Effectiveness of agri-environmental management on pollinators is moderated more by ecological contrast than by landscape structure or land-use intensity

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Abstract

Agri-environment management (AEM) started in the 1980s in Europe to mitigate biodiversity decline, but the effectiveness of AEM has been questioned. We hypothesize that this is caused by a lack of a large enough ecological contrast between AEM and non-treated control sites. The effectiveness of AEM may be moderated by landscape structure and land-use intensity. Here, we examined the influence of local ecological contrast, landscape structure and regional land-use intensity on AEM effectiveness in a meta-analysis of 62 European pollinator studies. We found that ecological contrast was most important in determining the effectiveness of AEM, but landscape structure and regional land-use intensity played also a role. In conclusion, the most successful way to enhance AEM effectiveness for pollinators is to implement measures that result in a large ecological improvement at a local scale, which exhibit a strong contrast to conventional practices in simple landscapes of intensive land-use regions.

INTRODUCTION

Modern agriculture with widespread agrochemical use, simplification of landscape structure, short crop rotations and high mechanization has impacted biodiversity significantly, leading to severe pollinator declines around the world during the late 20th and 21th century (Kovács-Hostyánszki *et al.* 2017). As a solution for negative agricultural impacts on pollinators and on overall biodiversity, the first agri-environmental schemes or management options (hereafter AEM) were created in the EU member states during the 1980s (Batáry *et al*. 2015). Since 1992 AEM has become mandatory for all EU member states (European Commission 2005).

The different historical trajectories of European countries and regions led to large differences in heterogeneity between agricultural landscapes through different levels of agricultural intensification (Fuchs *et al.* 2015; van Vliet *et al.* 2015). Effectiveness of AEM for various taxa has been studied for almost three decades and generally has been related to landscape context and land-use intensity. Published results vary greatly. Birkhofer *et al.* (2014) did not find that regional land-use intensity moderates benefits of organic farming for biodiversity across Central and Northern Europe. Also AEMs effects on bumblebees species richness, abundance and species composition did not differ between two different land-use intensity regions in Estonia (Marja *et al.* 2014). However, Aviron *et al.* (2007) found significant AEM effect for grassland butterflies in intensive, but not in extensive management region. Thus effectiveness of different types of AEM is not straightforwardly related to land-use intensity.

AEM effectiveness can be moderated by landscape structure (Tscharntke *et al.* 2005, 2012). In the meta-analysis of Batáry *et al*. (2011), the authors found that AEM in cropland was more effective in simple (less than 20% semi-natural habitats) than in complex landscapes. Similar results were found in two follow-up meta-analyses (Scheper *et al*. 2013; Tuck *et al*. 2014) in that positive effects of organic management or AEM on biodiversity improved with an increasing amount of cropland in the landscape which is usually related to an increasing simplification of the landscape.

Kleijn *et al*. (2011) hypothesised that landscape structure and land-use intensity, together with the implemented management, are ultimately expressed in the ecological contrast that is created between fields with AEM and conventional control fields. For instance, the increase in floral resources produced by the establishment of wildflower strips on conventionally managed cereal field margins is relatively high (Scheper *et al.* 2015; Marja *et al*. 2018), resulting in large ecological contrasts between margins with and without such strips. On the other hand, delayed mowing of intensively managed grasslands only produces small ecological contrasts, because it results in negligible increases in floral resources compared to conventional management (Kleijn *et al*. 2011). Only a few studies have examined whether ecological contrast is indeed related to the effectiveness of AEM (Scheper *et al*. 2013; Hammers *et al*. 2015). Scheper *et al.* (2013) found that ecological contrast in floral resources created by AEM does indeed drive the response of pollinators to management. However, their data on testing contrast was limited to only one dataset (Kleijn *et al.* 2006). Hammers *et al.* (2015) tested the effect of contrast alone without considering other potential moderators.

According to the hypothesis of Kleijn *et al.* (2011), biodiversity responses are primarily determined by the ecological contrast between AEM and non-AEM sites and landscape structure, land-use intensity and type of management are merely determining the strength of the ecological contrast. If we find general evidence for this hypothesis, ecological contrast should be more strongly related to AEM effectiveness than either landscape structure or land-use intensity. So far, this has never been tested. Therefore, this is the first meta-analysis that investigates the relative importance of these inter-related moderators of AEM effectiveness concurrently. Our expectations are graphically depicted in Fig. 1. Based on previous literature we assume that all three examined factors (ecological contrast, landscape structure, land-use intensity) are not of equal importance for pollinator species richness and are not acting independently from each other. The effects of landscape structure include effects of land-use intensity and ecological contrast, and the effect of ecological contrast includes the effects of land-use intensity and landscape structure. However, in combination of these factors, we hypothesized the highest AEM effectiveness for pollinator species richness in case of large ecological contrast (vs. small contrast), simple landscape structure (vs. complex landscape) and intensive land-use (vs. extensive land-use) regions.

MATERIAL AND METHODS

Data collection and exclusion/inclusion criteria

We conducted literature searches using ISI Web of Science Core Collection (WoS) and Elsevier Scopus databases ranging 1945–2016 (last search date: 24 November 2016). We used the following keyword combinations according to the PICO (Population, Intervention, Comparator and Outcome) combination of search terms (Higgins & Green 2008), which were linked with logical operators to include the maximum number of relevant studies covering the effect of AEM on pollinator' richness. We used the following keywords combinations for literature search: TITLE-ABS-KEY (pollinat* OR bee OR bumble* OR hover* OR syrph* OR butterfly) AND TITLE-ABS-KEY(agri-environment* OR organic* OR integrated OR hedge* OR "field margin" OR fallow OR set-aside OR "set aside") AND TITLE-ABS-KEY (diversity OR richness) AND SUBJAREA(MULT OR AGRI OR ENVI) AND (EXCLUDE(DOCTYPE,"re")). Our literature searches confirm with the common review guidelines for a comprehensive literature review (Koricheva *et al*. 2013; Collaboration for Environmental Evidence 2018).

We combined two searches based on Web of Science and Scopus databases in Mendeley (Mendeley 2015) and removed duplicates. We found in a total of 653 potential studies. After screening those studies by title, 340 studies remained, and after reading the abstracts, 120 studies remained for full text screening. Additionally we used meta-analysis databases with similar topics (Batáry *et al*. 2011; Scheper *et al*. 2013; Tuck *et al*. 2014) and our unpublished datasets to locate further potential data. PRISMA flow diagram representing the detailed selection process (i.e. the number of studies identified, rejected and accepted) is presented in Fig. S1.

We used Europe for our study, since the majority of EU member countries have been under the same agri-environmental policies and most studies examining the effectiveness of AEM have been carried out here. In North America and Australia, agri-environmental policies are different, which complicates comparisons. We set up following criteria for inclusion and exclusion to filter out only European (EU28 + Switzerland + Norway) AEM pollinator species richness studies. Inclusion criteria were: study focusing on pollinator' absolute richness (hereafter species richness); including set-aside, but not abandoned grassland studies, which cannot be considered as a conservation action. Exclusion criteria were: not about agri-environment management; not a European AEM study; if the number of replicates (at field or farm level) was less than three in AEM or in control group; single field experiments (blocks within fields or within field margins), i.e. only taking studies at field level, since management actions are more relevant at those levels. Finally, we decided to exclude too broad scale studies covering too large area of given countries with different regions, because we were then unable to determine the regional land-use intensity effect. In total we found 62 studies with 156 data points (n=134 published, n=22 unpublished) for analysis, resulting in, on average, 2.5 data points per study, which is sufficient for meta-analyses. We provide studies with exclusion arguments in Appendix S1.

Classifications of ecological contrast, landscape structure and land-use intensity

We used three variables to test our hypotheses: ecological contrast, landscape structure and land-use intensity. We classified all studies in large vs. small ecological contrast, simple vs. complex landscape and intensive vs. extensive land-use intensity using the following procedures.

Ecological contrast was determined based on plant/flower richness or flower cover between AEM and control group given in the specific studies. We selected plant data, because it is a key driver predicting pollinator richness (Goulson 2003; Ebeling *et al.* 2008). We compared plant data results between AEM and control group (usually conventional farming) and determined ecological contrast (large or small). If plant data was not available (approximately 20% of the studies), we used the input amount of nitrogen between AEM and control group. High nitrogen applications are often the main negative driver of the richness of plant communities in agricultural landscapes (Kleijn *et al.* 2009; Soons *et al.* 2017; Midolo *et al.* 2019). We used the ecological contrast level of significance (statistical differences of plant/flower richness or cover data or nitrogen input between AEM and control group), or in cases this information was not available, also group means, provided in the studies. Finally, if neither plant data nor amount of nitrogen was available in a given study, we used our expert knowledge. RM and PB determined together case by case ecological contrast, based on information available on scheme descriptions in these studies (Table S1). We did not use any threshold or formula for ecological contrast determination.

We used the original GIS dataset from authors to determine study areas. If GIS data was not available, we identified the areas based on their description in the text (published coordinates) or map of study areas in original studies. If study area was poorly described and coordinates or maps of study areas were not provided, we visually examined the Google Earth aerial photos and determined study areas similarly as in a previous meta-analysis (Tuck *et al*. 2014). After a study area had been identified, we followed the approach of Tuck *et al*. (2014), and placed five random 1000 m transects per study area. The positions of the five transects were defined by sets of three randomly generated numbers. First, we generated the random number between zero (central study area measuring point) and the radius of the study area, denoted how many metres from the central point the starting point of each transect would be situated. Second, we randomly generated the angle degree defining the direction of the study area's central point for which the start point of the transect should be placed. With these two random numbers we were able to define the transect location. Third, we randomly selected numbers between 0, 45, 90 and 180 degrees to specify the angle at which the transect should be drawn, 500 m to each side of the start point. Transects were not allowed to cross or being closer to each other than 2000 m to avoid pseudoreplication in the landscape structure information. In each of the five random transects we collected landscape data in a buffer area of 1 km.

For landscape structure, we used the Coordination of Information on the Environment Land Cover databases from years 1990–2018 (hereafter CORINE database, Büttner *et al.* 2004). Since our used case studies are from the last three decades, we used landscape structure information based on the version of CORINE that was closest to the year of study. The 17 categories starting with CORINE database codes three or four indicate semi-natural habitats and were used to calculate the proportion of these within a radius of 1000 m (Batáry *et al*. 2011). We classified landscape structure as simple and complex landscapes (Tscharntke *et al*. 2005). In simple landscape, the proportional area of semi-natural habitats was less than 20%, in complex landscapes more than 20%. We did not consider the classification of a cleared landscape (<1%) since we had only 10 data points. We therefore added these points to the simple landscape classification.

We used the agricultural land-use intensity database (pixel 1×1 km) available for the EU to determine land-use intensity for each study area (Verburg 2016). For identifying regional scale land-use intensity data, we first used the previously digitized landscape scale transects, with which we created a new polygon with the minimum polygon method to get a more exact study area. We then classified land-use intensity in two groups: extensive or intensive agricultural region. The classification was based on the majority of pixels of the above GIS database in each study area. If majority of pixels represented extensive arable or extensive grassland or both, then it was classified as extensive region. Otherwise, we classified regional land-use intensity as intensive because the rest of the classification in the database represents intensive agriculture: moderately intensive arable, intensive grassland or very intensive arable. However, the Verburg (2016) database does not cover Switzerland, including fourteen different studies in our meta-analysis. Therefore for Switzerland, we used land-use information provided in the studies or if not, then we used online land-use database (Switzerland Federal Office of Topography 2016). We used a similar approach as with the previous database and determined land-use based on majority of cover either intensive or extensive land-use.

Effect size calculation

We used Hedges' *g* as a measure of effect size, which is the unbiased standardized mean difference (Hedges 1981; Borenstein *et al.* 2009). We calculated effect sizes and their non-parametric estimates of variance (formulas are presented in Appendix S2) for all data points based on the mean, standard deviation and sample size of pollinator species richness of AEM and control groups (Hedges & Olkin 1985). Effect size was positive if pollinator species richness was higher in the AEM than in the control group. To calculate Hedges' *g*, we obtained (from tables, graphs or text) the mean values, sample sizes and some variability measure of AEM and control groups (variance, SD, SEM or 95% CI).

Statistical analysis

For performing the meta-analysis models, we used the "metafor" (Viechtbauer 2010) package of the statistical program R (R Core Team 2018). We used hierarchical models with country, study ID and region or habitat as nesting factors with restricted maximum likelihood (Appendix S3). If one study presented two different groups of pollinators (for instance bumblebees and butterflies), we treated them separately in statistical analysis. First, we fitted a model without moderators to test the general effect of AEM compared to control group. Second, we fitted a model with moderators (ecological contrast, landscape structure and land-use intensity) to test which of them moderate the relative effectiveness of AEM for pollinator species richness the most (hereafter additive model). Additive models compare the relative effects between used moderators. Third, we fitted a model with ecological contrast, landscape structure and land-use intensity, including their three-way interaction, to test whether and how they interact with each other (hereafter interaction model). In the final model, we were interested, which of the possible eight combination is the most or least effective (Fig. 1). The interaction model estimates the average effect for each factor level combination. We described effect sizes (small, medium, large) based on Cohan’s benchmarks (Cohen 1988). We also calculated the variance inflation factor between moderators, and identified no values exceeding 1.4, which suggests that no collinearity between moderators occurred.

We also controlled outliers of effect sizes in our dataset. Based on the method of Habeck & Schultz (2015) we evaluated the sensitivity of our analyses by comparing fitted models with and without effect sizes that we defined as influential outliers. We defined influential outliers as effect sizes with hat values (i.e. diagonal elements of the hat matrix) greater than two times the average hat value (i.e. influential) and standardized residual values exceeding 3.0 (i.e. outliers; from Habeck & Schultz 2015). Our analysis showed, that there were no outliers in additive or in interaction models.

A potential publication bias were detected by funnel plot (Fig. S2), the regression test for funnel plot and fail-safe numbers. The regression test for funnel plot asymmetry indicated no significant publication bias (z = 1.39, p = 0.163). Additionally, we examined publication bias using Rosenthal's method of fail-safe number (Rosenthal 1979), which estimates the number of unpublished or non-significant studies that need to be added to analysis in order to change the results from significant into non-significant (Rosenberg 2005). Thus, the higher the fail-safe number, the more credibility a significant result has (Langellotto & Denno 2004). The model without moderators was significant (see results) and Rosenthal's fail-safe numbers calculation indicated that 33319 studies might be needed that AEM positive effect became non-significant. Hence, there was no sign of publication bias in our dataset. However, there was a geographical bias in our dataset, as most studies originated from Western or Northern Europe (Fig. S3).

RESULTS

Sixty-two studies (total 156 individual data points) or unpublished datasets fulfilled our selection criteria. Most studies were conducted in Western or Northern Europe (see a map in Fig. S3). We found only few studies from Southern or Eastern Europe.

Pollinator species richness benefitted from AEM. The summary random-effects model without moderators showed a large positive effect of AEM (effect size 0.83, CIs 0.69– 0.96, $p<0.001$). The additive model indicated that the moderation effect of ecological contrast was larger than that of landscape structure and that land-use intensity was not significant on pollinator species richness (Fig. 2).

Results of the interaction model showed of pollinator species richness related to the AEM with the highest effect size in case of the combination of large contrast, simple landscape and intensive land-use (Fig. 3). We also found large positive effects in studies with large contrast, complex landscape and intensive land-use. Medium effects appeared in studies with small contrast, simple landscape and intensive land-use studies. AEM was not effective for species richness in case of small contrast, complex landscape and intensive land-use. AEM was effective for species richness in case of large contrast, complex landscape and extensive land-use (Fig. 3). All other effect size values for extensive land-use indicated no significant AEM effect for pollinator species richness, but in some combinations had low sample sizes. General moderator trends were, that large contrast always had higher effect size than small contrast; simple landscape always had higher effect size than complex landscape (Fig. 2 and Fig. 3).

Comparison of additive and interaction models indicated no significant difference (p=0.35; likelihood-ratio test=4.4, AICc presented in Table 1).

DISCUSSION

Our meta-analysis documents for the first time that the effectiveness of AEM for pollinator species richness is more strongly related to local ecological contrast than to landscape structure or regional land-use intensity. The results showed the highest AEM effectiveness in intensive land-use regions and simple landscapes with large ecological contrast. Lowest effectiveness of AEM was found in extensive land-use regions, in complex landscapes and at sites with small ecological contrast.

Co-moderation of local, landscape and regional scale effects for pollinators

The additive model indicated that the ecological contrast created by the AEM at the site of implementation had the largest effect on pollinator species richness and that the structure of the surrounding landscape had a medium effect in moderating the AEM effectiveness. Regional land-use intensity had the weakest and non-significant effect on pollinator species richness. Thus, based on our additive model results, the following scale-dependency pattern of AEM effectiveness for pollinators can be determined: local > landscape > regional scale effect. Our model variance inflation values showed additionally that the moderators are independent from each other.

Our interaction model results indicated that large ecological contrast had in all cases (except when sample size was too small) significant positive effects on pollinator species richness. We determined in most cases ecological contrast by the difference between AEM and control sites in the amount of suitable flower resources providing energy and food for pollinators (Wood *et al.* 2015; Marja *et al*. 2018). Therefore, effective AEM, which is targeted to enhance pollinator diversity, should be determined first of all by the availability of food resources. Thus, large contrast AEM are probably most sustainable solutions for enhancing pollinator diversity in countries like Germany, France, United Kingdom, which are dominated by intensive land-use regions and simple landscape structure (but such regions are also common in Central and Eastern European countries). Since ecological contrast is co-moderated by landscape structure and land-use intensity, effective AEM in Western-European countries should also include measures to protect or create ecologically valuable landscape elements and habitats (species rich grasslands, set-asides, hedgerows, un-cropped areas), because food resources for pollinators as well as wintering and nesting habitats are highly important to enhance pollinator diversity.

We used semi-natural habitats to determine landscape complexity and our results indicated that landscape complexity enhances pollinator species richness probably via key resources such as availability of nesting and wintering habitats as well food resources (Kennedy *et al.* 2013). Comparing landscape structure effects on pollinator species richness (simple vs complex landscape) under the same ecological contrast and in the same land-use intensity regions, based on the interaction model, the AEM effectiveness was always stronger in simple than in complex landscape. Particularly, this was confirmed in intensive land-use regions. We found similar tendency also in extensive land-use regions, where AEM was more effective in simple than in complex landscapes, but in some cases, sample size was too small to confirm this pattern. Hence, especially ecological contrast, but also landscape structure, are important factors that need to be considered in agri-environment planning for enhancing pollinators diversity. However, current evidence suggests effect size is linearly related to ecological contrast (Scheper *et al*. 2013; Hammers *et al*. 2015). Dividing studies into groups with either high or low ecological contrast may, if anything, result in conservative estimates of the moderating effects of this factor.

Effectiveness of small ecological contrast

Based on our results, it is evident to conclude that AEM for pollinators should primarily consider local scale activities such as providing high quality and sufficient food resources (large ecological contrast conditions). In species-rich landscapes, small contrast AEM can also play an important role in conserving biodiversity, albeit indirectly. For instance, extensively used Hungarian puszta grasslands with complex landscape structure, alvar grasslands around Baltic Sea or alpine grasslands are currently often preserved largely because of support from agri-environmental subsidies despite the fact that species richness is rarely enhanced (e.g. Aavik *et al.* 2008; Batáry *et al.* 2015). Cessation of such small contrast AEM may lead to agricultural abandonment and enhance extinction probability of rare species with small populations (Batáry *et al.* 2010; Báldi *et al.* 2013). Thus, the value of small contrast AEM effectiveness comes only indirectly from its contribution to maintain high biodiversity systems.

AEM with small contrast in simple landscape and under intensive land-use conditions can also promote pollinator diversity, although only to a smaller extent. In those conditions, threatened or vulnerable species are often already lost or close to extinction and might disappear soon when intensive agricultural practice continues (Batáry *et al.* 2010). For that reason it is likely that small contrast AEM is not a viable option supporting pollinators under intensive land-use and simple landscape structure conditions, for instance in countries like Germany, the Netherlands and United Kingdom, where the species pool is already much impoverished.

Pollinator-related trade-offs with agricultural production

Since pollinators are important for ecosystems and humans, it is essential to protect pollinator diversity for sustainable crop production (Winfree *et al*., 2018). One solution for this objective is to develop new AEM that focus on large ecological contrast. However, this will be challenging because large ecological contrast AEM may be costly and unattractive for producers (Austin *et al*. 2015). For instance, creating and maintaining species-rich wildflower field margins needs costly investments in productive, but also in non-productive land. Therefore, economic-ecological trade-offs of AEM need to be identified in future research (Batáry *et al.* 2017; Kleijn *et al*., 2019). All AEM used in this study have been voluntary options for producers. Growers generally prefer AEM that can easily be incorporated into their daily farming practices. Small contrast AEM might be more popular and acceptable for producers, since they need fewer investments and are less expensive (Austin *et al.* 2015).

AEM beyond Europe

Previous research from Australia showed that, for instance, birds may benefit from AEM also used in Europe (Attwood *et al.* 2009). Furthermore, our results indicated that large contrast AEM in simple landscape supported much higher pollinator species richness than the control sites. Such open and wide areas are common in the intensive agricultural areas of North America and Australia. Therefore also in outside European regions, large ecological contrast AEM should be most effective to enhancing pollinator diversity.

CONCLUSIONS

We quantify for the first time how the effectiveness of AEM for enhancing pollinator richness depends on local ecological contrast, which is moderated by landscape structure and regional land-use intensity. Based on our results, maintaining or restoring pollinator diversity in a sustainable way with effective AEM needs to focus on landscape planning prioritizing mostly at local, but also at landscape and regional scales to effectively restore biodiversity and to safeguard ecosystem service functioning for the future (see Senapathi *et al.* 2015, Winfree *et al*. 2018). This means in practice that AEMs must increase first of all local plant and/or flowers diversity and density. In addition, maintaining natural vegetation species-rich areas as well as complex landscapes is also important to maintain large populations and high diversity of pollinators and other species. Only the combination of such different approaches can make up a comprehensive strategy to keep and promote pollinators across Europe. Future research should investigate how much ecological contrast is needed to predict that a target AEM is effective for biodiversity conservation.

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Table captions

Table 1 Summary table of meta-analyses showing tests of moderator, residual heterogeneities and models AICc.

Model	Moderators	*d.f.*	*Q*	*p*	AICc
Model without moderators		155	638.8	<0.001	414.5
Additive model	Residuals	152	537.6	<0.001	377.84
	Moderators	3	25.4	<0.001	
Interaction model	Residuals	148	528.5	<0.001	382.56
	Moderators	8	130.1	<0.001	

Figure captions

Figure 1 Graphical hypotheses of agri-environment management (AEM) effectiveness relation with ecological contrast, landscape structure and land-use intensity. In combination of those factors, darkest green indicates the strongest additive effect, and effectiveness decreases lightening of the green colour. White box indicate expected lowest effect based on hypotheses generated from Kleijn *et al.* (2011). Land-use intensity information is based on GIS data by Verburg (2016). On the left map, green colour represents extensive, whereas on the right map, brown colour represents intensive land use. The four photos on the left are an illustrative and actual examples of ecological contrast implementation. Photo credits for ecological contrast photos: Sinja Zieger and RM; for landscape structure photos: Estonian Land Board WMS service; for pollinator photos: RM.

Figure 2 The mean effect size (Hedges' *g*) of pollinator species richness in response to land-use intensity, landscape structure and ecological contrast as results of an additive model with 95% CIs range and significance values are presented. Explanatory variables indicate between group comparisons for land-use intensity (intensive vs. extensive; "Land-use"), landscape structure (simple vs. complex; "Landscape") and ecological contrast (large vs. small; "Contrast"). Asterisk symbols represent statistically significant p-values below 0.05, and 0.001 (* and *** respectively).

Figure 3 Mean effect size (Hedges' *g*) of pollinator species richness in response to the land-use intensity ("Extensive land-use, Intensive land-use"), landscape structure ("simple, complex") and ecological contrast ("Small, Large") on the effectiveness of agri-environment management (interaction model) with 95% CIs range and significance values are presented. Asterisk symbols represent statistically significant p-values below 0.05, 0.01, and 0.001 (*, **

and, *** respectively). Numbers indicate sample size. Darkest green indicates the strongest effect, and effectiveness decreases with lightening of the green colour.

Fig. 1.

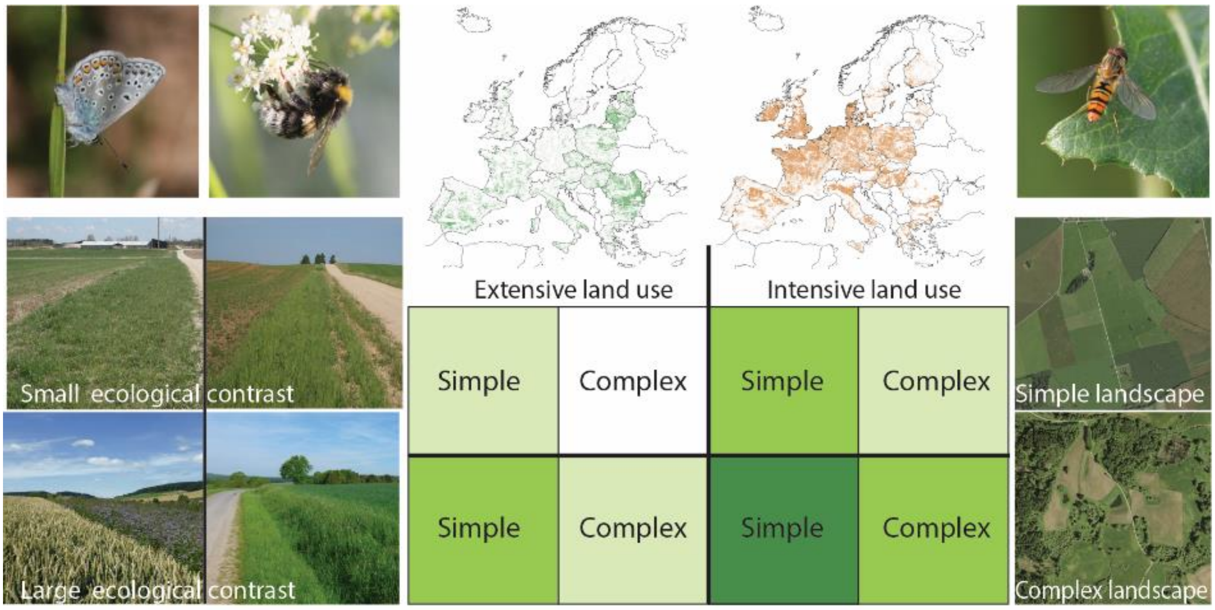

Extensive land use
Intensive land use
Simple
Complex
Simple
Complex
Simple
Complex
Simple
Complex
Small ecological contrast
Large ecological contrast
Simple landscape
Complex landscape

Fig. 2.

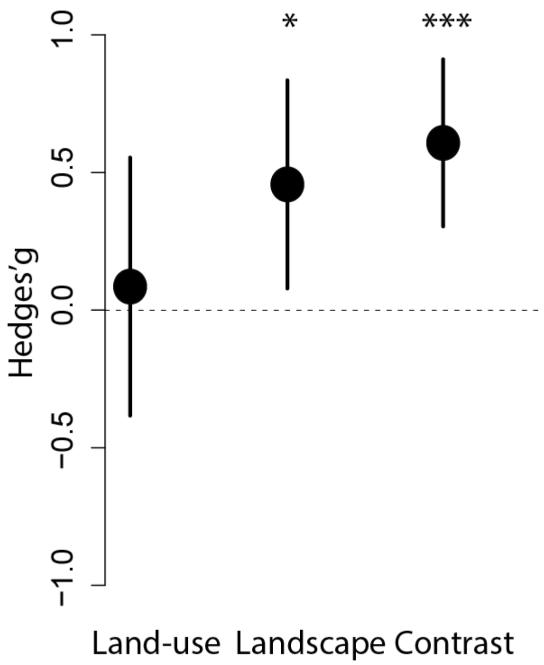
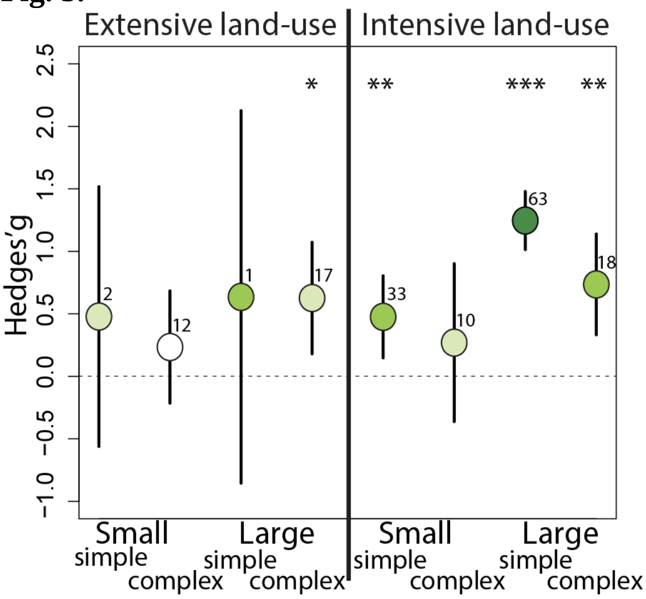


Fig. 3.


Extensive land-use
Intensive land-use
Hedges'g
−1.0
−0.5
0.0
0.5
1.0
1.5
2.0
2.5
*
**
***
**
2
12
1
17
33
10
63
18
Small
Large
Small
Large
simple
complex
simple
complex
simple
complex
simple
complex