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Effect of conservation management on bees and insect-pollinated grassland plant communities in three European countries

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ABSTRACT

It is now widely accepted that agricultural intensification drives the decline of biodiversity and related ecosystem services like pollination. Conservation management, such as agri-environment schemes (AES), has been introduced to counteract these declines, but in Western European countries these tend to produce mixed biodiversity benefits. Not much is known about the effects of AES in Central and Eastern European countries. We evaluated the effect of reduced stocking rates (0.5 cow/ha *vs.* >1 cow/ha) on bees and insect-pollinated plants in semi-natural pastures in Hungary. We sampled bees using sweep net and transect surveys in the edge and interior of the fields three times in 2003. On the same transects, we also estimated the cover of all plant species. We found no management effect on species richness and abundance with respect to cover of bees and insect-pollinated plants, but grazing intensity resulted in differences in species composition of insect-pollinated plants. Furthermore, we compared our results with those of a similar study carried out in Switzerland, and the Netherlands, but with different management regimes. There were positive effects of management in Switzerland, but conservation effects were lacking in the Netherlands. Species richness of both bees and insect-pollinated plants was highest in Hungary, intermediate in Switzerland and lowest in the Netherlands. Across all countries, the richness of insect-pollinated plants was a good predictor of bee species richness. Grassland extensification schemes were effective for bees and insect-pollinated plants in the country with intermediate land-use intensity and biodiversity only (Switzerland). The absence of effects in the Netherlands may have been caused by the management being highly intensive on both field types. In Hungarian grasslands biodiversity levels were high regardless of management and both investigated stocking rates may be qualified as conservation management. Therefore, agricultural policy in Hungary should encourage the maintenance of a variety of traditional grazing practices for conserving this still highly diverse pollinator fauna.

1. Introduction

It is now widely accepted that agricultural intensification poses the largest threat to biodiversity (Benton et al., 2003; Tscharntke et al., 2005). In many EU countries agri-environment schemes (AES) were initiated to halt the loss of biodiversity (Kleijn and Sutherland, 2003). The effectiveness of these schemes to increase biodiversity has been questioned (Kleijn et al., 2001; Kleijn and Sutherland, 2003).

Agricultural intensification may also adversely affect ecosystem services such as pollination (Tscharntke et al., 2005). Gallai et al. (2009) showed that pollinator decline may have considerable economic consequences. Klein et al. (2007) found that 87 of the leading food crops and 35% of global food production benefit from this ecosystem service. Furthermore, the proportion of land devoted to insect-pollinated crop production is increasing (Aizen et al., 2008). Recent studies seem to suggest that the most important pollinator group, bees, can be enhanced by conservation measures on farmland (e.g. Kleijn et al., 2001; Knop et al., 2006;

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Albrecht et al., 2007; Holzschuh et al., 2007; but see Sjödin et al., 2008; Winfree et al., 2008). It seems that bees, and the pollination services they provide, benefit also indirectly from the conservation measures through the increase of insect-pollinated plants (e.g. in organic cereal fields: Gabriel and Tscharntke, 2007; Holzschuh et al., 2007).

To date, however, most of the studies on the effectiveness of conservation measures were carried out in relatively intensively farmed landscapes. Tscharntke et al. (2005) hypothesized that effects of conservation measures would be most pronounced in landscapes with intermediate farming intensity. In simple landscapes, conservation measures may have little effect because less species are left to benefit from improved conditions. Complex landscapes still contain enough diversity for species spill over in poor quality sites, thus obscuring any effects of management. This hypothesis is difficult to confirm because data from extensively farmed areas are scarce and studies that can be used to compare the effects of management on bee species richness in both intensively farmed and extensively farmed areas are even rarer. One exception is the bumble bee study of Rundlöf et al. (2008), which was performed in paired organic and conventional cereal field borders and margins, with half of the pairs located in heterogeneous farmland and the remaining in homogeneous plains. They found an interaction effect between farming practice and landscape context so that species richness and abundance were only significantly higher on organic farms in homogeneous landscapes.

This study examines the effect of conservation management on bees and insect-pollinated plants in grasslands in Hungary, a post-socialist EU member country, where agricultural practices are markedly different from those in Western Europe (e.g. Donald et al., 2001). Moreover, because we collected our data in a similar fashion as Kohler et al. (2007), we were able to compare management effects, species richness and community composition in the extensively managed Hungarian grasslands with the more intensively farmed grasslands in Switzerland and the Netherlands. Our aims were (i) to analyse whether conservation management has an effect on species richness and composition of bees and insect-pollinated plants in Hungary, (ii) to compare the effects in Hungary to those in Switzerland and the Netherlands, and (iii) to discuss the differences among countries in the light of differences in land-use intensity.

2. Method

2.1. Study areas and field survey

The study sites ($N = 42$) in Hungary (HU) were extensively (0.5 cow/ha) and intensively (>1 cow/ha) grazed cattle pastures. All sites were unfenced (except 14% of the intensive ones) and grazed by cattle driven by a herdsman for at least five years from early spring until late autumn. The size of the pastures was often larger than 100 ha, on the average around 60 ha. About 95% of HU's grasslands are pesticide and fertiliser free providing valuable habitat for a rich flora and fauna, so in this sense they can be called semi-natural (Nagy, 1998). Our study areas are representative of this kind of Hungarian grassland. We selected 21 pairs of extensive and intensive fields in three study regions of the Hungarian Great Plain (seven pairs per region), the latter showing a clear difference in structural complexity. The area with the simplest landscape structure is situated in the Heves biogeographic region, which is characterised by large semi-natural grassland units sparsely dotted with bushes and trees. The Turján region, a mosaic of wet meadows, trees and marshlands has the most structurally complex landscapes. The third study area with intermediate structural complexity was situated parallel to the Danube, on the east side of the river on secondary alkali grassland (for detailed vegetation description and geographical location of study areas, see Báldi et al., 2005). Grassland management prescriptions of AES (implemented in Hungary in 2002) included restrictions on the density of livestock (0.5–1.2 animal/ha depending on pasture productivity) and termination of the use of artificial fertiliser or pesticides (Ángyán et al., 1999). Thus, in spite of the fact there was no Hungarian AES before 2002, the type of conservation management was similar to that of AES in Western European countries (Kleijn and Sutherland, 2003).

In each field, all samples were taken along two 95 m long transects: one along the edge and another, parallel to the first one, 50 m from the edge in the grassland interior. The extensive and intensive fields in a pair had the same soil type and groundwater level and were situated in similarly structured landscapes, therefore our paired design minimized the effects of possible confounding environmental variables. We sampled bees using sweep net (60 sweeps per transect per round) and transect surveys (15 min sampling per transect per round) in the edge and interior of the fields three times (May, June and July) in 2003. On the same transects, we also estimated the cover (in percentage) of all plant species in 20 plots ($5\text{ m} \times 1\text{ m}$) per field (840 plots in total). 10 plots, 5 m apart, were placed in the edge of the field, the other 10 were located in the interior of the field providing the edge resp. interior transects. For statistical analyses all data per field were pooled.

To put our study in a broader context we enriched our data with the data collected using the same sampling protocol by Kohler et al. (2007) in the Netherlands (NL) and Switzerland (CH) on grassland fields with and without AES. The Dutch scheme postponed the earliest seasonal mowing date and restricted pesticide use but not fertiliser use. Management prescriptions in CH were similar but prohibited the use of pesticides and fertilisers entirely. For a detailed description of schemes and study areas see Kohler et al. (2007).

2.2. Statistics

In order to measure the effect of extensification on insect diversity in the Hungarian fields, species richness and species abundances of insect-pollinated plants and bees (with and without honeybee, *Apis mellifera*) were analysed using General Linear Mixed Models (GLMM) with the Restricted Maximum Likelihood method. The models included the factors region, pair and management (extensive *vs.* intensive). Regions and pairs were random factors. If necessary, log-transformation was applied to handle non-normal distribution of residuals. Mean absolute plant cover values per field were arcsine transformed prior to analysis.

In order to study the management (extensive *vs.* intensive) effect on insect-pollinated plant species in detail, sub-groups of plant species were analysed. Thus, plant species were first classified according to traits, and subsequently their species richness and cover were analysed using the above-described GLMMs. We classified the plant species according to the following traits (see Appendix S1 in Supplementary Material): beginning of flowering early (January–April); May; June; late (July–September) and duration of flowering ($\leq$ two months; three months; four months; $\geq$ five months). Traits were extracted from the BIOFLOR database (Klotz et al., 2002). Calculations were made using the nlme package (version 3.1, Pinheiro et al., 2008) of R 2.8.0 software (R Development Core Team, 2008).

To measure the influence of management on bee and insect-pollinated plant species composition in Hungary, we applied partial redundancy analyses (RDA). The species matrices were constrained by management and geographical location (regions and pairs coded as factor variable). Each species matrix was

Table 1
Species richness and abundance of bees with or without *A. mellifera* and species richness and cover of insect-pollinated plants on extensively *vs.* intensively managed grasslands in Hungary, Switzerland and the Netherlands (mean ± SE per field). The data of the two latter countries are included for purpose of comparison (Kohler et al., 2007).

	Hungary		P	Switzerland		P	The Netherlands		P
	Extensive	Intensive		AES	non-AES		AES	non-AES	
Bee species richness with *A. mellifera*	7.7 ± 1.3	7.1 ± 1.1	ns	6.7 ± 0.7	4.8 ± 0.5	**	1.3 ± 0.3	1.4 ± 0.3	ns
Bee species richness without *A. mellifera*	6.8 ± 1.2	6.6 ± 1.0	ns	5.8 ± 0.6	3.8 ± 0.4	**	0.9 ± 0.2	1.3 ± 0.3	ns
Bee individuals with *A. mellifera*	11.3 ± 2.2	11.7 ± 1.9	ns	30.1 ± 4.3	45.5 ± 9.5	ns	3.0 ± 0.8	2.8 ± 0.9	ns
Bee individuals without *A. mellifera*	10.4 ± 2.0	11.1 ± 1.7	ns	11.4 ± 1.8	7.9 ± 1.3	*	2.3 ± 0.8	2.4 ± 0.9	ns
Insect-pollinated plant richness	35.8 ± 3.9	32.0 ± 3.0	ns	28.3 ± 2.1	21.8 ± 2.0	*	14.1 ± 0.9	12.6 ± 0.7	ns
Insect-pollinated plant cover	38.6 ± 5.4	30.4 ± 4.1	ns	35.6 ± 2.7	37.9 ± 3.1	ns	14.8 ± 1.5	11.6 ± 1.8	ns

* $P < 0.05$.
** $P < 0.01$.

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 1.16, Oksanen et al., 2008) of R 2.8.0 software (R Development Core Team, 2008).

Further analyses were done including data from the three countries. Firstly, to test the effect of management on bee species richness (with and without honeybee) in each country, and to compare between countries, we used sample-based rarefaction curves (Gotelli and Colwell, 2001). For each management type per country, an expected species accumulation (Mao Tau) curve was computed with the EstimateS 8.0 software (Colwell, 2006) using the bee data pooled across sampling periods and methods. To compare datasets we scaled the Mao Tau curves (and their 95% confidence interval curves) by individuals (Gotelli and Colwell, 2001). The rarified species richness observed in different types of management was significantly different if the 95% confidence intervals of sample-based rarefaction curves did not overlap.

Secondly, we related the total number of insect-pollinated plant species to the total number of bee species observed on AES (in HU extensive) and non-AES (in HU intensive) fields in the above-described GLMM models in the three countries. Management was not included in these models, because we expect that management would affect species richness. All models included the following random factors: region and pair. An additional analysis was also performed pooling the three countries' datasets and including the factor 'country' in the GLMM as the highest nesting level. In this model a variance function allowing different variances within countries was built, because the scatter plot of standardized residuals *vs.* fitted values was not uniform. Calculations were made using the nlme package (version 3.1, Pinheiro et al., 2008) of R 2.8.0 software (R Development Core Team, 2008).

3. Results

3.1. Effect of Hungarian grassland conservation

Overall, 483 bee individuals were caught in HU, belonging to 124 species (see Appendix S2 in Supplementary Material). The most abundant species were *Bombus terrestris*, *A. mellifera* and *Andrena flavipes* representing 9.3%, 6.4% and 6.2% respectively of all observed individuals. *A. mellifera* occurred on 31% of all fields. A total of 242 insect-pollinated plant species was observed, which represented 70% of all observed plant species.

We found no management effect on the number of species, abundance – or cover – of bees and insect-pollinated plants in HU (Table 1). For comparison, results of Kohler et al. (2007) are also included in Table 1. They mainly showed a positive effect on species richness in Switzerland and no effect in the Netherlands. Analysing the abundance of Hungarian insect-pollinated plants with different flowering times and periods, we found only two significant management effects: plant species which begin to flower in June had higher species richness on the extensive than on the intensive fields ($F_{1,20} = 6.013$, $p = 0.024$) and plants with short flowering periods (≤ two months) had higher cover on the extensive fields ($F_{1,20} = 5.914$, $p = 0.025$).

In the ordination analysis, geographical location (regions and pairs) explained a significant, but only small part of the variation in both the bee and insect-pollinated plant species matrices (4.6%, $p < 0.001$; 6.4%, $p = 0.001$, respectively). The effect of management

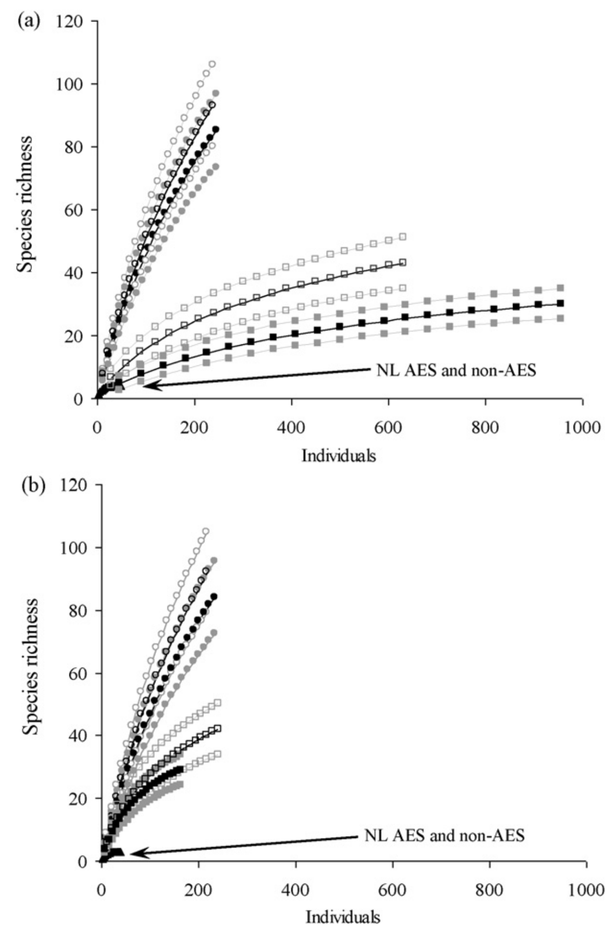

Fig. 1. Sample-based rarefaction curves standardized for number of bee individuals of extensively and intensively managed grasslands in Switzerland, Netherlands and Hungary. Error bars indicate 95% confidence intervals (faint dotted lines). The two Dutch curves, which cover each other, are very small compared to the other countries due to the very low species richness and number of individuals. (a) With and (b) without *A. mellifera*. (□) CH AES; (■) CH non-AES; (○) HU extensive; (●) HU intensive; (△) NL AES; (▲) NL non-AES.

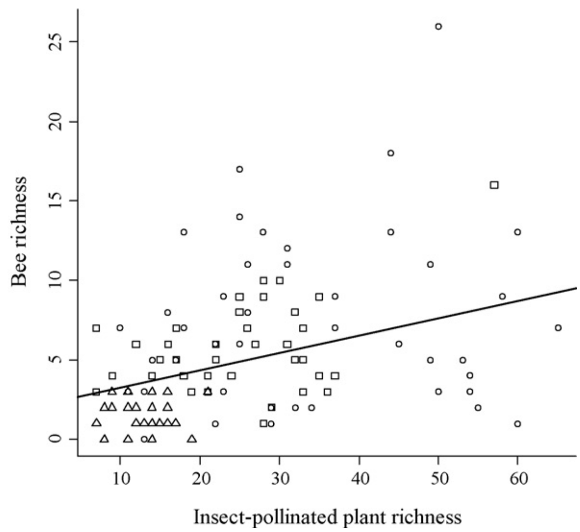

Fig. 2. Bee species richness is plotted against insect-pollinated plant species richness based on the data of CH, HU and NL (square, circle and triangle). The solid line represents the effect of insect-pollinated plant richness on bee species richness of the fitted linear mixed model.

was only significant for the insect-pollinated plants (4.0%, $p = 0.002$; bees: 3.4%, $p = 0.320$).

3.2. Comparison between countries

The rarefaction curves of bees suggest that there are no differences between fields with different management in any country except for bees, including honeybees, in Switzerland (Fig. 1). The observed difference in mean number of bee species per field in Switzerland was largely explained by difference in abundance (Table 1). The abundance of bees was higher on conventional fields, yielding significant differences in rarified species richness (95% confidence intervals did not overlap, Fig. 1a). Excluding honeybees resulted in lower bee abundance on conventional fields with no significant difference in rarified species richness between treatments (Fig. 1b). Although the mean number of observed species per field was similar in CH and HU the rarified species richness was more than twice as high in HU, regardless of management. This means that whereas the species richness' (alpha diversity) of these countries are similar, the species turnover (between-field beta diversity) is much higher in HU than in CH.

Analysing the effect of insect-pollinated plant richness on bee species richness, we found significant positive relationships in HU and CH ($F_{1,20} = 8.4$, $p = 0.009$; $F_{1,20} = 10.8$, $p = 0.004$, respectively), but no effect in NL ($F_{1,15} = 0.3$, $p = 0.603$). After pooling the data of the three countries, insect-pollinated plant richness was positively related to bee species richness (Fig. 2; $F_{1,57} = 13.7$, $p < 0.001$).

4. Discussion

The species richness and abundance of bees, and the species richness and cover of insect-pollinated plants on which they depend, were similar on intensively and extensively grazed grasslands in HU. This absence of management effect in HU may have been due to the fact that, compared to the Western European agricultural grasslands, land-use intensity was low on both Hungarian field types (Kleijn and Sutherland, 2003; Primdahl et al., 2003; Kleijn et al., 2009). A general lack of management effects was also observed on the same HU fields for other insect taxa that are strongly dependent on plant diversity or vegetation structure (grasshoppers and weevils; Batáry et al., 2007b,c), whereas habitat specialists were negatively affected by intensive management (grassland specialist leaf-beetles and birds; Batáry et al., 2007a,b). Yoshihara et al. (2008) found a general decline in the species richness of both insect-pollinated plants and pollinators in response to increased grazing intensity on the similarly unfertilised and unsprayed Mongolian steppe. However, in that study the most intensive grazing regime had almost three times as much cattle as the least intensive.

The differences in grazing intensity in HU, however, did result in differences in species composition of insect-pollinated plants. This was corroborated by the significantly higher cover of plant species with short flowering periods or with flowering periods starting in June on extensively grazed fields. Thus, even if small increases in grazing intensity do not result in declining species richness, it can cause changes in the plant composition (Loeser et al., 2007).

The number of bee species per field in HU was similar to that observed in CH but when corrected for differences in abundance, species richness in HU was more than twice as high (Fig. 1a and b). The high species evenness in Hungary is illustrated by the fact that the most dominant species (*B. terrestris*), contributed just 9% to the total bee abundance. On the other hand, in CH, the most dominant species (*A. mellifera*) made up to 74% of the total number of bees. In general, *B. terrestris* and *A. mellifera* were in the four most dominant pollinator species in all three countries despite the large differences in land-use intensity. These two generalist bee species might compete with the other native species (Goulson, 2003), which were less suppressed by them in the most extensive and diverse country (HU). This may suggest that increasing the intensity of land-use results in lower species evenness. In the long term, this might lead to a reduction of the species diversity rather than a shift towards a community consisting of better adapted species. Moreover, similar to Biesmeijer et al. (2006) and Ebeling et al. (2008), our results suggest that bees and insect-pollinated plants tend to decline in tandem. This might indicate that agricultural practice, which supports high species richness and cover of insect-pollinated plants, would be favourable to bees (Pywell et al., 2006; Potts et al., 2009). Additionally, AES can affect the plant-pollinator community from either direction. Management affecting plants will affect the resources available for pollinators. Similarly, management affecting the pollinator community e.g. through restrictions on pesticide applications, may have a knock-on effect on the plant community.

Although our study was by no means a rigorous test in the sense that we did not investigate the landscape scale effects within countries, our findings are in line with the hypothesis of Tscharntke et al. (2005) that effects of conservation management are most pronounced in landscapes with intermediate management intensity. No effects of conservation management were observed in areas supporting the most (HU) and the least diverse bee communities (NL). Significant effects were only observed in CH, where both land-use intensity and bee species richness were intermediate.

AES in very intensively farmed areas such as NL are not effective in supporting insect-pollinated plant or bee communities (Kohler et al., 2007). Tscharntke et al. (2005) and Kohler et al. (2008) suggested that in these landscapes, AES need to be implemented in the vicinity of semi-natural areas which could act as a species source. The findings of this study highlight that in addition to regional factors, such as the regional species pool, local factors determine the success of AES. The plant species on which bees depend need to be available as well. In HU, the country with the most diverse bee communities, bee species density was positively related to the diversity of insect-pollinated plants. This suggests, for example, that the effectiveness of the Swiss AES for pollinators could be even higher if the scheme would promote a higher diversity of insect-pollinated plants. This is in line with the findings of Albrecht et al. (2007), who observed that the number of small-sized and solitary bee species on Swiss AES and non-AES fields was strongly related to plant species richness. Maybe field scale effects of AES are most effective for species with lower mobility (such as small-sized bee species), which was shown in case of UK field margin moths (Merckx et al., 2009).In HU we found that both investigated management types supported diverse communities of insect-pollinated plants and bees. This suggests that management prescriptions limiting grazing intensities to 0.5–1.2 (animal/ha) and excluding the use of fertilisers and pesticides would make a valuable contribution to the recently initiated AES aiming to conserve this still highly diverse fauna. Kleijn et al. (2009) conclude that conservationists should invest more on these "intensification–prevention schemes" as it is easier to conserve than to reintroduce biodiversity. Since insect-pollinated plants showed the same results as bees across different grassland conservation management schemes, we suggest the evaluation of AES to focus on plant surveys because they can be carried out rapidly and cheaply by a non-invasive method, in addition to the target of the scheme and any red listed taxa.

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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.2009.11.004.

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