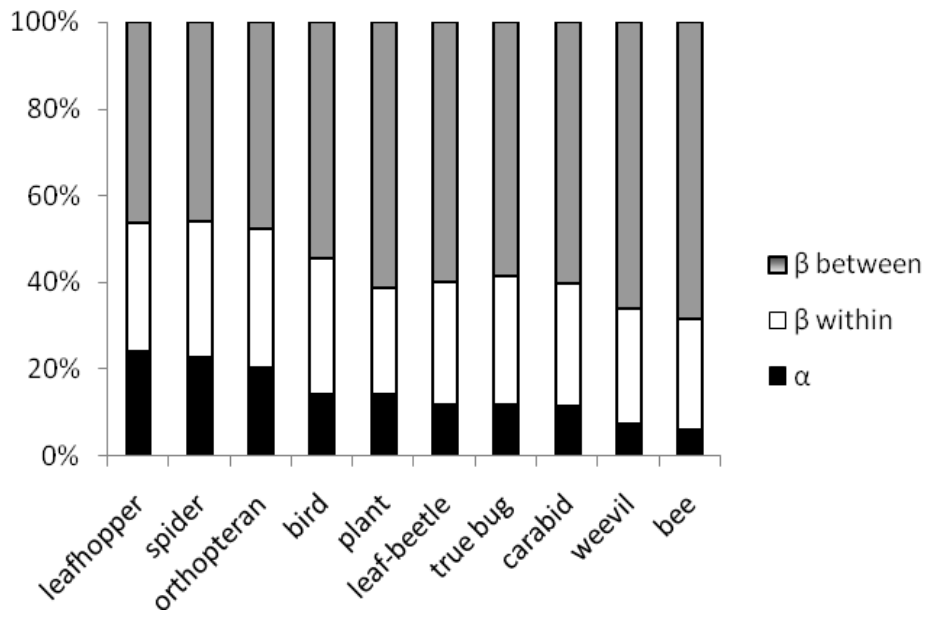


Fig. 2. Diversity partitioning of ten taxa from Hungarian semi-natural grasslands in order of decreasing α diversity. α is the mean species richness per field; β_{within} is the mean diversity within treatment (management and region); β_{between} is the between diversity among management and regions.

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.

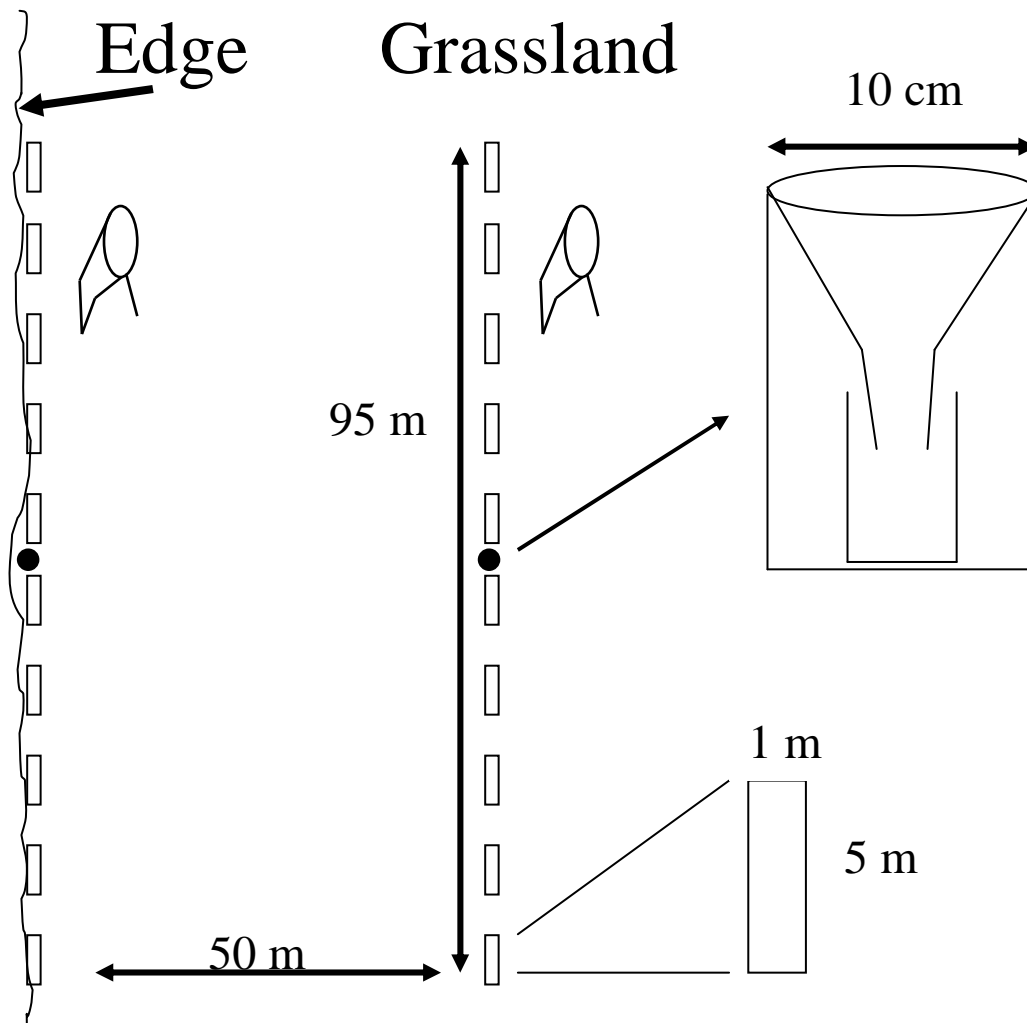


Fig. A3. Species richness (α diversity: white bars; and β_{within} diversity: grey bars) on the left figures, and abundance (percent coverage for plants, number of territories for birds, and number of individuals for all other taxa) on the right figures for ten taxa with SD. Data from Hungarian grasslands. 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.

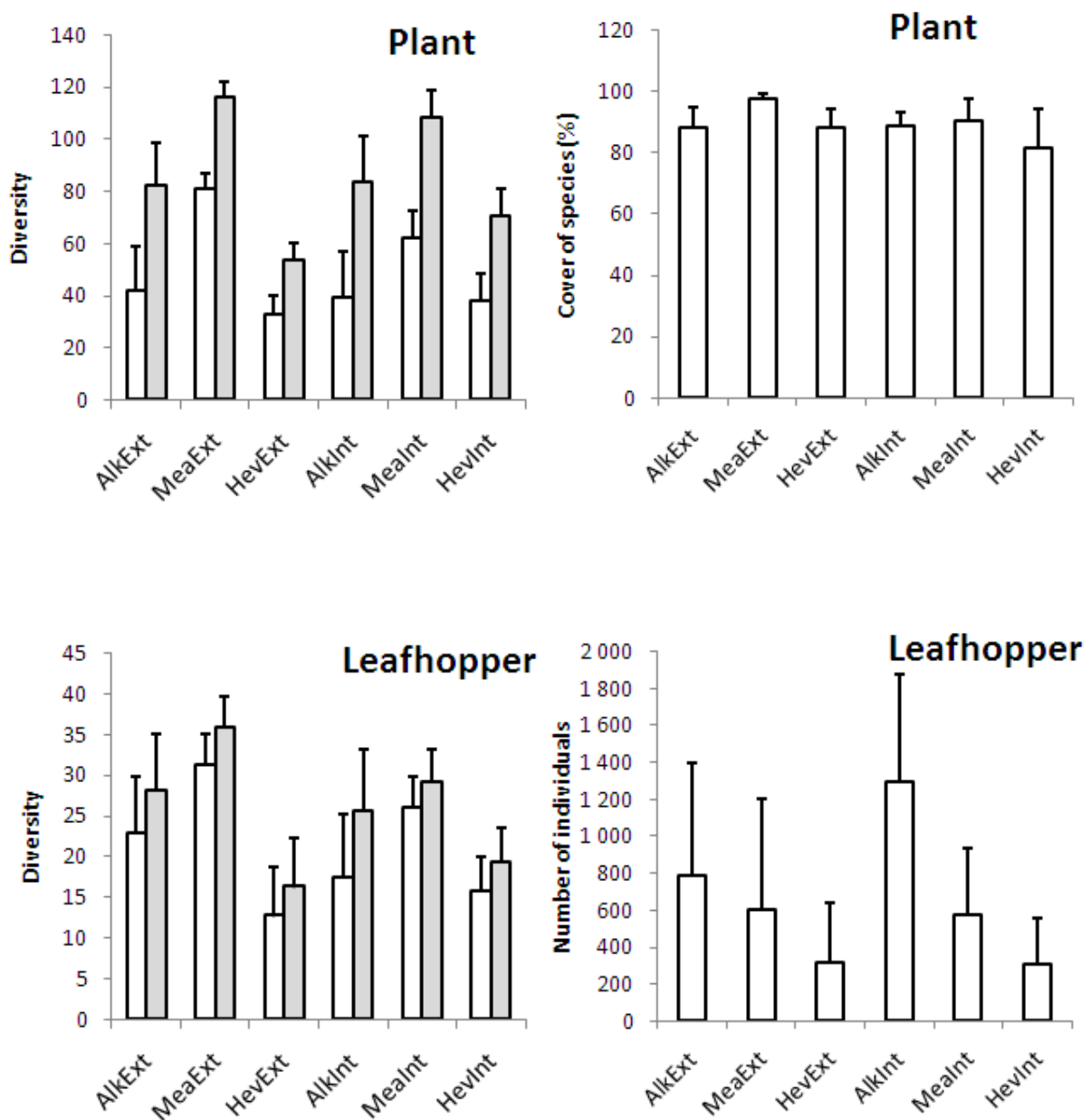


Fig. A3. Continued.

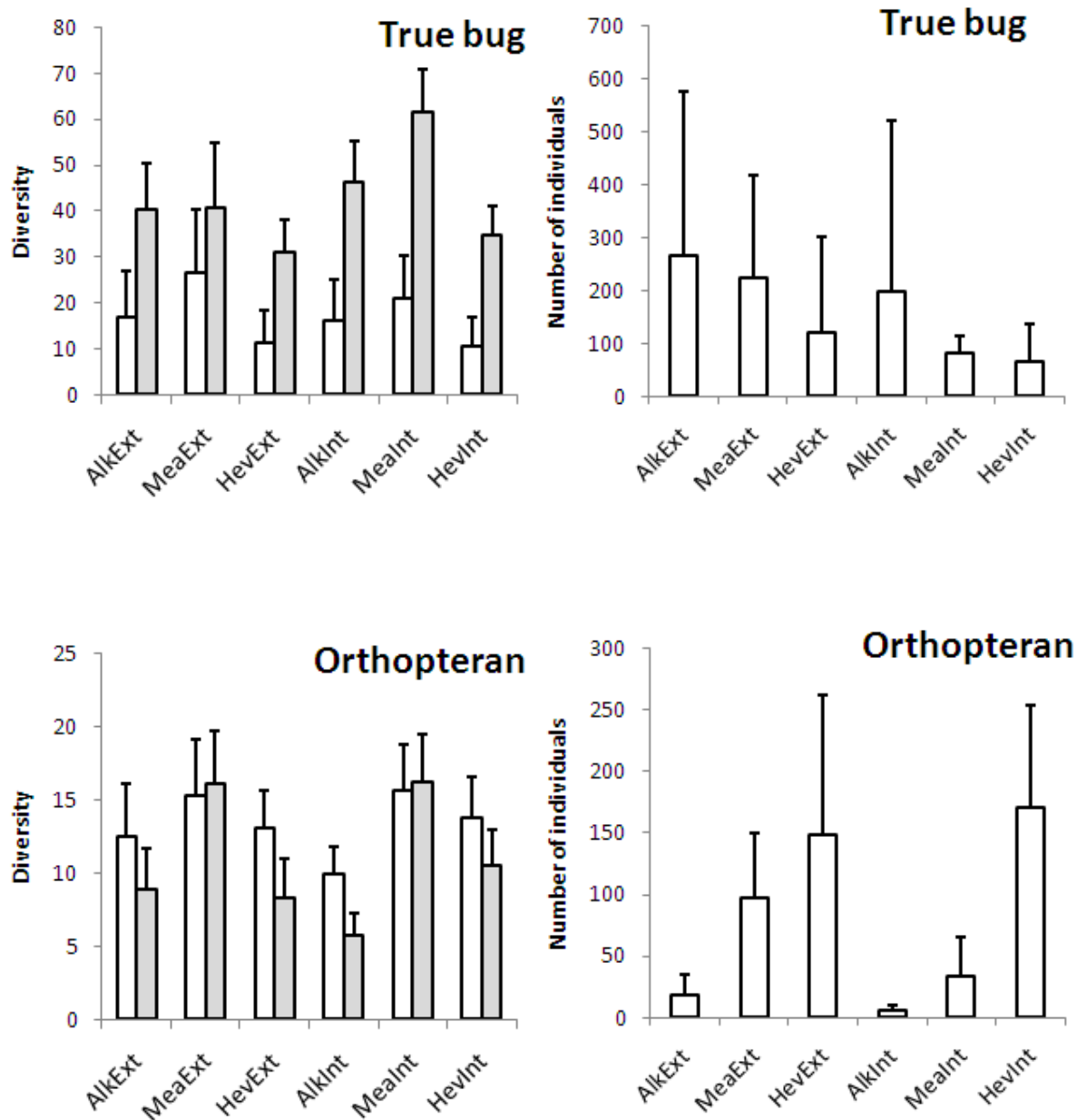


Fig. A3. Continued.

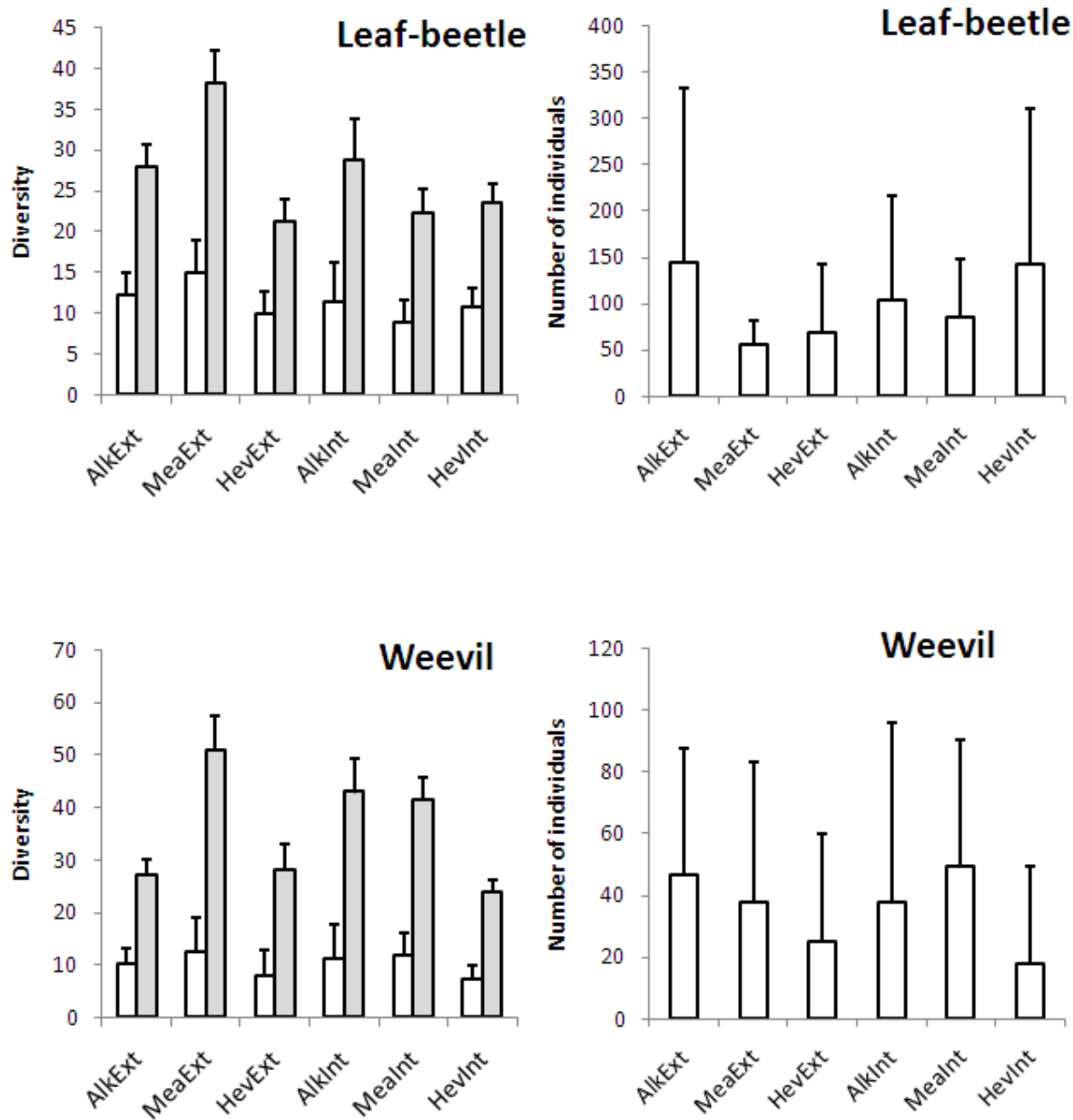


Fig. A3. Continued.

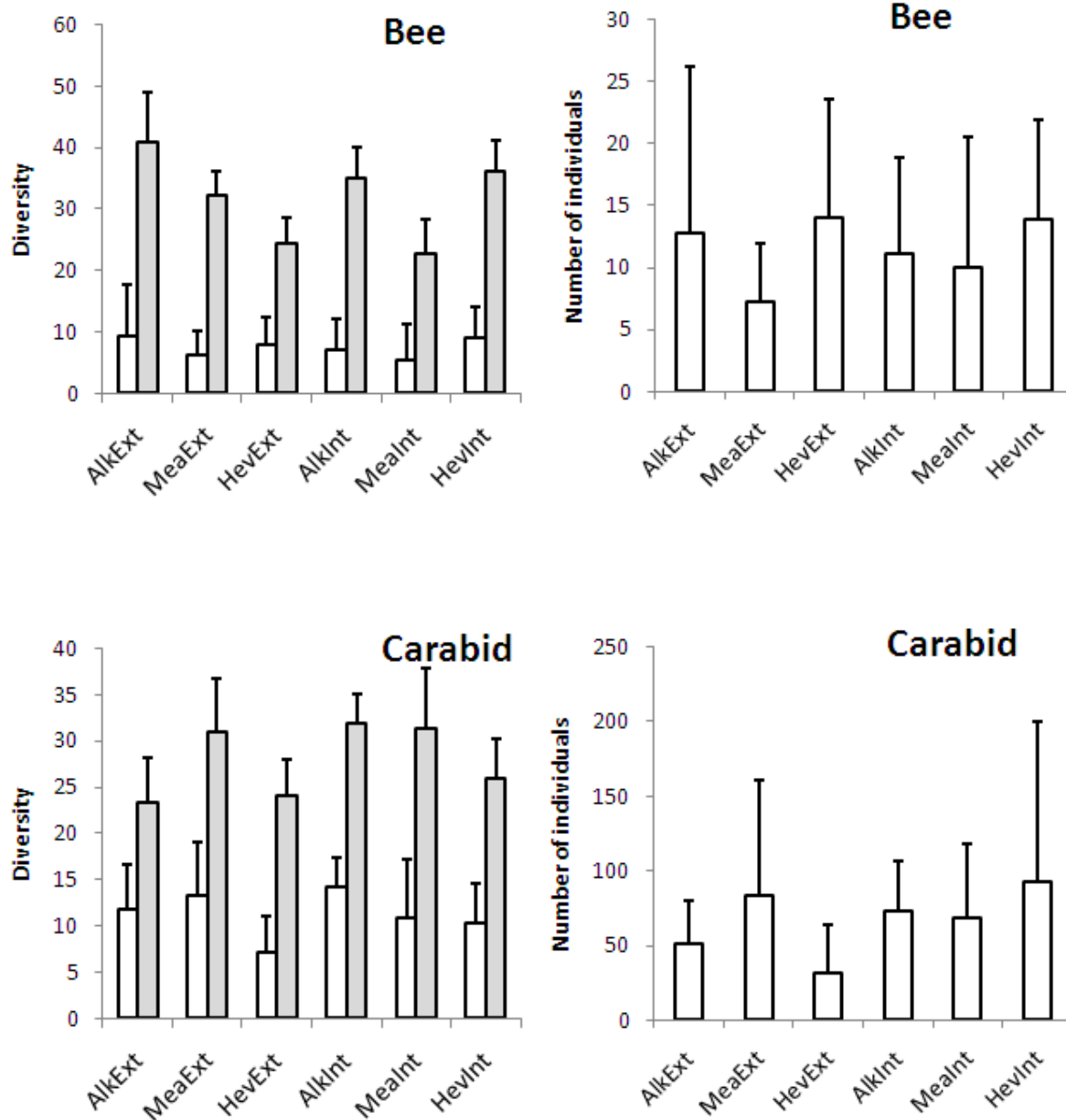


Fig. A3. Continued.

