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The effectiveness of agri-environment schemes for the conservation of farmland moths: assessing the importance of a landscape-scale management approach

Elisa Fuentes-Montemayor*, Dave Goulson and Kirsty J. Park

School of Biological and Environmental Sciences, University of Stirling, Stirling, Scotland FK9 4LA, UK

Summary

1. Agricultural intensification and expansion are regarded as major causes of worldwide declines in biodiversity during the last century. Agri-environment schemes (AES) have been introduced in many countries as an attempt to counteract the negative effects of intensive agriculture by providing financial incentives for farmers to adopt environmentally-sensitive agricultural practices.

2. We surveyed 18 pairs of AES and conventionally-managed farms in central Scotland (United Kingdom) to evaluate the effects of specific AES management prescriptions (field margins, hedgerows, species-rich grasslands and water margins) on farmland moths. We also measured the influence of the surrounding landscape on moth populations at three spatial scales (250 m, 500 m and 1 km radii from each trapping site) to assess at which scale management was most important for the conservation of farmland moths.

3. In general, percentage cover of rough grassland and scrub within 250 m of the trapping site was the most important landscape predictor for both micro- and macromoth abundance and macromoth species richness, although negative effects of urbanization were found at wider scales (within 1 km), particularly for macromoth species richness.

4. The abundance and species richness of micromoths was significantly higher within field margins and species-rich grasslands under AES management in comparison to their conventional counterparts, whereas AES water margins increased micromoth abundance, but not species richness. AES species-rich grasslands and water margins were associated with an increased macromoth abundance and species richness, and macromoths considered 'widespread but rapidly declining' also gained some benefits from these two AES prescriptions. In contrast, hedgerows under AES management enhanced neither micromoth nor macromoth populations.

5. *Synthesis and applications*. Our findings indicate that increasing the percentage cover of semi-natural environment at a local scale (e.g. within 250 m) benefits both micro- and macromoth populations, and that the implementation of simple AES management prescriptions applied to relatively small areas can increase the species richness and abundance of moth populations in agricultural environments.

Key-words: agri-environment schemes, conservation management, declining species, landscape, Lepidoptera, micromoths

Introduction

Declines in farmland biodiversity during the last century have been widely attributed to the intensification and expansion of modern agricultural practices (Krebs *et al*. 1999; Robinson & Sutherland 2002). This is of particular concern in the United Kingdom (UK) where approximately 75% of land is classed as agricultural (DEFRA 2008). Agri-environment schemes (AES) have been introduced in Europe and North America as an attempt to reverse declines in farmland biodiversity by providing financial incentives for farmers to adopt less

*Correspondence author. E-mail: ef12@stir.ac.uk

intensive, environmentally-sensitive agricultural practices. Approximately 45% of agricultural land in the UK is under AES management (DEFRA 2008) and large amounts of money are spent by the government on these schemes. For instance, the European Union (EU) funded the UK Rural Development Programme 2007–2013 with nearly €9 billion to support sustainable agriculture throughout the countryside (DEFRA 2008). Despite these high financial inputs, the implementation of AES has had mixed results for different taxa (Kleijn *et al.* 2006). Monitoring and evaluation of these schemes is imperative to improve their efficiency and maximize the conservation outcomes. It has been argued that uncommon species rarely benefit from these schemes (Hole *et al.* 2005; Kleijn *et al.* 2006; but see Merckx *et al.* 2010a). It has also been suggested that the effectiveness of AES depends on species' mobility (Merckx *et al.* 2009a), and that a landscape-scale management approach may be more beneficial than small-scale AES prescriptions (Donald & Evans 2006; Merckx *et al.* 2009b).

Lepidoptera are one of the most abundant and diverse insect orders, including more than 150 000 named species (Chinery 1993). They are commonly, but arbitrarily, divided into macrolepidoptera (a group of moth families containing mostly large species, 'macromoths', plus all butterfly families) and microlepidoptera or 'micromoths' (a group of moth families comprising mostly smaller species), of which *ca.* 900 and 1700 species occur in the UK respectively (Chinery 1993; Townsend & Waring 2007). Moths are an important food resource for many species of birds, bats and small mammals (Vaughan 1997; Wilson *et al.* 1999). They are also considered a sensitive indicator group for biodiversity (Jennings & Pocock 2009). To date, ecological research and conservation efforts for Lepidoptera have been largely focused on butterflies, with relatively little attention given to macromoths and even less to the conservation status and habitat requirements of micromoths, which comprise a large proportion of most local lepidopteran assemblages (Bland & Young 1996; New 2004). Some authors consider that the most important threat to moths is habitat change, involving fragmentation and loss of prime natural and semi-natural vegetation biotopes through the expansion of modern agricultural practices (Conrad *et al.* 2004; New 2004). Over a period of 35 years, over two-thirds of 337 common and widespread macromoth species studied by Conrad *et al.* (2006) showed significant population declines in Britain. Furthermore, for some of these species the declines have been so severe that if International Union for Conservation Nature (IUCN) criteria were to be applied at a national scale, 71 species would be regarded as threatened (Conrad *et al.* 2006). These species have now been added to the UK Biodiversity Action Plan (BAP) priority species list, labelled as 'widespread and common, but rapidly declining – research only'. The BAP working group mentions the need for research to look at wide changes in the countryside that may be affecting their populations (Fox *et al.* 2006).

Previous studies have shown positive effects of organic farming on moth populations (Wickramasinghe *et al.* 2004; Taylor & Morecroft 2009). Organic farming currently represents a very small proportion of the land area in the UK (*ca.* 2%; DEFRA 2008), so it may offer limited benefits to these groups on a large scale. In contrast, land under agri-environment schemes (other than organic farming) covers a much larger proportion (45%) of the UK's agricultural areas and there is evidence that some of these schemes may also benefit moth populations (Merckx *et al.* 2009a,b; Taylor & Morecroft 2009). However, to the best of our knowledge, at a local scale, only AES field margins and hedgerow trees as management options have been assessed and these studies deal exclusively with macromoths (Merckx *et al.* 2009a,b). Furthermore, although Merckx *et al.* (2009b) show that the degree of landscape-scale AES uptake matters, the direct impact of land-use composition of the surrounding landscape on moth communities has rarely been assessed (but see Ricketts *et al.* 2001; Summerville & Crist 2004; Kivinen *et al.* 2006; Ekroos, Heliölä & Kuussaari 2010). Hence, Merckx *et al.* (2009a,b)'s recommendation for moving from a field/farm-scale to a landscape-scale approach for farmland moth conservation requires further testing. Given that micromoths are usually low mobility species (Nieminen, Rita & Uuvana 1999), it is likely that they might be most influenced by their immediate surroundings and hence show stronger responses to AES applied at local scales than high mobility species (e.g. certain macromoths such as Noctuids). As a result, the latter might be more affected by the surrounding landscape at relatively larger scales (Ricketts *et al.* 2001) and may require a wider-scale conservation strategy (Merckx *et al.* 2009a).

To the best of our knowledge only one published study has compared biodiversity metrics of AES, as operated within Scotland, to conventional farming, and this study focused on the availability of foraging and nesting habitat resources for bumblebee queens (Lye *et al.* 2009). Here, we assess the benefits (or otherwise) of four different AES management prescriptions and the influence of the surrounding landscape at three spatial scales on farmland moth communities in Scotland. We addressed three questions in particular:

1. Do farmland moths benefit from common AES and, if so, which specific AES management options have the greatest effect on farmland moth abundance and diversity?

2. Is a landscape-scale management approach important for the conservation of farmland moths and, if so, which is the more appropriate scale?

3. Does the effectiveness of AES differ between micro- (low mobility) versus macro- (high mobility) moth species and, if so, are different conservation strategies required to enhance micro- and macromoth populations?

Materials and methods

STUDY SITES

We used a paired survey design to quantify moth abundance and species richness on 18 pairs of AES and conventional farms in central Scotland between June and September 2008. We selected 18 farms participating in the Scottish Rural Stewardship Scheme (RSS) since 2004. Each farm incorporated at least three of the following

AES management prescriptions: (i) field margins or beetlebanks; (ii) hedgerows; (iii) water margins; and (iv) species-rich grasslands. The following descriptions have been modified from Anonymous (2006).

1. *Management of field margins or beetlebanks in arable fields*. This prescription aims to provide habitat for beneficial insects, and cover and food for birds. It involves the creation and management of strips between 1.5 m and 6 m in width sowed with a suitable mix of grass seed, which may be located around or across an arable field. Fertilizer, pesticide and grazing restrictions apply.

2. *Management of hedgerows*. This prescription is aimed at providing improved habitat for invertebrates, birds and small mammals. It targets existing hedgerows and involves restrictions on pesticide input. Gaps in the hedge must be filled in, the hedge bottom must not be mown, cutting is restricted to once every 3 years and timing restrictions apply.

3. *Management of water margins*. This prescription aims to protect water margins from erosion and permit development of tall waterside vegetation for the benefit of freshwater life, invertebrates, water voles, otters and bats. It targets land bordering still water or watercourses. The water margin must be at least 3 m wide and fertilizer, pesticide, mowing and grazing restrictions apply.

4. *Creation and management of species-rich grassland*. This prescription aims to convert arable or improved grassland to species-rich grassland for the benefit of pollinator species such as butterflies and bumblebees. Its creation involves the destruction of any previously existing grassland cover and the establishment of a new sward by sowing the land with a low productivity grass and herb mix. Fertilizer and pesticides input restrictions apply, and mowing and grazing are not allowed during the summer.

We paired each AES farm with nearby conventionally-managed farms to act as counterparts. These were not involved in any AES and are referred to as conventional farms hereafter. Each of these conventional farms was within 8 km of its corresponding AES farm, conducted similar farming activities (arable, pastoral or mixed; 7, 2, and 9 pairs of farms respectively) and was of similar size (difference within paired sites 63 ± 36 ha; mean ± standard error (SE)). In each conventional farm we selected conventionally-managed field margins, hedgerows and water margins to compare with the equivalent habitat features under AES management. The selection of conventionally-managed features was performed carefully to control for as many variables as possible other than AES management. Activities conducted in adjacent fields (pastoral or arable) and proximity to non-targeted features such as woodland and roads were considered in the pairing design. AES species-rich grasslands were compared to either improved pasture or crop fields in the conventional farms; the selection of either of these two habitats was based upon land use of the species-rich grassland prior to AES conversion.

SAMPLING METHODS

We sampled each farm once during the summer of 2008. Farms within a pair (one AES farm and its conventional counterpart) were surveyed simultaneously to minimize the effects of weather variation on insect abundance. Temperature and wind speed were recorded on each farm immediately before and after sampling. If temperature fell below 8 °C, wind force exceeded Beaufort scale 4, or heavy rain occurred, sampling was abandoned. Moths were caught using portable 6 W heath light traps (3 – 4 traps per farm, depending on the number of AES prescriptions present at each site) powered with 12 V batteries. The traps were ≥100 m apart from each other to prevent the light traps from interfering with each other (Dodd, Lacki & Rieske 2008; Merckx *et al.* 2009b). The traps were activated 15 min after sunset adjacent to each AES management prescription (or equivalent conventional feature) and switched off after 4 h using automatic timers. The light traps were then sealed and transported to the laboratory. The collected insects were euthanized by dropping a cotton pad soaked with ethyl acetate into each trap and left overnight. Micromoths were wrapped in tissue paper and placed in sample bottles for later identification; individuals were dissected to examine genitalia whenever species identification required it. Macromoths were pinned for later identification following Townsend & Waring (2007). Rarer moths were unavoidably killed along with other insects after collection. However, trapping took place during one night only at each site, which is unlikely to adversely affect populations. No species protected under the UK's Wildlife and Countryside Act 1981 (as amended) were collected.

LANDSCAPE ANALYSIS

Using data from OS MasterMap Topography Layer (EDINA Digimap Ordnance Survey Service), we used ArcGIS 9·2 (ESRI Inc. 2006) to create circles of 250 m, 500 m and 1 km radius around the location of each trap. We selected these three different scales because the smallest (250 m) covers the dispersal distances of low mobility moth species, whereas the largest (1 km) approximates an upper limit to dispersal distances of many moth species (Nieminen, Rita & Uuvana 1999; Doak 2000; Ricketts *et al.* 2001; Summerville & Crist 2004; Merckx *et al.* 2009a). We reclassified the feature classes from the topography layers into five categories (hereafter referred to as biotope types). These were: (i) urban areas (buildings, structures and roads); (ii) farmland (both AES and conventionally-managed); (iii) water (inland and tidal water); (iv) semi-natural environment (rough grassland and scrub); and (v) woodland (coniferous, deciduous and mixed trees and areas covered by scattered trees). We then used Fragstats 3·3 (McGarigal *et al.* 2002) to calculate a selection of landscape metrics for each biotope type within the circles, including the proportion of land covered, the number of patches, mean patch area, total edge density, area-perimeter ratio and Euclidean nearest neighbour distance. A Shannon diversity index taking into account the number of different biotopes and their proportional abundance was also computed as a measure of landscape heterogeneity.

DATA ANALYSIS

We calculated diversity indices for micro- and macromoths using PAST (Hammer, Harper & Ryan 2001). We selected the α log series diversity index because of its good discriminant ability, its low sensitivity to sample size and the fact that a number of previous studies have shown the index to be particularly suited to the description of moth populations (Taylor, Kempton & Woiwod 1976; Magurran 1988).

All statistical analyses were conducted using R version 2.10 (R Development Core Team 2009). Linear regression analyses were used to evaluate the effect of a selection of landscape parameters at different spatial scales (Table 1) on moth abundance, richness and diversity. We selected the parameter that explained the highest variation in moth communities (highest R^2 value) and included this as one of the potential explanatory variables in subsequent models. To avoid pseudo-replication caused by overlapping buffers within sites, one trap per farm was randomly selected to be included in this analysis. We then performed Generalised Linear Mixed-Effects Models (GLMMs; Bates & Maechler 2009; Zuur *et al.* 2009) to determine which of the

Table 1. Description and summary statistics (mean ± SE) of landscape metrics used for landscape analysis and GLMMs

		Scale		
Landscape metric	Description	250 m	500 m	1 km
Shannon diversity[a]	Index of landscape heterogeneity. Equals minus the sum, across all biotope types, of the proportional abundance of each biotope type multiplied by that proportion.	0·32 ± 0·02	0·45 ± 0·02	0·57 ± 0·02
% Urban	Percentage of the landscape comprised of buildings, structures and roads.	2·16 ± 0·22	2·41 ± 0·18	3·06 ± 0·21
% Farmland	Percentage of the landscape comprised of agricultural land.	89·34 ± 1·09	86·02 ± 1·02	82·19 ± 1·03
% Water	Percentage of the landscape comprised of inland and coastal water.	0·42 ± 0·15	0·51 ± 0·18	0·72 ± 0·14
% Semi-natural	Percentage of the landscape comprised of rough grassland and scrub.	2·75 ± 0·67	2·59 ± 0·44	2·65 ± 0·42
% Woodland	Percentage of the landscape comprised of coniferous, deciduous, mixed woodland and scattered trees areas.	5·33 ± 0·75	8·47 ± 0·80	11·38 ± 0·81

[a]Modified from McGarigal *et al.* (2002).

Table 2. Summary table showing significance values of the explanatory variables and the goodness of fit of the final GLMMs. Management = farms involved in agri-environment schemes *vs.* conventionally-managed farms. Habitat = field margins, hedgerows, species-rich grasslands (improved grassland/arable fields in conventional farms) and water margins. Farming activity = arable and mixed

	Management	Habitat	Farming activity	Management × Habitat	Management × Activity	Surrounding landscape[c]	Final model R^2 [d]
Micromoth abundance[a,b]	-	-	**< 0·001**	**< 0·001**	0·550	0·699	72·25%
Micromoth richness	-	-	0·136	**0·073**	0·253	**0·007**	71·95%
Micromoth diversity	0·855	0·713	0·102	0·937	0·499	0·106	NA
Macromoth abundance	-	-	-	**0·001**	**0·015**	**< 0·001**	82·94%
Macromoth richness	-	-	0·156	**0·071**	0·102	0·312	67·62%
Macromoth diversity	0·775	0·135	0·566	0·201	0·809	0·630	NA
Declining species abundance	-	-	-	**0·011**	**< 0·001**	**< 0·001**	72·20%
Declining species richness	0·635	0·696	0·255	0·237	0·111	0·306	NA

[a]A dash indicates that the significance of a factor was not assessed in the model given that it was involved in a significant interaction.
[b]Significance values in bold indicate that a factor has been included in the final GLMM.
[c]Percentage cover of semi-natural environment at the 250 m scale was included in all models except for: (i) micromoth richness, percentage cover of farmland at the 1 km scale was used; (ii) micromoth diversity, percentage cover of urban areas at the 1 km scale was used and (iii) macromoth diversity, percentage cover of water at the 250 m scale was used.
[d]Pseudo R^2 values for each model were calculated by correlating the values predicted by the final GLMMs (fitted values) with the observed data. This value was not available (NA) when none of the evaluated factors remained significant in the final model.

variables evaluated had the greatest effect on farmland moths (the response variables are listed in Table 2). The following factors were included in the starting models as potential explanatory variables (fixed effects): land management type (AES or conventional), habitat feature (field margin, hedgerow, water margin, species-rich grassland or their equivalent conventional features), farming activity (arable or mixed; pastoral farms were excluded from this analysis as our sample size was too small, $n = 8$ trap samples) and the landscape parameter with the highest R^2 value for each response variable (see above). Two-way interactions between land management type and habitat feature and between land management type and farming activity were also included in the models. 'Pair' was included in the models as a random effect (grouping variable) to account for the paired-site sampling design. A backwards step-wise approach to model simplification was adopted. All models were also assessed using Akaike's information criterion (AIC). For all response variables the model selected by the stepwise approach matched the model with the lowest AIC value. Where the response variables were counts (e.g. moth abundance and species richness) models were fitted using Poisson errors (or quasi-Poisson whenever the data were overdispersed); for continuous variables (e.g. macromoth diversity) we used a Gaussian error on log transformed data (Crawley 2007). Whenever a significant effect was found, pair-wise *post hoc* comparisons were conducted to assess differences between groups; signifi-

Table 3. Summary table showing moth abundance, species richness and diversity indices at agri-environmental (AES) and conventionally-managed habitat features

Habitat feature	n traps	Micromoth abundance	Micromoth richness	Micromoth diversity	Macromoth abundance	Macromoth richness	Macromoth diversity	Declining spp. abundance	Declining spp. richness
Field margins	30	74	25	15·32	501	47	12·8	56	11
AES field margins	15	57	24	19·1	294	34	10·02	24	6
Conventional field margins	15	17	8	6·97	207	38	13·79	32	10
Hedgerows	26	145	36	16·22	422	40	10·95	57	9
AES hedgerows	13	64	25	16·99	219	33	10·93	26	6
Conventional hedgerows	13	81	25	12·99	203	32	10·78	31	7
Water margins	34	171	34	14·76	734	57	14·75	92	13
AES water margins	17	113	25	11·38	498	48	13·41	65	7
Conventional water margins	17	58	24	23·27	236	44	16·46	27	12
Species-rich grasslands	32	199	34	12·62	637	52	13·7	65	11
AES species-rich grasslands	16	156	24	8·40	366	46	14·38	44	10
Conventional species-rich grasslands	16	43	19	16·29	271	33	10·05	21	9
AES farms	61	390	51	16·77	1377	71	16·11	159	13
Conventional farms	61	199	43	18·61	917	61	14·88	111	17
All farms	122	589	61	18·18	2294	81	16·56	270	17

cance values were assessed using the Bonferroni method for multiple comparisons (Sokal & Rohlf 1995).

Results

We collected a total of 589 micromoths and 2294 macromoths from 122 trap samples (Table 3). A total of 61 micromoth species belonging to the families Blastobasidae, Coleophoridae, Crambidae, Elachistidae, Gelechiidae, Oecophoridae, Pterophoridae, Pyralidae, Tortricidae and Yponomeutidae, and 81 macromoth species from the families Noctuidae, Geometridae, Arctiidae and Hepialidae were identified (Appendix S1 in Supporting Information). Seventeen macromoth species sampled are classed as 'widespread but rapidly declining species' (hereafter referred to as 'declining macromoth species') and are of special conservation concern within Britain (Conrad *et al*. 2006; Fox *et al*. 2006).

EFFECTS OF THE SURROUNDING LANDSCAPE

A correlation matrix of all landscape metrics obtained from the landscape analysis showed that, for each biotope type, most are significantly correlated with each other ($P < 0{\cdot}05$ in 87% of cases). For instance, at the 250 m scale the percentage of land covered by semi-natural environment was significantly correlated with number of patches, mean patch area, total edge density and Euclidean nearest neighbour distance (all $P < 0{\cdot}05$) but not with area-perimeter ratio. As a result, we selected the proportion of each of five biotope types and a landscape heterogeneity index (Shannon diversity) at the three spatial scales as representative variables to be used in subsequent analyses. The following results are based on 36 traps (one trap per farm, randomly selected whilst ensuring that each AES prescription or equivalent conventional feature was represented by the same number of traps). However, results were largely unchanged when all traps ($n = 122$) were included in the landscape analysis.

Micromoth abundance was positively related to the percentage cover of semi-natural environment ($t = 3{\cdot}1$, df = 35, $P = 0{\cdot}004$, $R^2 = 38{\cdot}49\%$; Fig. 1a) at the 250 m scale. None of the landscape metrics at either 500 or 1000 m scale was significantly related to micromoth abundance. Micromoth species richness was positively related to landscape heterogeneity ($t = 2{\cdot}1$, df = 35, $P = 0{\cdot}048$, $R^2 = 7{\cdot}55\%$) whilst negatively related to the percentage cover of farmland ($t = 2{\cdot}0$, df = 35, $P = 0{\cdot}049$, $R^2 = 8{\cdot}20\%$, Fig. 1b) at the 1 km scale, although the amount of variation in species richness explained by these landscape parameters was relatively low. Micromoth diversity was not significantly related to any of the landscape parameters (Fig. 1c).

A very high proportion of the variation in macromoth abundance was positively explained by the percentage of semi-natural environment at the 250 m scale ($t = 3{\cdot}4$, df = 35, $P = 0{\cdot}002$, $R^2 = 41{\cdot}73\%$; Fig. 1d). Macromoth species richness was also positively related to the percentage cover of semi-natural environment at the 250 m scale ($t = 3{\cdot}5$, df = 35, $P = 0{\cdot}001$, $R^2 = 36{\cdot}91\%$; Fig. 1e), whilst negatively (but not significantly) related to the percentage cover of urban areas at the 1 km scale ($t = 1{\cdot}8$, df = 35, $P = 0{\cdot}088$, $R^2 = 11{\cdot}38\%$). Macro-

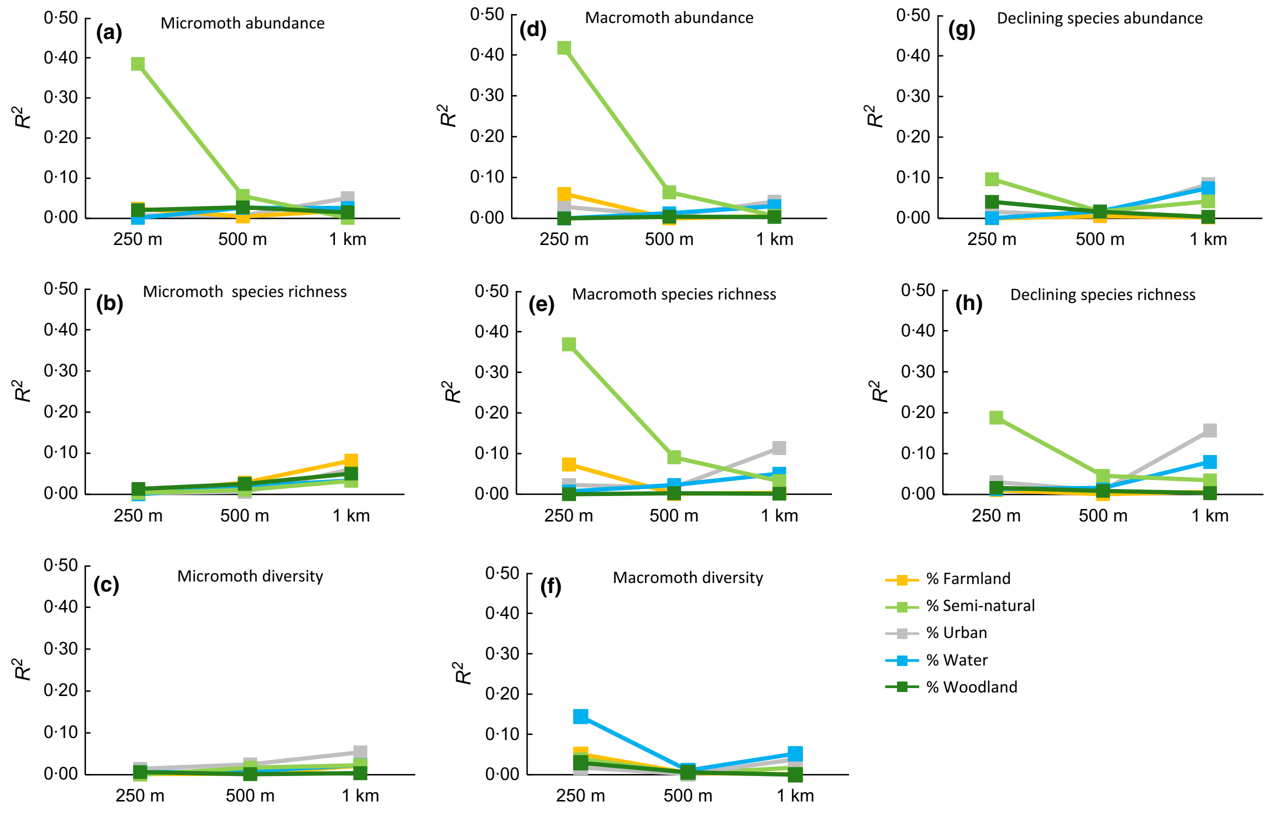

Fig. 1. R^2 values obtained from regression analyses between the percentage of land covered by each biotope type at three spatial scales and: (a) micromoth abundance, (b) micromoth species richness, (c) micromoth diversity, (d) macromoth abundance, (e) macromoth species richness, (f) macromoth diversity, (g) declining macromoth species abundance and (h) declining macromoth species richness. To avoid pseudo-replication caused by overlapping buffers within sites, one trap per farm was randomly selected to be included in this analysis, therefore $n = 36$. Significance values are discussed in the text.

moth diversity was positively related to the percentage of water at the 250 m scale ($t = 2{\cdot}4$, df = 35, $P = 0{\cdot}022$, $R^2 = 14{\cdot}41\%$; Fig. 1f).

Declining macromoth species showed a similar response to the surrounding landscape as all macromoth species combined. Their abundance showed the strongest response to the percentage cover of semi-natural environment at the 250 m scale, although this was not statistically significant ($t = 1{\cdot}4$, df = 35, $P = 0{\cdot}165$, $R^2 = 9{\cdot}63\%$; Fig. 1g). Declining macromoth species richness was positively related to the percentage cover of semi-natural environment at the 250 m scale ($t = 2{\cdot}2$, df = 35, $P = 0{\cdot}034$, $R^2 = 18{\cdot}77\%$; Fig. 1h); and negatively (but non-significantly) to the percentage cover of urban areas at the 1 km scale ($t = 1{\cdot}8$, df = 35, $P = 0{\cdot}078$, $R^2 = 15{\cdot}61\%$).

The landscape analysis consistently indicated that the percentage cover of semi-natural environment within 250 m of the sampling site was the most important predictor for both micro- and macromoth abundance and macromoth species richness. This parameter was included as a potential explanatory variable in the subsequent models, except for: (i) micromoth species richness; (ii) micromoth diversity and (iii) macromoth diversity, where the percentage cover of farmland within 1 km, urban areas within 1 km, and water within 250 m, were respectively selected as the best potential landscape predictors to include in the models.

EFFECTS OF AGRI-ENVIRONMENT SCHEMES

All final models explained a large proportion of the variation observed within the datasets, except for macromoth diversity and declining macromoth species richness models, where none of the variables included were significant (Table 2). *Post hoc* analyses for significant factors are summarized in Table 4. Significance values given in the text are not corrected for multiple comparisons, but they remained significant in all cases after using the Bonferroni correction method. The magnitude of the differences (based on median values) between AES prescriptions and conventionally-managed features is also shown.

In general, moth abundance and species richness were higher in farms participating in AES than in non-participating farms (Table 3). For micromoth abundance, there was a significant interaction between land management type (AES *vs.* conventional) and habitat feature (Table 2). More micromoths were found adjacent to AES field margins ($z = 4{\cdot}782$, $P < 0{\cdot}001$, $3{\cdot}7 \times$ more), water margins ($z = 4{\cdot}789$, $P < 0{\cdot}001$, $2{\cdot}2 \times$ more) and species-rich grasslands ($z = 7{\cdot}940$, $P < 0{\cdot}001$, $4{\cdot}0 \times$ more) than on their conventional counterparts, but no differ-

Table 4. Summary table showing *post hoc* analyses for: (a) pair-wise comparisons between agri-environmental prescriptions and conventionally-managed features (interaction between management and habitat type), and (b) pair-wise comparisons for the interaction between management type (agri-environmental *vs.* conventional) and farming activity (arable and mixed). Only the models with significant interactions are shown. Negative values indicate that moth abundance/richness was lower in conventionally-managed features/farms (with bold font indicating where this is significant at α 0·05). Significance values shown are not corrected for multiple comparisons, but they remained significant in all cases after using the Bonferroni correction method

	Field margins		Hedgerows		Species-rich grasslands		Water margins	
	Estimate[a]	SE	Estimate	SE	Estimate	SE	Estimate	SE
(a)								
Micromoth abundance	**−1·330*****	0·278	0·039	0·174	**−1·381*****	0·174	**−0·788*****	0·164
Micromoth richness	**−1·249*****	0·378	−0·170	0·248	**−0·706***	0·286	−0·413	0·244
Macromoth abundance	−0·174	0·120	0·099	0·127	0·098	0·114	**−0·347****	0·112
Macromoth richness	−0·027	0·166	−0·014	0·167	**−0·565*****	0·171	−0·206	0·141
Declining species abundance	−0·220	0·303	−0·490	0·317	**−1·009****	0·325	**−1·340*****	0·312

	Arable		Mixed	
	Estimate[a]	SE	Estimate	SE
(b)				
Macromoth abundance	**−0·439 *****	0·098	−0·174	0·120
Declining species abundance	**−1·079 ****	0·330	−0·022	0·467

[a]significance codes: '***' $P \leq 0{\cdot}001$, '**' $P \leq 0{\cdot}01$ and '*' $P \leq 0{\cdot}05$.

ence was observed between AES and conventionally-managed hedgerows (Table 4a & Fig. 2a). The same interaction was also significant for micromoth species richness, with more species present at AES field margins ($z = 3{\cdot}463$, $P < 0{\cdot}001$, 3·8 × more) and species-rich grasslands ($z = 2{\cdot}565$, $P = 0{\cdot}010$, 2·3 × more) than at their conventional counterparts (Fig. 3a).

There was also a significant interaction between land management type (AES *vs.* conventional) and habitat feature for macromoth abundance (and marginally significant for macromoth species richness), with higher abundances recorded at AES water margins ($z = 3{\cdot}110$, $P = 0{\cdot}002$, 1·6 × more) and more species collected at AES species-rich grasslands ($z = 3{\cdot}313$, $P = 0{\cdot}001$, 1·8 × more) than at their conventionally-managed counterparts (Figs. 2b and 3b). No differences were observed between AES and conventionally-managed hedgerows or field margins. The abundance of declining macromoth species showed the same significant interaction between land management type and habitat, again with AES water margins ($z = 4{\cdot}298$, $P < 0{\cdot}001$, 1·5 × more) and species-rich grasslands ($z = 3{\cdot}111$, $P = 0{\cdot}002$, 1·1 × more) having higher abundance than their conventional counterparts (Fig. 2c). The number of declining macromoth species collected at each farm (on any habitat) was not affected by participation within AES.

Farming activity had a significant effect on micromoth abundance, with more micromoths being collected at arable farms than at mixed ones (Table 2). For macromoth abundance, there was a significant interaction between land management type and farming activity, with the effect of adopting agri-environment schemes being noticeable on arable farms but not in mixed farms (Table 4b & Fig. 4b). This was also true for the abundance of declining macromoth species (Fig. 4c).

Discussion

In this study we assessed the value of AES as they currently operate in Scotland and the influence of the surrounding landscape (up to 1 km from trapping site) on assemblages of both macromoths and the relatively poorly studied micromoths. The diversity of the latter group and the fact that it comprises a substantial proportion of lepidopteran assemblages highlights its potential to yield relevant information to land managers (New 2004).

The results derived from the landscape analysis revealed that moth populations are enhanced by a high proportion of nearby semi-natural environment (rough grassland or scrub). Kuussaari *et al.* (2007) also found that semi-natural grasslands benefit lepidopteran communities. Micromoth abundance was significantly related to the percentage cover of semi-natural environment within 250 m of the collection site, but this parameter was not a significant predictor when included in the final explanatory model. Although the percentage of farmland cover within 1 km was significantly related to micromoth species richness, this variable only explained a very small amount (< 10%) of the variation in the data. This indicates that micromoths are influenced mainly by nearby habitat features and suggests that some of the currently operating AES prescriptions applied to relatively small areas are enhancing micromoth populations. Macromoth abundance and species richness were both also most strongly influenced by the percentage cover of semi-natural environment within 250 m of the trap. Given that most micromoths usually have lower dispersal abilities than macromoths (Nieminen, Rita & Uuvana 1999), it is somewhat surprising that the response of the two groups to the surrounding landscape was similar, although the negative

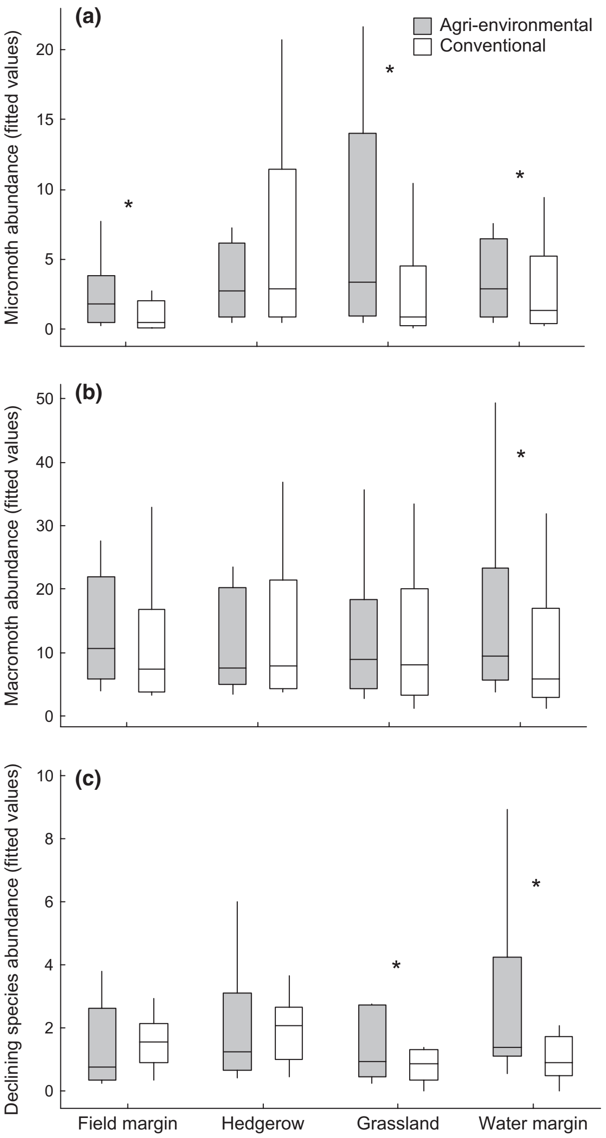

Fig. 2. Fitted values predicted by the final GLMMs showing the effect of the interaction between management type and habitat feature on the abundance of: (a) micromoths, (b) macromoths and (c) declining macromoth species. Stars indicate significant differences within a habitat feature due to management type.

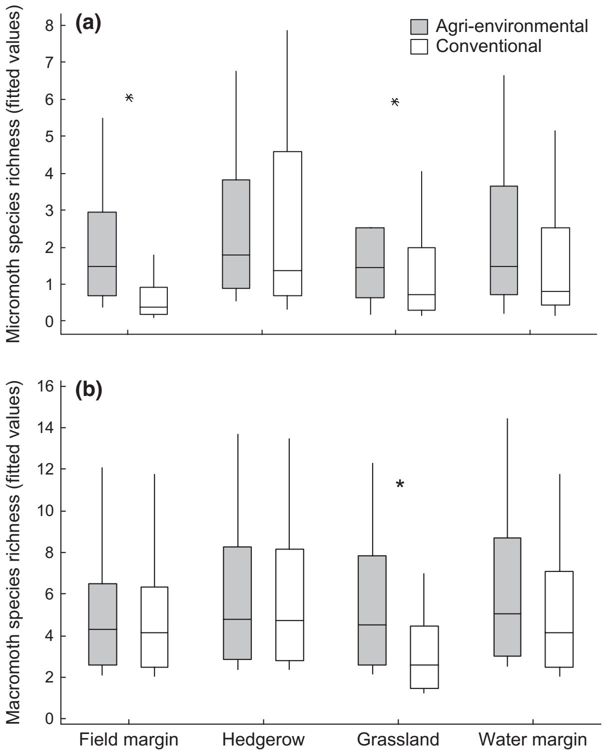

Fig. 3. Fitted values predicted by the final GLMMs showing the effect of the interaction between management type and habitat feature on species richness of: (a) micromoths and (b) macromoths. Stars indicate significant differences within a habitat feature due to management type.

effects of urbanization for macromoth (and declining macromoth) species richness at a wider scale of 1 km was higher, in general, than for micromoths.

Our assessment of Scottish AES revealed that, in general, the abundances and species richness of both micro- and macromoths were higher on farms involved in agri-environment schemes than on conventionally-managed farms. Most of the specific AES prescriptions assessed (except hedgerows) had at least some positive effects on moth populations. Our results concur with those of Taylor & Morecroft (2009) who investigated the impact of the implementation of AES on an English farm and found that moth abundance and species richness significantly increased over a 12-year monitoring period which started prior to the implementation of the schemes. However, since this study focuses at the farm level (e.g. assessing overall abundance at the farm) we cannot use it to evaluate specific prescriptions. Furthermore, their results cannot clearly be attributed to the implementation of AES given that these were implemented at their study site at the same time that conversion to organic farming took place (Taylor & Morecroft 2009).

We evaluated the effects of the implementation of specific AES management prescriptions and found that water margins and species-rich grasslands showed the most general benefits for all groups. The presence of agri-environmentally managed field margins promoted only micromoth (but not macromoth) populations, whereas hedgerows under AES management did not offer any benefit over conventionally-managed hedges for micromoths or macromoths.

AES management of water margins consistently increased the abundance of micro- and macromoths (including rapidly declining species). These wide grassy strips (≥3 m) look fairly similar to AES field margins, but often show higher structural complexity (e.g. taller non-woody vegetation, presence of shrubs and young trees; EFM pers. obs.) which might be beneficial for insect communities (Dennis, Young & Gordon 1998; Kuussaari *et al.* 2007). They also differ from AES field margins in that AES water margins management prescriptions do not involve the sowing of a seed mix and so the vegetation associated with them results from natural regeneration. This suggests that simple and inexpensive AES management options may fulfil the habitat requirements of some farmland moths.

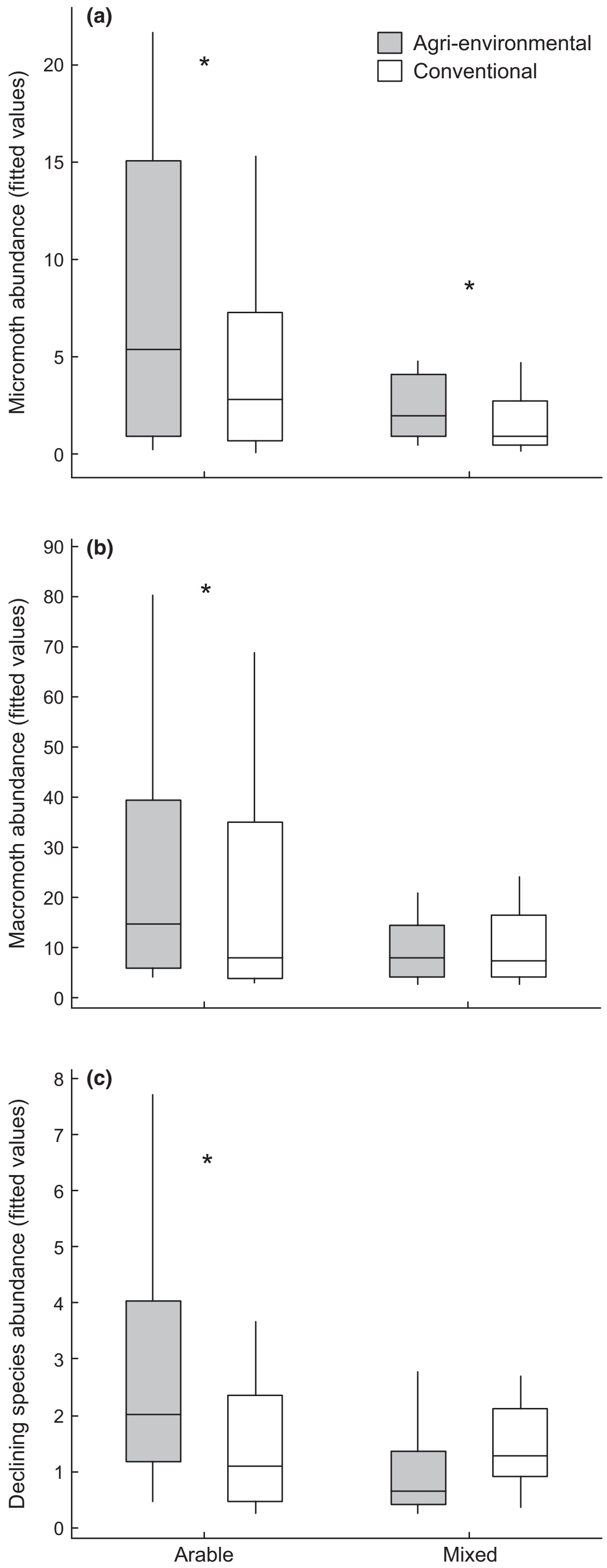

Fig. 4. Fitted values predicted by the final GLMMs showing the effect of the interaction between management type and farming activity on the abundance of: (a) micromoths, (b) macromoths and (c) declining macromoth species. Stars indicate significant differences within arable or mixed farms due to management type. Pastoral farms were excluded from this analysis due to small sample size.

Agri-environmentally managed species-rich grasslands also resulted in higher moth abundance and species richness than their conventional counterparts. The vegetation of this habitat is taller than that of its conventional equivalent (Lye *et al.* 2009) and this increased structural diversity could be providing moths with shelter and protection from predators in addition to providing more feeding niches. This is supported by Kuussaari *et al.* (2007), who found a positive effect of increased vegetation height of semi-natural grasslands on moth species richness.

Field margins managed under AES agreements increased the abundance and species richness of micromoths; this prescription increases plant species richness and vegetation height which may provide higher larval food plant availability and shelter from potential predators (Marshall, West & Kleijn 2006; Lye *et al.* 2009). In contrast, abundance, species richness and diversity of macromoths were similar to conventional field margins. This relates to the findings of Merckx *et al.* (2009a), who show that low mobility species (such as micromoths) exhibit stronger responses to the presence of this prescription than more mobile species.

Hedgerows under AES management did not offer any benefit over conventionally-managed hedges for micromoths or macromoths. Similarly, Lye *et al.* (2009) found that hedgerows under AES management were no more attractive to queen bumblebees (*Bombus* sp.) than conventional hedges, raising questions as to the value of this scheme option as it currently operates. Merckx *et al.* (2009b, 2010b) recommend the establishment and retention of hedgerow trees to be incorporated into AES hedgerow prescriptions, as it has the potential to increase macromoth abundance and diversity.

The effects of implementing AES management for both macromoths (all species) and declining macromoth species abundance was only significant on arable farms. Micromoth abundance and richness were higher at both AES arable and AES mixed farms than at their conventional counterparts, although more micromoths were collected at arable farms than at mixed ones. These effects could be due to the detrimental effects of grazing, which have been noted for moths and other insects in previous studies (Young & Barbour 2004; Pöyry *et al.* 2005; Littlewood 2008; Redpath *et al.* 2010). Grazing over the summer months does not allow for plants to flower and seed, and may therefore result in changes to vegetation composition and structure (Stewart & Pullin 2008). Even though most of the current AES prescriptions do incorporate restrictions regarding grazing regimes over the summer months, a farm-scale effect due to the presence of grazing stock in neighbouring fields may be limiting moth populations regardless of the operation of AES applied at a field scale. Therefore, the implementation of AES at larger scales, increasing not only the area but also the connectivity between patches of suitable habitat (e.g. species-rich grasslands), may be required as part of a more effective conservation strategy, and this might be particularly important in farms involving pastoral activities.

It has often been argued that differences observed between conventional and agri-environmental farms are not necessarily derived directly from the implementation of AES. Farms involved in these schemes might intrinsically be of higher environmental quality than conventionally-managed farms (Hole *et al.* 2005); also, farmers involved in AES may be more inclined to manage their land in an environmentally-friendly way than farmers who choose not to take part in such schemes (Kleijn & Sutherland 2003). These effects are difficult to disentangle, but the approach of Taylor & Morecroft (2009) in using a long-term study to follow the conversion of a conventional farm to an organic farm involved in AES suggests that some differences at least are due to the implementation of less intensive agricultural practices.

Some authors (e.g. Conrad *et al.* 2006) have highlighted the importance of monitoring population changes, not only of common species, but also of rare ones. In this study we show that declining macromoth species seem to respond to AES management prescriptions and to the surrounding landscape as do the rest of the macromoth species. Therefore, a conservation strategy beneficial to macromoth communities in general, would also benefit some species of special conservation concern. Conservation of natural habitats without specific focus on individual species has been regarded as an effective strategy because greater inclusive benefits may occur when focusing at the community level (New 2004).

In summary, our findings demonstrate that the implementation of current AES management prescriptions, targeted to relatively small areas, is an effective method to enhance both micro- and macromoth populations in agricultural environments. However, amendments are required to improve the performance of AES hedgerow management prescriptions and to minimize the detrimental effects of pastoral activities on farmland moths, where actions such as increasing the percentage cover of semi-natural environment in adjacent fields (within 250 m) may be required to maximize the benefits that moth populations gain from existing agri-environment schemes.

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References

Anonymous (2006) *The Rural Stewardship Scheme*. The Scottish Executive, Edinburgh.

Bates, D. & Maechler, M. (2009) *lme4: Linear mixed-effects models using S4 classes*. R package version 0.999375-32. http://CRAN.R-project.org/package=lme4.

Bland, K. & Young, M.R. (1996) Priorities for conserving Scottish moths. *Conserving Scottish Insects* (eds G.E. Rotheray & I. MacGowan), pp. 27–36. Edinburgh Entomological Club, Edinburgh.

Chinery, M. (1993) *Collins Field Guide: Insects of Britain and Northern Europe*, 3rd edn. HarperCollins, London.

Conrad, K.F., Woiwod, I.P., Parsons, M. & Fox, R. (2004) Long-term population trends in widespread British moths. *Journal of Insect Conservation*, **8**, 119–136.

Conrad, K.F., Warren, M.S., Fox, R., Parsons, M.S. & Woiwod, I.P. (2006) Rapid declines of common, widespread British moths provide evidence of an insect biodiversity crisis. *Biological Conservation*, **132**, 279–291.

Crawley, M.J. (2007) *The R Book*. John Wiley & Sons, West Sussex.

DEFRA (2008) Agriculture in the United Kingdom. Department for Environment Food and Rural Affairs, Department of Agriculture and Rural Development (Northern Ireland), Welsh Assembly Government – The Department for Rural Affairs and Heritage, The Scottish Government – Rural and Environment Research and Analysis Directorate.

Dennis, P., Young, M.R. & Gordon, I.J. (1998) Distribution and abundance of small insects and arachnids in relation to structural heterogeneity of grazed, indigenous grasslands. *Ecological Entomology*, **23**, 253–264.

Doak, P. (2000) Population consequences of restricted dispersal for an insect herbivore in a subdivided habitat. *Ecology*, **81**, 1828–1841.

Dodd, L.E., Lacki, M.J. & Rieske, L.K. (2008) Variation in moth occurrence and implications for foraging habitat of Ozark big-eared bats. *Forest Ecology and Management*, **255**, 3866–3872.

Donald, P.F. & Evans, A.D. (2006) Habitat connectivity and matrix restoration: the wider implications of agri-environment schemes. *Journal of Applied Ecology*, **43**, 209–218.

EDINA Digimap Ordnance Survey Service. OS MasterMap Topography Layer. http://edina.ac.uk/digimap, accessed February 2010.

Ekroos, J., Heliölä, J. & Kuussaari, M. (2010) Homogenization of lepidopteran communities in intensively cultivated agricultural landscapes. *Journal of Applied Ecology*, **47**, 459–467.

ESRI Inc. (2006) ArcGIS 9.2. http://www.esri.com.

Fox, R., Conrad, K.F., Parsons, M.S., Warren, M.S. & Woiwod, I.P. (2006) *The State of Britain's Larger Moths*. Butterfly Conservation and Rothamsted Research, Wareham, Dorset.

Hammer, O., Harper, D.A.T. & Ryan, P.D. (2001) PAST: Paleontological Statistics software for education and data analysis. *Palaeontologia Electronica*, **4**, 9.

Hole, D.G., Perkins, A.J., Wilson, J.D., Alexander, I.H., Grice, P.V. & Evans, A.D. (2005) Does organic farming benefit biodiversity? *Biological Conservation*, **122**, 113–130.

Jennings, N. & Pocock, M.J.O. (2009) Relationships between sensitivity to agricultural intensification and ecological traits of insectivorous mammals and arthropods. *Conservation Biology*, **23**, 1195–1203.

Kivinen, S., Luoto, M., Kuussaari, M. & Helenius, J. (2006) Multi-species richness of boreal agricultural landscapes: effects of climate, biotope, soil and geographical location. *Journal of Biogeography*, **33**, 862–875.

Kleijn, D. & Sutherland, W.J. (2003) How effective are European agri-environment schemes in conserving and promoting biodiversity? *Journal of Applied Ecology*, **40**, 947–969.

Kleijn, D., Baquero, R.A., Clough, Y., Diaz, M., De Esteban, J., Fernandez, F., Gabriel, D., Herzog, F., Holzschuh, A., Johl, R., Knop, E., Kruess, A., Marshall, E.J.P., Steffan-Dewenter, I., Tscharntke, T., Verhulst, J., West, T.M. & Yela, J.L. (2006) Mixed biodiversity benefits of agri-environment schemes in five European countries. *Ecology Letters*, **9**, 243–254.

Krebs, J.R., Wilson, J.D., Bradbury, R.B. & Siriwardena, G.M. (1999) The second silent spring? *Nature*, **400**, 611–612.

Kuussaari, M., Heliölä, J., Luoto, M. & Pöyry, J. (2007) Determinants of local species richness of diurnal Lepidoptera in boreal agricultural landscapes. *Agriculture, Ecosystems & Environment*, **122**, 366–376.

Littlewood, N.A. (2008) Grazing impacts on moth diversity and abundance on a Scottish upland estate. *Insect Conservation and Diversity*, **1**, 151–160.

Lye, G., Park, K., Osborne, J., Holland, J. & Goulson, D. (2009) Assessing the value of Rural Stewardship schemes for providing foraging resources and nesting habitat for bumblebee queens (Hymenoptera: Apidae). *Biological Conservation*, **142**, 2023–2032.

Magurran, A.E. (1988) *Ecological Diversity and Its Measurement*. Princeton University Press, Princeton, NJ.

Marshall, E.J.P., West, T.M. & Kleijn, D. (2006) Impacts of an agri-environment field margin prescription on the flora and fauna of arable farmland in different landscapes. *Agriculture, Ecosystems & Environment*, **113**, 36–44.

McGarigal, K., Cushman, S.A., Neel, M.C. & Ene, E. (2002) *FRAGSTATS: Spatial Pattern Analysis Program for Categorical Maps*. University of Massachusetts, Amherst.

Merckx, T., Feber, R., Dulieu, R.L., Townsend, M.C., Parsons, M.S., Bourn, N.A.D., Riordan, P. & Macdonald, D.W. (2009a) Effect of field margins on moths depends on species mobility: field based evidence for landscape-scale conservation. *Agriculture, Ecosystems and Environment*, **129**, 302–309.

Merckx, T., Feber, R., Riordan, P., Townsend, M.C., Bourn, N.A.D., Parsons, M.S. & Macdonald, D.W. (2009b) Optimizing the biodiversity gain from agri-environment schemes. *Agriculture, Ecosystems and Environment*, **130**, 177–182.

Merckx, T., Feber, R., Parsons, M.S., Bourn, N.A.D., Townsend, M.C., Riordan, P. & Macdonald, D.W. (2010a) Habitat preference and mobility of *Polia bombycina*: are non-tailored agri-environment schemes any good for a rare and localised species? *Journal of Insect Conservation*, **14**, 523–533.

Merckx, T., Feber, R., Mclaughlan, C., Bourn, N.A.D., Parsons, M.S., Townsend, M.C., Riordan, P. & Macdonald, D.W. (2010b) Shelter benefits less mobile moth species: the field-scale effect of hedgerow trees. *Agriculture, Ecosystems and Environment*, **138**, 147–151.

New, T.R. (2004) Moths (Insecta: Lepidoptera) and conservation: background and perspective. *Journal of Insect Conservation*, **8**, 79–94.

Nieminen, M., Rita, H. & Uuvana, P. (1999) Body size and migration rate in moths. *Ecography*, **22**, 697–707.

Pöyry, J., Lindgren, S., Salminen, J. & Kuussaari, M. (2005) Responses of butterfly and moth species to restored cattle grazing in semi-natural grasslands. *Biological Conservation*, **122**, 465–478.

R Development Core Team (2009) *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria.

Redpath, N., Osgathorpe, L.M., Park, K. & Goulson, D. (2010) Crofting and bumblebee conservation: the impact of land management practices on bumblebee populations in northwest Scotland. *Biological Conservation*, **143**, 492–500.

Ricketts, T.H., Daily, G.C., Ehrlich, P.R. & Fay, J.P. (2001) Country side biogeography of moths in a fragmented landscape: biodiversity in native and agricultural habitats. *Conservation Biology*, **15**, 378–388.

Robinson, R.A. & Sutherland, W.J. (2002) Post-war changes in arable farming and biodiversity in Great Britain. *Journal of Applied Ecology*, **39**, 157–176.

Sokal, R.R. & Rohlf, F.J. (1995) *Biometry: the principles and practice of statistics in biological research*, 3rd edn. W.H.Freeman, New York.

Stewart, G.B. & Pullin, A.S. (2008) The relative importance of grazing stock type and grazing intensity for conservation of mesotrophic 'old meadow' pasture. *Journal of Nature Conservation*, **16**, 175–185.

Summerville, K.S. & Crist, T.O. (2004) Contrasting effects of habitat quantity and quality on moth communities in fragmented landscapes. *Ecography*, **27**, 3–12.

Taylor, M.E. & Morecroft, M.D. (2009) Effects of agri-environment schemes in a long-term ecological time series. *Agriculture, Ecosystems and Environment*, **130**, 9–15.

Taylor, L.R., Kempton, R.A. & Woiwod, I.P. (1976) Diversity statistics and the log series model. *Journal of Animal Ecology*, **45**, 255–271.

Townsend, M. & Waring, P. (2007) *Concise Guide to the Moths of Great Britain and Ireland*. British Wildlife Publishing, Dorset.

Vaughan, N. (1997) The diets of British bats (Chiroptera). *Mammal Review*, **27**, 77–94.

Wickramasinghe, L.P., Harris, S., Jones, G. & Jennings, N.V. (2004) Abundance and species richness of nocturnal insects on organic and conventional farms: effects of agricultural intensification on bat foraging. *Conservation Biology*, **18**, 1283–1292.

Wilson, J.D., Morris, A.J., Arroyo, B.E., Clark, S.C. & Bradbury, R.B. (1999) A review of the abundance and diversity of invertebrate and plant foods of granivorous birds in northern Europe in relation to agricultural change. *Agriculture, Ecosystems and Environment*, **75**, 13–30.

Young, M.R. & Barbour, D.A. (2004) Conserving the new forest burnet moth (*Zygaena viciae*) in Scotland; responses to grazing reduction and consequent vegetation changes. *Journal of Insect Conservation*, **8**, 137–148.

Zuur, A.F., Ieno, E.N., Walker, N.J., Savaliev, A.A. & Smith, G.M. (2009) *Mixed Effects Models and Extensions in Ecology with R*. Springer Science + Bussines Media, New York.

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Appendix S1. List of moth species collected during study.

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